Fisheries New Zealand

Tini a Tangaroa

Aquatic Environment and Biodiversity Annual Review 2018

A summary of environmental interactions between the seafood sector and the aquatic environment

Growing and Protecting New Zealand

ACKNOWLEDGEMENTS

In addition to the thanks due to members of AEWG and BRAG working groups, special acknowledgement is due to researchers who contributed heavily to preparing initial and working draft chapters for Working Group discussion and finalisation. Noted contributors to existing chapters were David Thompson (Seabirds), Phil Sutton (Climate and Ocean) and Carolyn Lundquist (Biodiversity).

DISCLAIMER

This document is published by the Ministry for Primary Industries which was formed from the merger of the Ministry of Fisheries, the Ministry of Agriculture and Forestry, and the New Zealand Food Safety Authority in 2010 and 2011. All reference to the Ministry of Fisheries in this document should, therefore, be taken to refer to the legal entity, the Ministry for Primary Industries. The information in this publication is not government policy. While every effort has been made to ensure the information is accurate, the Ministry for Primary Industries does not accept any responsibility or liability for error of fat, omission, interpretation or opinion that may be present, nor for the consequence of any decisions based on this information. Any view or opinion expressed does not necessarily represent the view of the Ministry for Primary Industries.

PUBLISHER

Fisheries Science Team
Ministry for Primary Industries
Charles Fergusson Building, 34-38 Bowen Street
PO Box 2526, Wellington 6140
New Zealand
www.mpi.govt.nz
Telephone: 0800 00 83 83
Fascimile: +64 4 894 0300
ISBN (Print) 978-1-98-859482-8
ISBN (Online) 978-1-98-859499-6
© Crown Copyright June 2019 – Ministry for Primary Industries

Cover Image – Austin Burgess

Printed by Graphics Press and Packaging, Levin.

PREFERRED CITATION

Ministry for Primary Industries (2019). Aquatic Environment and Biodiversity Annual Review 2018. Compiled by the Fisheries Science Team, Ministry for Primary Industries, Wellington New Zealand. 704 p.

The 2018 edition of the Aquatic Environment and Biodiversity Annual Review, the eighth in the series, updates previous editions but does not expand coverage. It summarises information on a range of issues related to the environmental effects of fishing and aspects of marine biodiversity and productivity relevant to fish and fisheries. This review is a conceptual analogue of the Ministry's annual Fisheries Assessment Plenary reports. It summarises the most recent data and analyses on particular aquatic environment issues and, where appropriate, assesses current status against any specified targets or limits. Whereas the Fisheries Assessment Plenary reports are organised by fishstock, the Aquatic Environment and Biodiversity Annual Review is organised by issue (e.g. protected species, bycatch, benthic impacts etc.), and almost all issues involve more than one fishstock or fishery.

Three Fisheries New Zealand Science Working Groups contribute substantially to the Aquatic Environment and Biodiversity Annual Review. These are the Aquatic Environment Working Group (AEWG), the Antarctic Working Group (ANTWG) and the Biodiversity Research Advisory Group (BRAG). A wide variety of research is summarised in the Aquatic Environment and Biodiversity Annual Review, and some of this is peer-reviewed through processes other than Fisheries New Zealand science working groups. In particular, the Department of Conservation funds and reviews research on protected species, and the Ministry of Business, Innovation and Employment and Universities fund a wide variety of research, some of which is relevant to fisheries. Where such research is relevant to fisheries and meets MPI's Research and Science Information Standard, it is considered for inclusion in the review.

Further improvement of this review is anticipated over time and each chapter will be reviewed regularly and updated when required, although not necessarily every year. The appendix summarising aquatic environment, Antarctic and marine biodiversity research since 1998 will be regularly updated. The status of each Chapter is clearly stated in the first row of the overview table for each chapter e.g., This chapter has been updated for the AEBAR 2018. Data acquisition, modelling, and assessment techniques are progressively improving, and I anticipate that risk assessments and reference points to guide fisheries management decisions on environmental issues will be further developed over time. Both will lead to changes to the current chapters and the structure of the document. We hope the condensation of information from numerous reports published on a variety of platforms into this annual review will continue to assist fisheries managers, stakeholders and other interested parties to understand the issues, locate relevant documents, track research progress and make informed decisions. One addition trialled this year is the development of A3 sized summaries for three chapters (New Zealand's climate and oceanographic setting, New Zealand seabirds and biodiversity chapters) we welcome feedback on this initiative.

This revision has been led by the Science Group within the Directorate of Fisheries Science and Information in Fisheries New Zealand (primarily Rich Ford, William Gibson, Mary Livingston, Ben Sharp, Karen Tunley, Joshua van Lier, Marianne Vignaux and Nathan Walker), but has also relied on the input of members of the AEWG, ANTWG and BRAG, as well as the Department of Conservation's Conservation Services Programme Technical Working Group and other individuals who were commissioned to assist. I would especially like to recognise and thank the large number of research providers and scientists from research organisations, academia, the seafood industry, environmental NGOs, Māori customary, DOC and MPI, along with all other technical and non-technical participants in numerous AEWG, ANTWG, BRAG and CSP-TWG meetings for their substantial contributions to this review. My sincere thanks to each and all who have contributed.

I am pleased to endorse this document as representing the best available scientific information on the environmental effects of fishing and marine biodiversity, as at June 2019.



Rich Ford.

Science Manager

Science and Fisheries Information Directorate

Fisheries New Zealand

CONTENTS

PREFACE 3

1. INTRODUCTION – FULLY UPDATED 7
   - 1.1 Context and Purpose 7
   - 1.2 Legislation 7
   - 1.3 Policy Setting 9
   - 1.4 Science Processes 13
   - 1.5 References 14

2. RESEARCH THEMES COVERED IN THIS SECTION – NO UPDATES 15

3. SPATIALLY EXPLICIT FISHERIES RISK ASSESSMENT – PARTIALLY UPDATED 18
   - 3.1 Context 19
   - 3.2 Methods 22
   - 3.3 Model Inputs 37
   - 3.4 Alternative Implementations of the SEFRA Framework 52
   - 3.5 References 54

THEME 1: PROTECTED SPECIES 56

4. New Zealand Sea Lion (*Phocarctos hookeri*) – PARTIALLY UPDATED 57
   - 4.1 Context 57
   - 4.2 Biology 61
   - 4.3 Global Understanding of Fisheries Interactions 72
   - 4.4 State of Knowledge in New Zealand 72
   - 4.5 Indicators and Trends 93
   - 4.6 References 95

5. New Zealand Fur Seal (*Arctocephlaus fosteri*) – PARTIALLY UPDATED 101
   - 5.1 Context 101
   - 5.2 Biology 102
   - 5.3 Global Understanding of Fisheries Interactions 107
   - 5.4 State of Knowledge in New Zealand 107
   - 5.5 Indicators and Trends 118
   - 5.6 References 119

6. Hector's Dolphin (*Cehphalorhynchus hectori hectori*) and Māui Dolphin (*C. h. maui*) – NO UPDATES 123
   - 6.1 Context 123
   - 6.2 Biology 125
   - 6.3 Global Understanding of Fisheries Interactions 139
   - 6.4 State of Knowledge in New Zealand 139
   - 6.5 Indicators and Trends 160
   - 6.6 References 161

7. New Zealand Common Dolphin (*Delhinus delphis delphis*) – PARTIALLY UPDATED 166
   - 7.1 Context 166
   - 7.2 Biology 167
   - 7.3 Global Understanding of Fisheries Interactions 173
   - 7.4 State of Knowledge in New Zealand 174
   - 7.5 Indicators and Trends 186
   - 7.6 References 186

8. New Zealand Seabirds – FULLY UPDATED 190

Seabird Chapter Snapshot 190

8.1 Context 193
8.2 Biology 199
8.3 Global Understanding of Fisheries Interactions 200
8.4 State of Knowledge in New Zealand 201
8.5 Indicators and Trends 284
8.6 References 286

THEME 2: NON-PROTECTED BYCATCH 295

9. Fish and Invertebrate Bycatch – FULLY UPDATED 296
9.1 Context 297
9.2 Global Understanding 298
9.3 State of Knowledge in New Zealand 299
9.4 Indicators and Trends 338
9.5 References 340

10. Chondrichthyans (Sharks, Rays and Chimaeras) – FULLY UPDATED 343
10.1 Context 343
10.2 Biology 344
10.3 Global Understanding of Fisheries Interactions 345
10.4 State of Knowledge of Fisheries Interactions in New Zealand 346
10.5 Indicators and Trends 355
10.6 References 359

THEME 3: BENTHIC IMPACTS 363

11. Benthic (Seabed) Impacts – PARTIALLY UPDATED 364
11.1 Context 364
11.2 Global Understanding 365
11.3 State of Knowledge in New Zealand 373
11.4 Indicators and Trends 392
11.5 References 394

THEME 4: ECOSYSTEM EFFECTS 399

12. New Zealand's Climate and Oceanic Setting – FULLY UPDATED 400

Climate and Oceans Chapter Snapshot 400

12.1 Context 403
12.2 Indicators and Trends 406
12.3 Ocean Climate Trends and New Zealand Fisheries 420
12.4 References 422

13. Trophic and Ecosystem-Level Effects – NO UPDATES 426
13.1 Context 426
13.2 What Causes Trophic and Ecosystem-Level Effects? 431
13.3 What Types of Ecosystems are likely to be Most Affected? 435
13.4 Over what Spatial Scales do Trophic and Ecosystem-Level Change Occur? 438
13.5 How can Trophic and Ecosystem-Level Effects be Detected? 440
13.6 Discussion 445
13.7 Conclusions 447
13.8 References 448

14. Habitats of Particular Significance for Fisheries Management – NO UPDATES 461
14.1 Context 461
14.2 Global Understanding 462

14.3 State of Knowledge in New Zealand 465
14.4 Indicators and Trends 467
14.5 References 467

15. Land-Based Effects on Fisheries, Aquaculture and Supporting Biodiversity – NO UPDATES 471
15.1 Context 471
15.2 Global Understanding 473
15.3 State of Knowledge in New Zealand 475
15.4 Indicators and Trends 479
15.5 References 480

16. Ecological Effects of Marine Aquaculture – NO UPDATES 484
16.1 Context 484
16.2 Global Understanding 486
16.3 State of Knowledge in New Zealand 486
16.4 References 506

17. Antarctic Science – PARTIALLY UPDATED 512
17.1 Context 512
17.2 Ross Sea Ecosystem 518
17.3 Ecosystem Effects of Fishing in the Antarctic 522
17.4 Indicators and Trends 534
17.5 References 539

THEME 5: MARINE BIODIVERSITY 547

18. Biodiversity – FULLY UPDATED 548

Biodiversity Chapter Snapshot 548

18.1 Introduction 551
18.2 The Fisheries New Zealand Biodiversity Research Program 552
18.3 Marine Biodiversity Trends in New Zealand 552
18.4 Overall Progress in Fisheries New Zealand Marine Biodiversity Research 559
18.5 Evaluation of Research Effectiveness 570
18.6 Concluding Remarks 573
18.7 References 574

APPENDICES 583

19. Appendices – FULLY UPDATED 583
19.1 Terms of Reference for the Aquatic Environment Working Group (AEWG) for 2017 Onwards 583
19.2 Terms of Reference for the Biodiversity Research Advisory Group (BRAG) for 2017 Onwards 585
19.3 Terms of Reference for the Antarctic Working Group (ANTWG) for 2017 Onwards 587
19.4 Generic Terms of Reference for Fisheries Assessment Working Groups for 2017 Onwards 588
19.5 Membership and Protocols for all Science Working Groups 590
19.6 Fisheries 2030 594
19.7 Other Strategic Policy Documents 597
19.8 Appendix of Aquatic Environment and Biodiversity Funded and Related Projects 602
19.9 Appendices from Chapter 9 Fish and Invertebrate Bycatch 659
19.10 Appendices from Chapter 10 Chondrichthyans 680
19.11 References 687

1 INTRODUCTION

Status of Chapter	This chapter has been fully updated for AEBAR 2018.

1.1 CONTEXT AND PURPOSE

The Aquatic Environment and Biodiversity Annual Review (AEBAR) 2018 contains a summary of scientific information and research on aquatic environment issues relevant to the performance of New Zealand fisheries management that was available as of December 2018. The document is prepared annually to complement Fisheries New Zealand's (Fisheries NZ) annual Reports from Fisheries Assessment Plenaries (e.g., the May plenary, MPI 2018a, and the November plenary, MPI 2018b). Fisheries Assessment Plenaries report on the assessments of individual fish stocks, whereas the AEBAR reports on aquatic environment fisheries-related issues and biodiversity responsibilities that often cut across many fish stocks, fisheries, or activities, and sometimes across the responsibilities of multiple agencies.

The AEBAR has been developed by the Aquatic Environment Fisheries Science Team over many years at Fisheries NZ, and is updated and drafted each year with assistance from Working Group members (primarily the Aquatic Environment Working Group, AEWG; Biodiversity Research Advisory Group, BRAG) and selected research providers. As with the Reports from Fisheries Assessment Plenaries, it has already grown substantially since its first publication in 2011 and is expected to grow and evolve further as new information becomes available and more issues are considered. The 2018 edition has no new chapters. While Fisheries NZ aims to update as many chapters as possible each year, this is not always necessary. Chapters not updated are clearly marked as such.

The AEBAR has a broad, national focus on each environmental issue and the general approach is to discuss the issue at a national scale. For instance, the benthic (seabed) effects of mobile bottom-fishing methods are dealt with at the level of all such fisheries combined rather than at the level of a target fishery that, although it might be locally important, might contribute only a small proportion to the total impact. The details of benthic impacts by individual fisheries are documented in selected chapters in the May or November Report from the Fisheries Assessment Plenary, and linked there to the fine detail and analysis in research reports.

The first part of this document describes the legislative and policy context for aquatic environment and biodiversity research commissioned by Fisheries NZ, and the science processes used to generate and review that research. The second, and main part of the document contains chapters on key aquatic environment issues for fisheries management. Those chapters are under five broad themes: protected species; non-QMS (mostly fish) bycatch; benthic effects; ecosystem issues (including New Zealand's oceanic setting); and marine biodiversity. The final part of the review includes appendices for reference.

This review summarises the best available information on the issues covered. Each chapter has been reviewed by the appropriate working group at least once.

1.2 LEGISLATION

The primary legislation for the management of fisheries, including the effects of fishing on the aquatic environment, is the Fisheries Act 1996 (Table 1.1). The main guidance to avoid, remedy, or mitigate any adverse effect of fishing on the aquatic environment is in sections 8, 9, and 15, although sections 10, 11, and 13 are also relevant to decision-making under this Act (Table 1.2). Fisheries NZ also administers a range of other acts on behalf of the Ministry for Primary Industries (MPI) and there are some Acts administered by other agencies (Table 1.1) that lead to a requirement for Fisheries NZ to work with other government departments (especially Department of Conservation (DOC), the Ministry for the Environment

(MfE), the Natural Resource Sector[1]) and with various territorial authorities (especially Regional Councils) to a much greater extent than is required for most fisheries stock assessments.

Under the primary legislation lie various layers of Regulations and Orders in Council (see http://www.legislation.govt.nz/). It is beyond the scope of this document to summarise these.

In addition to its domestic legislation, the New Zealand government is a signatory to a wide variety of International Instruments and Agreements that bring with them various International Obligations (

Table 1.). Section 5 of the Fisheries Act requires that the Act be interpreted in a manner that is consistent with international obligations and with the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992.

Table 1.1: New Zealand Acts of environmental relevance to Fisheries NZ. * Denotes the Act of primary importance for the management of fisheries.

Fisheries NZ administers
Fisheries Act 1996*
Fisheries Act 1983 (residual parts)
Treaty of Waitangi (Fisheries Claims) Settlement Act 1992
Fisheries (Quota Operations Validation) Act 1997
Maori Fisheries Act 2004
Maori Commercial Aquaculture Claims Settlement Act 2004
Aquaculture Reform (Repeals and Transitional Provisions) Act 2004
Driftnet Prohibition Act 1991
Antarctic Marine Living Resources Act 1981
Requiring Fisheries NZ to work with others
Wildlife Act 1953
Marine Mammals Protection Act 1978
Marine Reserves Act 1971
Conservation Act 1987
Hauraki Gulf Marine Park Act 2000
Resource Management Act 1991
Exclusive Economic Zone and Continental Shelf Environmental Effects) Act 2012
Environmental Reporting Act 2015

[1] The Natural Resources Sector is a network of government agencies established to enhance collaboration. Its main purpose is to ensure a strategic, integrated and aligned approach is taken to natural resources development and management across government agencies. The network is chaired by the Chief Executive of the Ministry for the Environment.

Table 1.2: Sections of the Fisheries Act 1996 relevant to the management of the effects of fishing on the aquatic environment and biodiversity.

Fisheries Act 1996
s8 Purpose – (1) The purpose of this Act is to provide for the utilisation of fisheries resources while ensuring sustainability, where (2) "Ensuring sustainability" means – (a) Maintaining the potential of fisheries resources to meet the reasonably foreseeable needs of future generations: and (b) Avoiding, remedying, or mitigating any adverse effects of fishing on the aquatic environment: "Utilisation" means conserving, using, enhancing, and developing fisheries resources to enable people to provide for their social, economic, and cultural well-being. **s9 Environmental Principles.** associated or dependent species should be maintained above a level that ensures their long-term viability; biological diversity of the aquatic environment should be maintained: habitat of particular significance for fisheries management should be protected. **s10 Information Principles** All persons exercising or performing functions, duties, or powers under this Act, in relation to the utilisation of fisheries resources or ensuring sustainability, shall take into account the following information principles: a. decisions should be based on the best available information: b. decision makers should consider any uncertainty in the information available in any case: c. decision makers should be cautious when information is uncertain, unreliable, or inadequate: d. in the absence of, or any uncertainty in, any information should not be used as a reason for postponing or failing to take any measure to achieve the purpose of this Act. **s11 Sustainability Measures.** The Minister may take into account, in setting any sustainability measure, (a) any effects of fishing on any stock and the aquatic environment; **S13, 2b Total Allowable catch**. The Minister may set a TACC that enables the level of any stock whose current level is below that which can produce the maximum sustainable yield to be altered within a period appropriate to the stock, having regard to the biological characteristics of the stock and any environmental conditions affecting the stock; **S13, 2A b Total Allowable catch.** For the purposes of setting a total allowable catch under this section, if the Minister considers that the current level of the stock or the level of the stock that can produce the maximum sustainable yield is not able to be estimated reliably using the best available information, the Minister must have regard to the interdependence of stocks, the biological characteristics of the stock, and any environmental conditions affecting the stock; **s15 Fishing-related mortality of marine mammals or other wildlife.** A range of management considerations are set out in the Fisheries Act 1996, which empower the Minister to take measures to avoid, remedy or mitigate any adverse effects of fishing on associated or dependent species and any effect of fishing-related mortality on any protected species. These measures include the setting of catch limits or the prohibition of fishing methods or all fishing in an area, to ensure that such catch limits are not exceeded.

1.3 POLICY SETTING

1.3.1 STRATEGIC INTENTIONS AND OUR STRATEGY

Fisheries NZ is the principal adviser to the Government on fisheries and aquaculture. This is a business unit located in the Ministry for Primary Industries that also includes other business units which together have responsibilities for fisheries, agriculture, horticulture, forestry, food safety, animal welfare, and the protection of New Zealand's primary industries from biological risk, ie biosecurity. MPI's Strategic Intentions (Formerly called Statement of Intent, SOI), document is an important guiding document for the short to medium term. This document is available on the Fisheries NZ's website at: http://www.mpi.govt.nz/document-vault/9602

Aspects of the Strategic Intentions document (SI) of key relevance to fisheries include supporting the understanding of sustainable limits to natural resource use as part of Medium-Term Outcome 5 *The primary sector maximises the use and productivity of natural resources within*

Table 1.3 International agreements and regional agreements to which New Zealand is a signatory, that are relevant to the management of the effects of fishing on the aquatic environment. * The current NZ Biodiversity Strategy is currently undergoing a refresh to be launched in 2020.

International Instruments	Regional Fisheries Agreements
• **Convention on the Conservation of Migratory Species of Wild Animals (CMS).** Aims to conserve terrestrial, marine and avian migratory species throughout their range. • **Agreement on the Conservation of Albatrosses and Petrels (ACAP).** Aims to introduce a number of conservation measures to reduce the threat of extinction to the Albatross and Petrel species. • **Convention on Biological Diversity (CBD)** Provides for conservation of biological diversity and sustainable use of components. States accorded the right to exploit resources pursuant to environmental policies. • **United Nations Convention on the Law of the Sea (UNCLOS)** Acknowledges the right to explore and exploit, conserve and manage natural resources in the State's EEZ…with regard to the protection and preservation of the marine environment including associated and dependent species, pursuant to the State's environmental policies. • **Convention on the International Trade in Endangered Species of Wild Fauna and Flora (CITES).** Aims to ensure that international trade in wild animals and plants does not threaten their survival. • **United Nations Fishstocks Agreements.** Aims to lay down a comprehensive regime for the conservation and management of straddling and highly migratory fish stocks. • **International Whaling Commission (IWC)** Aims to provide for the proper conservation of whale stocks and thus make possible the orderly development of the whaling industry. • **Wellington Convention** Aims to prohibit drift net fishing activity in the convention area. • **Food and Agriculture Organisation – International Plan of Action for Seabirds (FAO-IPOA Seabirds)** Voluntary framework for reducing the incidental catch of seabirds in longline fisheries. • **Food and Agriculture Organisation – International Plan of Action for Sharks (FAO –IPOA Sharks)** Voluntary framework for the conservation and management of sharks. • **Noumea Convention.** Promotes protection and management of natural resources. Parties to regulate or prohibit activity likely to have adverse effects on species, ecosystems and biological processes. • **Food and Agriculture Organisation - Code of Conduct for Responsible Fisheries** Provides principles and standards applicable to the conservation, management and development of all fisheries, to be interpreted and applied to conform to the rights, jurisdiction and duties of States contained in UNCLOS.	• **Convention for the Conservation of Southern Bluefin Tuna (CCSBT)** Aims to ensure, through appropriate management, the conservation and optimum utilisation of the global Southern Bluefin Tuna fishery. The Convention specifically provides for the exchange of data on ecologically related species to aid in the conservation of these species when fishing for southern bluefin tuna. • **Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR).** Aims to conserve, including rational use of Antarctic marine living resources. This includes supporting research to understand the effects of CCAMLR fishing on associated and dependent species, and monitoring levels of incidental take of these species on New Zealand vessels fishing in CCAMLR waters. • **Convention on the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean (WCPFC).** The objective is to ensure, through effective management, the long-term conservation and sustainable use of highly migratory fish stocks in accordance with UNCLOS. • **South Tasman Rise Orange Roughy Arrangement.** The arrangement puts in place the requirement for New Zealand and Australian fishers to have approval from the appropriate authorities to trawl or carry out other demersal fishing for any species in the STR area • **Convention on the Conservation and Management of High Seas Fishery Resources in the South Pacific Ocean** (a Regional Fisheries Management Organisation, colloquially **SPRFMO**) has recently been negotiated to facilitate management of non-highly migratory species in the South Pacific. • **New Zealand Biodiversity Strategy 2000 and the Aichi Agreements*.** In 2016 New Zealand released an updated Biodiversity Action Plan which as goals and targets relevant to fisheries management

New Zealand's future prosperity and the natural capital that underpins New Zealand's production systems. The SI notes a growing demand for more support for the primary sector's market claims about product attributes that help capture premium prices, including assurances about sustainability in fishing. For these claims to be effective, they need to be underpinned by science-based, transparent, verifiable information.

Another important role is supporting third-party certification of fisheries by, for example, the Marine Stewardship Council as part of Medium-Term Outcome 1 *Export success is enhanced by the integrity of primary sector products and increasing the use of New Zealand's unique culture and brand*. New Zealand's export sectors derive significant benefits (including lower market access costs) and competitive advantage from New Zealand's reputation for safe and suitable food, favourable animal and plant health status and market assurances. To leverage these advantages, Fisheries NZ needs new ways of assisting New Zealand exporters to access and succeed in international markets and gain additional export value from the New Zealand brand, including its Māori dimension.

The Fisheries NZ's broad approach was updated in 2017 with a refresh of *Our Strategy 2030*. The new strategy was called *Our Strategy* (Figure 1.1) and is available on the Ministry's website at: http://www.mpi.govt.nz/about-mpi/our-strategy/. The Ministry's purpose is unchanged in *Our Strategy* as "growing and protecting New Zealand" but a new ambition is defined as "*New Zealand is the most trusted source of high value natural products in the world*". Four key outcomes are also outlined:

- Growth: New Zealand's food and primary sector grows the value of its exports;
- Sustainability: New Zealand's natural resources are sustainable, in the primary sector;
- Protection: New Zealand is protected from biological risk and our products are safe for all consumers; and
- Participation: New Zealanders participate in the success of the primary industries

To provide relevant information to fulfil these roles, Fisheries NZ contracts the following types of research (relevant to this document):

- aquatic environment research to assess the effects of fishing on marine habitats, protected species, non-target species of fish, and to understand habitats of special significance for fisheries;
- marine biodiversity and productivity research to increase our understanding of the systems that support resilient ecosystems and productive fisheries, including their trophic linkages and the effects of climate change.

1.3.2 FISHERIES PLANS

Fisheries planning processes for deepwater, highly migratory species, inshore finfish, inshore shellfish and freshwater fisheries use objective-based management to drive the delivery of services. The planning processes are guided by a series of National Fisheries Plans, which recognise the distinctive characteristics of these fisheries. The first National Plans for Deepwater and Highly Migratory species were approved by the Minister in September 2010 and a suite of three draft plans for inshore species was released in July 2011. Fisheries NZ is currently reviewing the plans and is, or will be, consulting on such reviews. Fisheries plans establish management objectives for each fishery, including those related to the environmental effects of fishing. All are available on the Fisheries NZ's websites together with a wide variety of other information on the management of these fisheries.

Deepwater and middle depth fisheries:
http://www.mpi.govt.nz/growing-and-harvesting/fisheries/fisheries-management/deepwater-fisheries/

Highly migratory species (HMS) fisheries:
http://www.mpi.govt.nz/growing-and-harvesting/fisheries/fisheries-management/highly-migratory-species/

Inshore fisheries (comprising finfish, shellfish, and freshwater fisheries):
http://www.mpi.govt.nz/growing-and-harvesting/fisheries/fisheries-management/inshore-fisheries/



Figure 1. The Ministry for Primary Industries' *Our Strategy, released in 2017*

Antarctic and other international (high seas) fisheries are not covered by fisheries plans but, rather, by the plans and strategies of the respective international organisations (CCAMLR, SPRFMO, WCPFC, CCSBT, etc.).

1.3.3 OTHER STRATEGIC DOCUMENTS

A number of strategies or reviews have been published that interface with fisheries values and fisheries research requirements. These include: the New Zealand Biodiversity Strategy (2000); the Biosecurity Strategy (2003, followed by its science strategy 2007 and more recently Biosecurity 2025); the MPA Policy and Implementation Plan (2005); MfE's discussion paper on Management of Activities in the EEZ (2007, now translated to the Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012); Fisheries 2030 (2009); MfE's Roadmap for Environment Science (2016); the Revised Coastal Policy Statement (2010); the National Plan of Action to Reduce the Incidental Catch of Seabirds in New Zealand Fisheries (2004, revised and updated by Fisheries in 2013); and the New Zealand National Plan of Action for the Conservation and Management of Sharks (2013); MfE and Stats New Zealand Environmental Reporting Act 2015, and New Zealand's Biodiversity Action Plan 2016. Links to these documents are provided in Appendix 17.8 .

In 2012, the Natural Resource Sector cluster formed a Marine Director's Group to improve data sharing and information exchange across key agencies with marine environmental responsibilities, particularly Fisheries NZ, DOC, MfE, EPA, LINZ, MBIE, STATS NZ.

1.4 SCIENCE PROCESSES

1.4.1 RESEARCH PLANNING

Fisheries NZ has adopted an overall approach of specifying management objectives for fisheries in Fisheries Plans and using these to develop implementation strategies and required services, including research. Services specific to a fisheries plan are identified in Annual Operational Plans that are updated each year (available via the links in section 1.3.2). Alongside this process, and in close consultation with fisheries managers and the Department of Conservation, Fisheries NZ also develops a portfolio of research on aquatic environment issues related to fisheries. This portfolio is designed to meet information needs that span multiple fisheries (e.g., incidental captures of seabirds across multiple fisheries in multiple areas) as well as the specified needs of individual fisheries plans. Marine biodiversity and productivity research has a much broader and more strategic focus and planning of such research is conducted with assistance from the Biodiversity Research Advisory Group (BRAG) chaired by Fisheries NZ.

1.4.2 RESEARCH REVIEW AND CONTRIBUTING WORKING GROUPS

Any research that is intended or likely to inform fisheries management decision-making must be reviewed against the requirements of the Research and Science Information Standard for New Zealand Fisheries (RSIS, 2011) (https://www.mpi.govt.nz/dmsdocument/3692-research-and-science-information-standard-for-new-zealand-fisheries).

The main contributing working groups for this document are Fisheries NZ's Aquatic Environment Working Group (AEWG), the Antarctic Working Group (ANTWG) and the Biodiversity Research Advisory Group (BRAG). The Department of Conservation's Conservation Services Programme Technical Working Group (CSP-TWG, see http://www.doc.govt.nz/our-work/conservation-services-programme/meetings-and-project-updates/) also considers a wide range of DOC-funded projects related to protected species, sometimes in joint meetings with the AEWG. Fisheries NZ Fishery Assessment Working Groups occasionally consider research relevant to this review where there is particular relevance to a fishery.

Terms of reference for Fisheries NZ working groups are periodically revised and updated (see Appendices 19.1–19.7 for those of working groups relevant to this document).

The AEWG is convened for Fisheries NZ peer review purposes with an overall purpose of assessing, based on scientific information, the effects of fishing, aquaculture, and enhancement on the aquatic environment for all New Zealand fisheries. The purview of AEWG includes: bycatch and unobserved mortality of protected species, fish, and other marine life; effects of bottom fisheries on benthic biodiversity, species, and habitat; effects of fishing on biodiversity, including genetic diversity; changes to ecosystem structure and function as a result of fishing, including trophic effects; and effects of aquaculture and fishery enhancement on the environment and on fishing. Where possible, AEWG may explore the implications of any

effects, including with respect to any standards, reference points, and relevant indicators. The AEWG is a technical forum to assess the effects of fishing or environmental status and make projections. It has no mandate to make management recommendations or decisions. Membership of AEWG is open (attendees for 2017 are listed in Appendix 19.2).

The ANTWG is convened for the Fisheries NZ's peer review purposes with an overall purpose of assessing, based on scientific information, the stock status and the effects of fishing for Antarctic fisheries. The purview of ANTWG includes: stock status of target species, bycatch and unobserved mortality of protected species, fish, and other marine life; effects on biodiversity and benthic biodiversity, species, and habitat; and changes to ecosystem structure and function as a result of fishing, and including trophic effects. The ANTWG also provides peer review of documents and papers submitted to the scientific working groups of CCAMLR to aid and inform its management. The ANTWG is a technical forum to assess the stock status, effects of fishing or environmental status, and make projections. It has no mandate to make management recommendations or decisions. Membership of ANTWG is open (attendees for 2017 are listed in Appendix 19.2).

The two main responsibilities of BRAG are: to review, discuss, and convey views on the results of marine biodiversity research projects contracted by the Fisheries NZ; and to discuss, evaluate, make recommendations and convey views on Medium Term Biodiversity Research Plans and constituent individual projects. Both tasks have evolved from the strategic goals in both the New Zealand Biodiversity Strategy (2000) and the Strategy for New Zealand Science in Antarctica and the Southern Ocean (2010). More recently, the programme has become aligned to emerging issues such as climate change and ocean acidification. BRAG has provided advice and oversight of some large cross-government survey projects such as NORFANZ, BIOROSS, Fisheries and Biodiversity Ocean Survey 20/20; and International Polar Year (IPY) Census of Antarctic Marine Life (IPY-CAML). Membership of BRAG is open and current attendees are listed in Appendix 19.2.

Following consideration at one or more meetings of appropriate working groups, final reports from individual projects are also technically reviewed by Fisheries NZ before they are finalised for use in management and/or for public release. Fisheries Assessment Reports, FARs, and Aquatic Environment and Biodiversity reports, AEBRs, are also subject to editorial review whereas Final Research Reports, FRRs, and Research Progress Reports, RPRs, are not. Finalised FARs, AEBRs, historical FARDs (discontinued Fisheries Assessment Research Documents) and MBBRs (discontinued Marine Biodiversity and Biosecurity Reports), and some FRRs can be found in the Document library at: http://fs.fish.govt.nz/Page.aspx?pk=61&tk=297. More recent reports are available from the Fisheries NZ website at: http://www.mpi.govt.nz/news-and-resources/publications/.

1.4.3 REVIEW OF RESEARCH NOT FUNDED BY FISHERIES NZFISHERIES NEW ZEALAND

Almost all research of direct relevance to management of fish stocks is commissioned by Fisheries NZ and reviewed through Fisheries NZ fishery assessment working groups. This is a structured approach to meeting the requirements of the RSIS. However, research on various aspects of the environmental effects of fishing is also commissioned by a range of external organisations and is commonly published in science journals. It is not always clear that the requirements of the RSIS have been met in these cases. Fisheries NZ working groups, including AEWG, can provide an excellent and well-informed forum to discuss such research, and researchers are encouraged to bring their work on the environmental effects of fishing to this forum for review and assessment against the requirements of the RSIS. Whether or not a working group has considered them, reports or journal papers that are intended or likely to inform fisheries management decision-making are technically reviewed by the Fisheries NZ's fisheries science team before they can be used.

1.5 REFERENCES

Ministry for Primary Industries (2017a). Fisheries Assessment Plenary, May 2017: stock assessments and stock status. Compiled by the Fisheries Science Group, Ministry for Primary Industries, Wellington, New Zealand. 1556 p.

Ministry for Primary Industries (2017b). Fisheries Assessment Plenary, November 2016: stock assessments and stock status. Compiled by the Fisheries Science Group, Ministry for Primary Industries, Wellington, New Zealand. 500 p.

Ministry of Fisheries (2011). Research and Science Information Standard for New Zealand Fisheries. Ministry of Fisheries, Wellington, New Zealand. 31 p.

2 RESEARCH THEMES COVERED IN THIS DOCUMENT

Status of chapter	This chapter has not been updated for AEBAR 2018.

The Ministry has identified five broad categories of research on the environmental effects of fishing (Figure 2.1): incidental capture and fishing-related mortality of protected species; bycatch of non-protected species, primarily non-QMS fish; modification of benthic habitats (including seamounts); and various ecosystem effects (including fishing and non-fishing effects on habitats of particular significance for fisheries management, effects of aquaculture on the environment and wild-capture fisheries, trophic relationships and Antarctic Science). A risk assessment chapter has also been added prior to the research themes, due to its cross-cutting nature, although this structure may be reconsidered in future. Other emerging issues (such as the genetic consequences of selective fishing) are not dealt with in detail in this edition but it is anticipated that those that turn out to be important will be dealt with in future iterations. The fifth theme for this document is MPI research on marine biodiversity. The research has been driven largely by the Biodiversity Strategy but has strategic importance for fisheries in that it provides for better understanding of the ecosystems that support fisheries productivity.

Our understanding is not uniform across these themes and, for example, our knowledge of the quantum and consequences of fishing-related mortality of protected species is much better developed than our knowledge of the consequences of mortalities of non-target fish, bottom trawl impacts, or land management choices for ecosystem processes or fisheries productivity. Ultimately, the goal of research described in this synopsis is to complement information on fishstocks to ensure that the Ministry has the information required to underpin the ecosystem approach to fisheries management envisaged in the future. Stock assessment results have been published for many years in Fisheries Assessment Reports, Final Research Reports, and the Annual Report from the Fishery Assessment Plenary ('the plenary'). Collectively, these provide a rich and well-understood resource for fisheries managers and stakeholders. In 2005, an environmental section was included in the hoki plenary report as part of the characterisation of that fishery and to highlight any particular environmental issues. Similar, fishery-specific sections have since been developed for several other fisheries and included in the plenary, but work on environmental issues has otherwise been more difficult to access for fisheries managers and stakeholders. The Ministry explored better ways to document, review, publicise, and integrate information from environmental assessments with traditional fishery assessments, including annual publication of this document. This will rely heavily on studies that are published in Aquatic Environment and Biodiversity Reports and Final Research Reports but, given the overlapping mandates and broader scope of work in this area, also on results published by other organisations and in the scientific literature. The integration of all this work into a single source document analogous to the Report from the Fishery Assessment Plenary has advanced considerably since the first edition in 2011 but it will take time for all issues to be included.



THEME	RESEARCH QUESTIONS	CURRENT WORK
1.PROTECTED SPECIES • Marine mammals • Seabirds • Turtles • Protected fish • Corals	• How many of each NZ-breeding protected species are caught and killed in our fisheries (and out of zone)? • How many unobserved deaths are caused? • What is the likely effect of fishing-related mortality on protected species populations? • Which species/populations are most at risk? • Which fisheries cause the most risk and where are the most cost-effective gains in mitigation to be made? • What mitigation approaches are most successful and in what circumstances? • What levels of bycatch would lead to different population outcomes?	• Estimation of annual bycatch of protected species by fishery • Abundance and productivity of key seabird populations • Abundance and productivity of Hector's & Māui dolphins • Semi-quantitative risk assessment for all seabirds • Semi-quantitative risk assessment for marine mammals • Full quantitative risk assessment for selected populations • Modelling to assess links between bycatch and population outcomes
2. OTHER BYCATCH • Non-QMS fish & invertebrates	• How much non-target fish is caught and discarded in our fisheries? • What is the effect of that bycatch? • What do trends in bycatch show?	• Continued monitoring cycle for deepwater and highly migratory • Data collection to allow improved risk assessment for sharks and Non-QMS fish
3. BENTHIC EFFECTS • Distribution of habitats & trawling • Effects of trawling on each	• What seabed habitats occur where in our TS/EEZ and how much of each is affected by trawling or shellfish dredging? • How sensitive is each habitat to disturbance and how do we lose (or gain) when each is disturbed? • What are the likely consequences of different management approaches? • What are benthic recovery times in different habitats?	• Testing habitat suitability models at a range of spatial scales • Monitoring and recovery rate of some key inshore habitats • Assessment of relative sensitivity of habitats • Mapping of deepwater and inshore trawl footprints • Development of a risk assessment
4. ECOSYSTEM EFFECTS • Trophic studies • Habitats of significance • Ecosystem indicators • Land-use effects • Climate variability • Climate Change • System productivity	• How do the ecosystems that support our fisheries function? • What are the key predator-prey or synergistic relationships in these systems? • Are our fisheries affecting food webs or ecosystem services? • What changes are occurring in the ecosystems that support our fisheries? • What is "habitat of particular significance for fisheries management"? • How do fisheries and/or land management affect fish habitat and fisheries production?	• Habitat of significance: Marlborough Sounds blue cod habitat • Monitoring and indicators of environmental change for deepwater fisheries • Much of the more strategic work in this theme is conducted under the biodiversity programme (see below)
5. MARINE BIODIVERSITY • Characterising NZ biodiversity • Functional ecology • Genetic diversity • Ocean climate • Metrics & indicators • Threats & impacts • Ross Sea	• What are the key drivers of pattern in New Zealand's marine biodiversity? • How does biodiversity contribute to the resilience of ecosystems to perturbation and climate change? • What are the major risks and opportunities from ocean-climate variability and trends? • What drives genetic connectivity within species? • What do we need to measure and monitor to assess risks and change? • How are biota adapted to polar conditions and what is their sensitivity to perturbation?	• Climate change risk & opportunity for the seafood industry • SPRFMO benthic habitats • Genetic connectivity between New Zealand and the Louisville Ridge • Ocean acidification modelling • Experimental response of shellfish to warming and acidification • Monitoring surface plankton • BPA biodiversity • Sublethal effects of OA on fish productivity • Coralline and macroalgae habitats

Figure 2.1: Summary of themes in the Aquatic Environment and Biodiversity Annual Review 2017. [Continued on next page]

CURRENT STATE OF KNOWLEDGE
• Aggregate "on deck" captures of seabirds (and approximate species composition), marine mammals, and large sharks known reasonably well for offshore trawl and longline fisheries, but less well for inshore fisheries (where observer coverage has historically been low). • Incidental, cryptic, or unobserved mortality are poorly known (and difficult to assess). • Factors affecting fishing related mortality are well known for most seabirds and marine mammals. • Knowledge of population abundance is increasing for some key seabird species and well known for sea lions, but poorly known or dated for other seabirds, some species of dolphins, fur seals and most sharks. • Semi-quantitative risk assessments on all seabird and marine mammal species, fully quantitative risk assessments have been completed for five seabird species, Hector's / Maui dolphins and sea lions. • Impact of fishing-related mortality on most protected species remains uncertain because of some key knowledge gaps. • Some methods of mitigating fishing-related mortality have been formally tested.
• Bycatch and discards are monitored and reported using observer records for the main deepwater and highly migratory fisheries. • Bycatch and discards for inshore vessels remain poorly known. • Some mitigation approaches have been assessed (e.g., for scampi trawl).
• Modelled predictions are available of the distribution of seabed habitats at a broad scale using classifications (BOMEC) and at finer scale for seamounts and some biogenic habitats. • Excellent understanding of the distribution of bottom trawling in offshore waters, good understanding in coastal waters, but poor understanding for most shellfish dredge fisheries. • Good understanding of the effects of trawling on some nearshore and deepwater habitats. • General understanding of the effects of trawling on biogeochemical processes. • General understanding of the relative sensitivity of different habitats. • Tools to design candidate spatial management measures have been developed and tested (including for High Seas fisheries)
• The nature and variability in the diets of key commercial species in the Chatham Rise ecosystem are reasonably well understood. • A preliminary trophic model of the Sub-Antarctic ecosystem suggests a low productivity system supporting a simple food chain with high transfer efficiencies. • The distribution of spawning, pupping, egg-laying, and juveniles of key species has been described in public atlases. • Land-based effects on fish habitat and coastal biodiversity has been reviewed and documented. • Basic monitoring of ecosystem change over time (through fish-based indicators calculated from trawl survey data and acoustic time series of mesopelagic biomass) has started. • Ocean climate variability and historical change are reasonably well understood. • Broad reviews have been completed of the impacts of climate variability on fisheries (especially recruitment), but the likely impacts of ocean climate change or acidification remain poorly known. • This theme has links and synergies with MBIE, DOC, universities and the MPI biodiversity programme.
• Taxonomy and ID Guides have been produced and specimens recorded in National Collections. • Biodiversity surveys completed on local scale (Fiordland, Spirit's Bay, seamounts) and larger fishery scale (Norfolk Ridge, Chatham Rise, Challenger Plateau, Bay of Islands). • Measures and indicators for marine biodiversity measures and ecosystem have been developed. • Predictive modelling techniques have been applied and habitat classification methods improved • Productivity in benthic communities has been measured. • Specimens from New Zealand have been genetically assessed and entered into the barcode of life. • Seamount connectivity, land-sea connectivity, and endemism have been studied. • Two measures of marine conditions and marine biodiversity have been evaluated as tier 1 statistics. • Demersal fish trophic studies on the Chatham Rise have been completed. • A review of NZ data from deep-sea and abyssal habitats has been completed. • A multidisciplinary study of long-term (1000 years) changes to NZ marine ecosystem is complete. • Latitudinal gradient project, ICECUBE and 2 large scale surveys in the Ross Sea have been conducted. • The MPI Marine Biodiversity Research Programme theme has links and synergies with MBIE, DOC, universities and the MPI AEWG programmes

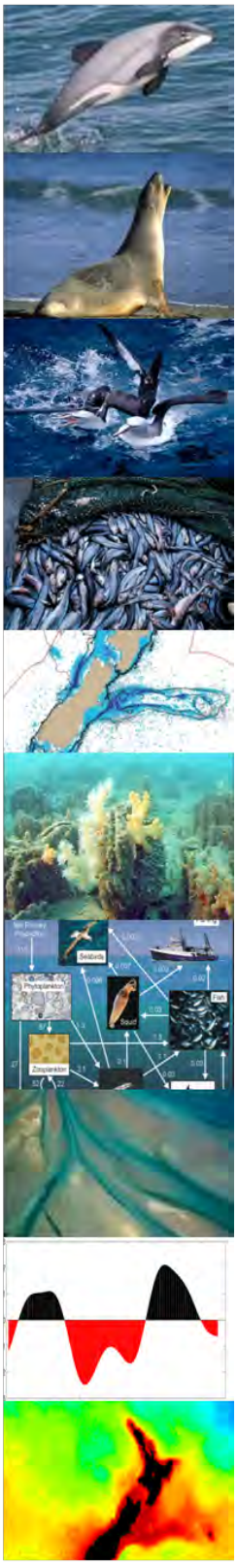


Figure 2.1 [Continued]: Summary of themes in the Aquatic Environment and Biodiversity Annual Review 2017.

3 SPATIALLY EXPLICIT FISHERIES RISK ASSESSMENT (SEFRA):

Status of chapter	This chapter is reprinted with only slight amendments from its original publication in the AEBAR 2016. It was subsequently reviewed by an independent international panel (Lonergan et al. 2017), and will be modified for publication in the primary literature. Reference for citation in its current form is as follows: Sharp, BR (2018). Spatially Explicit Fisheries Risk Assessment: A framework for quantifying and managing incidental commercial fisheries impacts on non-target species'. Chapter 3 in: Aquatic Environment and Biodiversity Annual Review 2018. Ministry for Primary Industries, New Zealand. Lead author for correspondence: ben.sharp@mpi.govt.nz
Scope of chapter	This chapter describes New Zealand's Spatially Explicit Fisheries Risk Assessment method, which has been designed to estimate fisheries impact and risk for non-target species, and to inform risk management responses within a quantitative and statistically rigorous framework. The chapter includes: i) a description of the conceptual and specific mathematical application of this method to New Zealand seabird and marine mammal species; ii) a description of required data inputs and potential pitfalls in the application of this specific method; and iii) a more general discussion of other planned or in progress applications of the SEFRA framework, e.g., applied to non-target fish or benthic invertebrates, for which the conceptual approach is the same but modified methods will be developed in the implementation stage.
Area	The SEFRA method can be applied at any spatial scale at which spatial data representing species distributions and fishing effort distributions are available. The most fully developed implementations, for New Zealand seabirds and marine mammals, have been applied at the scale of the New Zealand EEZ.
Focal localities	Outputs from each implementation of the SEFRA method will identify different key locations at which fisheries risk occurs, based on the spatial overlap between species distributions and the fishing effort to which that species is most vulnerable.
Key issues	To assess and manage fisheries risks across large numbers of potentially affected non-target populations, fisheries managers are forced to make difficult decisions in the context of poor and/or sparse information. Innovative methods are required to enable maximum use of available data in a transparent and statistically rigorous framework. Application of the SEFRA method to the New Zealand Seabird Risk Assessment (NZSRA) has been iteratively improved since the initial design of the method in 2009. The updated NZSRA will constitute a 'full' implementation of the method as designed, providing a useful methodological template for other risk assessments.
Emerging issues	The first application of the SEFRA method to the New Zealand Marine Mammal Risk Assessment (NZMMRA) is now complete (Abraham et al. 2017), closely following the method template provided by the NZSRA. Modified applications of the method are in development (for individual protected species, global seabirds, non-target fish, and pelagic protected fish) or planned as future work.
MPI research (current)	The current New Zealand Seabird Risk Assessment (NZSRA) is delivered under contract PRO-2014-06. The current New Zealand Marine Mammal Risk Assessment (NZMMRA) is delivered under contract PRO-2012-02. A customised user-driven query and simulation tool to inform risk management is in development under contract PRO-2016-06. Cetacean spatial distribution modelling to inform an improved MMRA is delivered under contract PRO-2014-01. A global seabird risk assessment is in progress under contract PRO-2013-13. SEFRA implementations for particular mammal or bird species are in progress under SEA2016-30, PRO2017-12, and PRO2017-10, and PRO2017-19.
NZ government research (current)	Risk assessment outputs are routinely used to inform the prioritisation of biological and population monitoring research under the DOC Conservation Services Programme (CSP) and MPI protected species programme research, to focus research efforts on populations or variables for which uncertain parameter inputs have significant effects on risk estimation for species of interest.
Related chapters/issues	Results of the NZMMRA are summarised in species-specific marine mammal chapters for NZ sea lions, New Zealand fur seals, Hector's and Māui dolphins, and common dolphins (i.e., Chapters 4–7) Results of the NZSRA are included in New Zealand seabirds, Chapter 8. Future implementations of the SEFRA framework may inform updates of these chapters and/or Chapters 9–11.

3.1 CONTEXT

3.1.1 SCOPE

The scope of the Spatially Explicit Fisheries Risk Assessment framework (hereafter SEFRA) is *to assess the population-level risk to non-target species arising from direct incidental mortality in commercial fisheries.* The SEFRA framework combines an impact assessment to estimate the level of incidental fisheries mortality with a biological assessment of the associated effect on the population, as a function of population size and demographic parameters influencing population productivity. The SEFRA framework does not address potential indirect fisheries effects, e.g., trophic effects.

This paper outlines the conceptual and mathematical basis for the application of the SEFRA framework to estimate fisheries risk to seabirds and marine mammals, for which the method is nearly identical. Other applications of the framework, e.g., applied to non-target fish or benthic invertebrates, are in progress but will require modifications to the mathematical framework described below. These will be described separately.

3.1.2 BACKGROUND

The SEFRA framework was developed initially with specific reference to commercial fisheries impacts on New Zealand seabirds. The scope and nature of the SEFRA framework was designed to address the specific information needs of fisheries managers charged with managing seabird impacts by New Zealand fisheries, and with reference to the level and quality of available data in New Zealand to inform the risk assessment process. Risk assessments that carefully consider management needs and data limitations in the design stage are likely to be more effective than generic templates applied universally for different kinds of threats and for a wide range of management applications (such as the templates described by Hobday et al. 2007).

The specific New Zealand seabird context is as follows:

- At a global scale New Zealand has a disproportionately high number of resident or breeding seabird populations. For many of these species, reliable demographic or population data are unavailable, and are not feasible to obtain, for example due to remote colony locations.
- New Zealand seabirds are exposed to risk from a wide variety of fishing methods. The quality and availability of fisheries observer data useful to estimate incidental capture rates varies greatly, from relatively well-observed deepwater fisheries (30–50% of fishing events observed) to very poorly observed primarily inshore fisheries (often less than 1% of fishing events observed).
- Fisheries observer coverage is variable, and what data is available is almost always spatially unrepresentative of the whole, due to spatially non-random distribution of observers and highly variable vessel interaction rates with seabirds in different locations. Direct estimation of seabird impacts from observed capture rates without reference to spatial overlap patterns therefore has the potential to be dangerously biased.
- Some seabirds have very low population sizes, or are impossible for non-expert fisheries observers to identify reliably at sea, so that observed capture rates on a species-specific basis are not a reliable means of estimating population-level risk.

Data availability and the needs of fisheries managers drove the following decisions in the design and application of the SEFRA framework to New Zealand seabirds:

- The fundamental unit at which risk is assessed is *per seabird species or distinct population*. Biological risk assessment only makes sense with reference to units that are biologically meaningful. Only subsequently does it make sense to disaggregate and assign the risk to particular fisheries or areas. Assessment frameworks that assign risk on the basis of administrative categories but do not relate these to total risk at the species or population level (e.g., Campbell & Gallagher 2007) are inadequate for this purpose.
- The SEFRA method *can be applied to every species of seabird for which spatial distributions have been estimated.*
- The risk assessment stage *does not rely on species-specific population models or monitoring studies;* these are unavailable for most species.
- The impact assessment *does not rely on the existence of universal or representative fisheries observer data* to estimate seabird mortality. Fisheries observer coverage is generally too low and/or too spatially

unrepresentative to allow direct estimation of seabird bycatch at a species level. *The SEFRA method can be applied for any fishery for which some observer data exists, and modifications of this method (see Section 3.2 below) are useful even where no observer data are available to estimate capture rates.*

- The SEFRA framework assigns risk to each species in an *absolute* sense, i.e., species are not merely ranked relative to one another (e.g., as in the PSA approach; Hobday et al. 2007, Waugh et al. 2008). An absolute as opposed to a relative risk score is required to set clear performance standards to meet conservation goals, and to track changes in performance over time arising from mitigation or management.

- Risk is estimated as a function of population-level impact and of biological parameters that are generally available from published sources, reducing reliance on new or location-specific population data which are often unavailable. Risk can be estimated even for species for which no estimate of population size is available.

- Both impact and risk are *quantitative* and *objectively scalable* between fisheries or areas, so that risk at a species level can be disaggregated and assigned to different fisheries or areas based on their proportional contribution to total impact. This allows managers to identify risk hotspots to target management interventions effectively, to track location- or fishery-specific change over time, and to equitably assign responsibility for necessary risk management responses. It also provides tangible incentive for the adoption of mitigation to reduce impact on a location- or fishery-specific basis.

- *The estimation of risk for each species is quantitative and repeatable* without reference to subjective interpretation or expert knowledge, enabling managers to utilise a consistent decision framework for necessary management action to meet performance standards, and to track changing risk over time.

- *The SEFRA framework allows explicit (Bayesian) treatment of uncertainty, and does not conflate uncertainty with risk* (see Kaplan 1997). Because risk is calculated from numerical inputs for which confidence intervals are explicit, it is possible to track the propagation of uncertainty from uncertain parameter inputs and/or noisy data through to output estimates of risk. The outputs distinguish between situations where information is sufficient to ascertain that impacts are unacceptably high (i.e., high impact, low uncertainty, requiring management intervention) and those where information is insufficient to estimate impacts reliably (i.e., unknown impact, high uncertainty, suggesting the need for additional data collection). It is also possible to identify the origins of the uncertainty (i.e., which input parameters are most responsible for uncertainty of the output estimates) to target new research most effectively.

- The SEFRA framework is designed to *readily incorporate new information.* Assumptions in the impact assessment stage are transparent and testable; as new data become available or assumptions change, the consequences for the subsequent impact and risk calculations arise logically *without the need to revisit other assumptions or repeat the entire risk assessment process,* which would otherwise constitute a major and cost-prohibitive institutional burden to managers.

3.1.3 ITERATIVE DEVELOPMENT OF THE NEW ZEALAND SEABIRD RISK ASSESSMENT

The SEFRA method was initially developed arising from a New Zealand Ministry of Fisheries workshop hosted 18–19 February 2009 (described in Sharp et al. 2011) to support the revision of New Zealand's National Plan of Action – Seabirds. Subsequent to the workshop, application of the SEFRA method has been updated and substantially improved in multiple iterations of the New Zealand seabird risk assessment (hereafter NZSRA), arising from productive collaboration between MPI scientists and contracted research providers, with input from the MPI Aquatic Environment Working Group and the Seabird Stakeholder Advisory Group. Sequential iterations of the seabird risk assessment from 2009–15 are described in Waugh et al. (2009), Richard et al. (2011), Richard & Abraham (2013b), Richard & Abraham (2015), and Richard et al. (2017).

Cognisant of structural or methodological improvements that had not yet been actioned in SEFRA implementations to date, the full method framework was described here for the first time (in 2017) to guide future work. Subsequently, the first SERFA implementation fully consistent with the method described herein was a single-species assessment for Hector's- Maui dolphins (Roberts et al. 2019). A multi-species seabird implementation consistent the framework is in development (D Webber, in prep).

In the National Plan of Action – seabirds (NPOA-Seabirds; Ministry for Primary Industries 2013), the SEFRA method was adopted as the means by which species-level risk to seabirds is assessed, and to provide a performance metric by which risk-reduction goals are defined and evaluated.

3.1.4 APPLICATIONS OF THE SEFRA FRAMEWORK TO OTHER RISK ASSESSMENTS

It is planned that variations on the SEFRA method will be used in New Zealand to deliver risk assessments across a wide range of direct fisheries impacts. In addition to the New Zealand seabird risk assessment, the method has been or is being applied also as follows:

- Waugh et al. (2012) applied a variation of the SEFRA method to characterise risk to multiple seabird species on a global scale associated with tuna fishing effort under the Commission for the Conservation of Southern Bluefin Tuna (CCSBT).

- Currey et al. (2013) used a simplified precursor to the SEFRA method to estimate commercial trawl and set-net fishery risk to Māui dolphins, as part of an expert workshop to characterise risk to this species from both fisheries and non-fisheries threats. Outputs of this workshop were subsequently used to evaluate the relative efficacy of alternate risk-reduction strategies and inform management.

- The first iteration of a New Zealand Marine Mammal Risk Assessment (hereafter NZMMRA) was completed in 2017 (Abraham et al. 2017).

- A species-specific implementation of the SEFRA method focused on Māui and Hector's dolphins is in progress (MPI project SEA2016-30) to estimate fisheries risk and inform the evaluation of hypothetical risk management scenarios.

- The SEFRA framework will be adapted to also address non-fishery threats in a multi-threat risk assessment (PRO2017-12) to inform the update of the Māui and Hector's dolphin Threat Management Plan in 2018.

- Species-specific implementations of the SEFRA are planned for New Zealand sea lions and fur seals once available satellite telemetry has been analysed to estimate spatial foraging distributions (PRO2017-10).

- A Southern Hemisphere seabird risk assessment is currently in progress to assess risk to globally distributed New Zealand seabird species from all

commercial High Seas and EEZ fishing effort.

- Adaptations of the SEFRA method are being considered to evaluate harvest rates for non-target and/or low information fish species.

Adaptations of this method are also being considered to evaluate fisheries risk to other protected species and harvest rates for non-target fish in other areas. The SEFRA method is also fully compatible with a spatially explicit bottom fishing impact assessment method described in Sharp et al. (2009) and further developed (with simulations including recovery from impacts and management strategy evaluation) in Mormede & Dunn (2012). The existence of comprehensive spatially explicit risk assessments evaluating all fisheries impacts simultaneously, and with the ability to evaluate alternate management scenarios via management strategy evaluation (MSE), will provide a powerful tool to inform fisheries management.

3.1.5 CHAPTER OVERVIEW

This chapter describes the SEFRA framework at the conceptual and methodological level, without reference to one particular implementation of the method. Section 3.2 outlines the mathematical formulation a multi-species implementation of the method, which applies a fully integrated Bayesian model to estimate capture rates and risk across multiple species and different fisheries simultaneously, as in the current NZSRA and NZMMRA. Section 3.3 describes in detail the structural assumptions and necessary input parameters to inform the model formulation outlined in Section 3.2. Section 0 briefly describes potential alternative applications of the method to address different types of problems, or to accommodate situations where the data are not available to inform all of the standard inputs in the fully integrated Bayesian modelling method.

Where appropriate, the method description is illustrated with examples from one or more of the existing SEFRA implementations listed above, or where necessary from unpublished implementations still in development. Because the SEFRA method was first designed in the context of the NZSRA, many of these examples are extracted or reproduced from Richard & Abraham (2015) or from the unpublished subsequent iteration of the NZSRA described in Chapter 8, but where alternative methodological choices are best illustrated by other existing risk assessments, these are cited in turn. Results of the most recent NZSRA are included separately in the seabird chapter of this AEBAR, Chapter 8. Results of the NZMMRA are published separately in Abraham et al. (2017).

3.2 METHODS

3.2.1 INTEGRATED BAYESIAN MULTISPECIES IMPACT ESTIMATION: MATHEMATICAL OVERVIEW

Mathematical parameters and their support are summarised in Table 3.1.

3.2.1.1 OVERLAP

The SEFRA method estimates the encounter rate between non-target species and fishing effort as a function of the *overlap* (in space and time) between mapped species distributions and mapped fishing effort distributions. Every fishing event *i* is assumed to be within the 2-dimensional space $\mathbb{X}$ (i.e., $i \in \mathbb{X}$) and to occur at some time (i.e., $i \in \mathbb{T}$).

For each species s, at the location and time of every fishing event *i*, O_{si} is the *overlap* parameter, estimated as the product of the fishing intensity a_i and species probability density p_{si} at the location of fishing event *i*, i.e.:

$$O_{si} = a_i * p_{si} \quad (1)$$

where a_i is a metric of *fishing effort intensity* (e.g., number of hooks, kilometres of net) assigned to every fishing event *i*; and p_{si} is the *species probability density* at that location and time, i.e., the probability that an individual of species s selected at random from the population occupies that spatial cell at the time of the fishing event; the sum of all cells in the spatial domain must equal one.

Table 3.1: Mathematical variables and their support as utilised in equations (1) – (30).

Variable	Support	Description
Indices		
i		Fishing event index
s		Species index
z		Species group index
g		Fishery group index (all fishing events *i* are assigned to a fishery group denoted *g*)
Covariates		
a_i	$a_i > 0$	Fishing intensity per event (e.g., number of tows, number of hooks, length of nets)
p_{si}	$p_{si} \geq 0$	Species (individual) probability density
k_{zg}	$k_{zg} \geq 1$	Cryptic mortality multiplier
r_{zg}	$0 \leq r_{zg} \leq 1$	Live release rate
L_{zg}	$0 \leq L_{zg} \leq 1$	Live release survival rate
Derived quantities		
O_{si}	$O_{si} \geq 0$	Species (individual) overlap
∂_{zi}	$\partial_{zi} \geq 0$	Species group density in space
Θ_{zg}	$\Theta_{zg} \geq 0$	Species group density overlap
q_{sg}	$q_{sg} \geq 0$	Catchability
$\varkappa_{zg}$	$\varkappa_{zg} \geq 0$	Total fisheries related deaths multiplier
I_{zg}	$I_{zg} \geq 0$	Fishery interactions
D_{sgi}	$D_{sgi} \geq 0$	Fisheries related deaths
U_s	$\mathrm{U}_s \geq 0$	Species impact ratio
R_s	$\mathrm{R}_s \geq 0$	Species risk ratio
PST_s	$\mathrm{PST}_s \geq 0$	Population Sustainability Threshold
Data		
C_{zgi}	$C_{zgi} \geq 0$	Observable captures
C'_{zgi}	$C'_{zgi} \geq 0$	Observed captures
Parameters		
v_g	$v_g \geq 0$	Fishery group vulnerability
v_z	$v_z \geq 0$	Species group vulnerability
$\mathbb{N}_{si}$	$\mathbb{N}_{si} \geq 0$	Available population size
N_{si}	$\mathrm{N}_{si} \geq 0$	Biological population size
φ	$\varphi \geq 0$	PST adjustment factor
r_{max}	$r_{max_s} \geq 0$	Maximum population growth rate

3.2.1.2 FISHERY GROUPS

All fishing events *i* are assigned to *fishery groups g* within which the gear configuration and vessel behaviour is assumed to be similar, such that species catchability and vulnerability estimates for each species group can be applied uniformly to all effort in the fishery group. The overlap of a species with all fishing effort in the fishery group is obtained by summing across all fishing events in the group.

$$O_{sg} = \sum_{\mathrm{i}} O_{sgi} \tag{2}$$

3.2.1.3 TOTAL OBSERVABLE CAPTURES

A *capture* is an event whereby an individual of the non-target species in question is entangled or restrained by fishing gear (alive or dead) and is unable to free itself under its own power. *Captures* include animals that are killed and their bodies recovered on board the vessel, plus animals released alive, but exclude *cryptic deaths* (see below). *Observable captures* include all captures that occur and would be recorded if 100% of fishing events were observed. *Observed captures* refer to only that subset of observable captures that are actually recorded by fisheries observers.

Total observable captures *C* of each species per fishing event in fishery group *g* is a product of the probability of encounter per individual (proportional to overlap *O*), times the probability of capture per encounter (*q*), times the available population size at time *t* of fishing event *i*:

$$C_{sgi} = q_{sg} O_{sgi} * \mathbb{N}_{si} = q_{sg} \theta_{sgi} \tag{3}$$

where $C_{sgi} \geq 0$ is implied.

> $q_{sg} \geq 0$ is the catchability for species s in fishery group g; (analogous to catchability in a fisheries context, hence abbreviated *q*); and
>
> $\mathbb{N}_{si} \geq 0$ is the *available population size* of species *s* at time *t*, i.e., the biological population size *N* adjusted to reflect the proportion of that population that is within the spatial domain of the assessment at the time of fishing event *i*.
>
> $\theta_{sgi} \geq 0$ is the *density overlap* of species *s* with fishing event *i* (see below).

Total observable captures in fishery group *g* is obtained by summing across captures at all events:

$$C_{sg} = \sum_{\mathrm{i}} C_{sgi} \tag{4}$$

3.2.1.4 DENSITY OVERLAP

The overlap term *O* represents the probability or frequency that a particular individual animal selected at random from the population will encounter a fishing event of a particular fishery group. In contrast, the *density overlap* θ represents the number or frequency of encounters for all individuals of that species. Overlap is converted to a *density overlap per event* by multiplying by species *available population size*:

$$\theta_{sgi} = O_{sgi} * \mathbb{N}_{si} \tag{5}$$

where:

> $\mathbb{N}_{si}$ is the *available population size*, i.e., the number of animals of species s that are present within the spatial domain of the risk assessment at the time t corresponding to fishing event i;

Note that where available population size is seasonally variable (i.e., $\mathbb{N}_{si}$ is not the same for all events i throughout the year), density overlap $\theta_s$must first be calculated at the level of fishing events as in equation (5) and only subsequently summed across events in a fishery group. One consequence is that relative values of O between species reflect relative exposure to fishing effort per individual animal, which scales directly with risk, whereas θ values reflect absolute encounter rates per species, which scales with expected captures but not risk because θ is confounded with population size. For this reason *O* rather than θ is used until such time as actual densities are estimated across all species in a species group (equations (8) – (9)).

3.2.1.5 IMPROVED CATCHABILITY ESTIMATION USING SPECIES GROUPS

In its most rigorous application, the SEFRA method allows fully quantitative estimation of species-level catchability, applying Bayesian inference to estimate capture rates per encounter for each combination of species x fishery group (q_{sg}), as a function of observable captures $C_{sg}$and overlap O_{sg}, as in equation (3). However risk assessment methods are designed for application to data-poor problems; if sufficient data existed to estimate catchability for every species x fishery group combination individually, it is unlikely that a risk assessment approach would be required at all; instead captures could simply be estimated directly. In New Zealand, direct estimation is used to estimate captures of the most commonly caught seabirds by the most well observed fisheries (Abraham & Richard 2017; see http://data.dragonfly.co.nz/psc), but this approach is not feasible for the majority of species and fishery groups. In early iterations of the NZSRA (e.g., Richard et al. 2011) application of the approach in equation (3) to species x fishery group combinations for which there were few or no observed captures yielded unacceptably unconstrained answers: estimates of q_{sg} and C_{sg} sometimes varied by more than two orders of magnitude, and extended into biologically implausible bounds.

To better estimate q the dimensionality of the model can be reduced by aggregating individual species s into species groups z on the basis of common physical and behavioural characteristics thought to affect capture rates, such that all species in the group are assumed to have the same catchability q_{zg}.

$$C_{zi} = \sum_{s} q_{zg} O_{si} * \mathbb{N}_{si} \quad (6)$$

3.2.1.6 COMBINED DENSITY OVERLAP

To combine species within a species group, probability density values for each species in the location of every fishing event *i* (p_{si}) at time *t* are converted to actual animal densities and summed across all species in the species group *z* per fishing event, as follows:

$$\partial_{zi} = \sum_{s} (p_{si} * \mathbb{N}_{si}) \quad (7)$$

where:

∂_{zi} is the actual density of all individuals of species group *z* at the time and location of fishing event *i*;

$\mathbb{N}_{si}$ is the *available population size* (see below) of species *s* at the time t of the fishing event *i*.

The use of *available population size* $\mathbb{N}_{st}$ in equations (5) and (7) recognises that the number of individuals actually present in the spatial domain of the risk assessment at the moment of fishing event *i* may be different than the size of the biological population *N* against which impacts are evaluated.

Subsequently, the *density overlap* between species group z and fishery group g (Θ_{zg}) can be estimated simultaneously across all fishing events *i*, by combining equations (1) and (7), as follows:

$$\Theta_{zg} = \sum_{i} (a_{gi} * \partial_{zi}) \quad (8)$$

Note that *density overlap* Θ is different from the previously used overlap *O* in that it refers to the combined actual density of all individuals rather than a probability distribution per individual; this is necessary in order to accurately reflect variable abundances across species when summing distributions across multiple species in a species group. Total observable captures per species group across all fishing events is then:

$$C_{zg} = q_{zg} \Theta_{zg} \quad (9)$$

3.2.1.7 CRYPTIC MORTALITY AND TOTAL FISHERIES RELATED DEATHS

Especially for protected species such as seabirds and marine mammals, not all observable captures result in death, and conversely not all deaths arising from fishery interactions result in an observable capture. Estimation of fishery related deaths D_{sg} from captures data is as follows:

$$D_{sg} = \left(C_{sg} * k_{sg}\right) - \left(C_{sg} * r_{sg} * L_{sg}\right)$$
$$= C_{sg}\left(k_{sg} - \left(r_{sg} * L_{sg}\right)\right)$$

(10)

where:

k_{sg} is the *cryptic mortality multiplier,* i.e., a multiplier of the observed captures to account for the additional individuals that die as a direct result of their interaction with the fishing effort but are not recovered on board the vessel and recorded as captures; and

r_{sg} is the *live release rate*, i.e., the proportion of captured individuals that are released alive; and

L_{sg} is the *live release survival rate*, i.e., proportion of live releases expected to survive.

To aid subsequent algebraic manipulation, it is useful to combine these parameters (with uncertainty) into a *total fisheries related deaths multiplier denoted by* $\varkappa$ (kappa), to facilitate conversion between total observable captures *C* and total fishery related deaths *D*, as follows:

$$D_{sg} = C_{sg} * \varkappa_{sg}$$

(11)

where $\varkappa_{sg} = \left(k_{sg} - \left(r_{sg} * L_{sg}\right)\right)$

3.2.1.8 SPECIES VULNERABILITY TO INTERACTION

Non-target species capture rates are modelled separately within each of several broadly defined *fishing methods*. The NZSRA defines four such fishing methods: trawls, bottom longlines, surface longlines and set nets. The NZMMRA includes also purse seines as a fifth method. *Fishery groups* are nested subsets of *fishing methods*.

Within each such method-specific model, interaction rates between species groups and fishery groups are estimated at the level of *interaction incidents* rather than deaths or captures in isolation. Interactions I_{zg} are defined as captures (alive or dead) plus cryptic deaths, i.e.,

$$I_{zg} = C_{zg} * k_{zg} = q_{zg}\Theta_{zg}k_{zg}$$

(12)

Species *vulnerability v* is defined as the probability of interaction per encounter with fishing effort (i.e., vulnerability *v* includes captures plus cryptic deaths, as opposed to catchability *q*, which is the probability of capture excluding cryptic mortality).

$$v_{zg} = q_{zg}k_{zg}$$

(13)

$$I_{zg} = v_{zg}\Theta_{zg}$$

(14)

A major innovation first utilised in the third iteration of the NZSRA (Richard & Abraham 2013b) was to split the vulnerability parameter v_{zg} into two parameters representing species group vulnerability v_z and fishery-group vulnerability v_g separately, as follows:

$$I_{zg} = v_z v_g \Theta_{zg}$$

(15)

The species group vulnerability term v_z reflects that some species groups are more attracted to fishing vessels, or otherwise more susceptible to capture or cryptic death than other species groups. The structural assumption imposed by splitting the vulnerability parameter in this way is that the relative difference in species group vulnerability will apply across all fishery groups within a broadly defined fishing method (e.g., a bird species that aggressively interacts with trawl fisheries will be more vulnerable to capture in all trawls than is a less aggressive bird species, reflected by a higher v_z, and this relationship will be constant across trawl fishery groups).

Similarly, the fishery group vulnerability term v_g reflects that within each fishing method, some fishery groups will be expected to capture or kill non-target species more often than do other fishery groups, e.g., reflecting

mitigation uptake or offal discard practices, and this fishery group effect will apply across all species groups in common.

By separating the vulnerability term v_{zg} into these separate components, this model structure effectively allows capture rates in data-limited species x fishery group combinations to be informed or constrained by data from species x fishery group combinations for which more data are available (i.e., because of higher populations, or higher capture rates, or higher levels of observer coverage). In the example of the NZSRA, replacing the single-parameter approach in equation (3) with the split-parameter approach in equation (15) yielded substantially improved model power.

Estimation is applied to *interactions* rather than *captures* (i.e., *vulnerability* not *catchability*) on the assumption that the inherent species group and fishery group properties represented by the v_z and v_g terms affect the rate at which the species will physically interact with fishing gear, but that subsequent retention of corpses affecting the cryptic mortality multiplier k_{zg} (hence capture rate C_{zg}) may operate independently per combination of fishery x species group. This formulation has significant implications for the way that cryptic mortality multipliers are applied, especially in poorly estimated fishery group x species group combinations. Most or all of the factors affecting cryptic mortality multipliers are by necessity estimated outside the integrated model, using input priors to represent uncertainty (see below).

Re-expressing capture rates (for which fisheries observer data are useful) in terms of vulnerability rather than catchability yields:

$$C_{zg} = \frac{v_z v_g \Theta_{zg}}{k_{zg}} \tag{16}$$

3.2.1.9 BAYESIAN ESTIMATION OF CATCHABILITY

To estimate total observable captures and catchabilities from available fisheries observer data, the most rigorous application of the SEFRA method applies a Bayesian model for each of the broadly defined fishing methods (e.g., trawl, surface longline, bottom longline and set net), using data from observed fishing events to estimate capture rates and species vulnerability simultaneously across all species and fishery groups within the fishing method.

Total observable captures C_{zg} are estimated across all fishing events per fishery group on an annual basis. Because protected species capture rates refer to relatively infrequent events resulting in individual animal deaths, in the NZSRA and NZMMRA total observable captures are modelled using a Poisson distribution as follows:

$$C_{zg} \sim Poisson(\hat{\lambda}_{zg}) \tag{17}$$

Other error distributions may be appropriate for other implementations of the SEFRA method, e.g., non-target fish bycatch or benthic invertebrate impacts.

Modifying equation (16),

$$\hat{\lambda}'_{zg} = \sum_{i} \frac{v_z v_g \Theta'_{zgi}}{k_{zg}} * \epsilon_{zg} \tag{18}$$

where the ' suffix is used to denote parameters referring only to the observed subset of total fishing effort, as follows:

$\hat{\lambda}'_{zgi}$ is the estimated *observed* captures of all species in species group z associated with fishing group g.

Θ'_{zi} is the observed density overlap of species group z with observed fishing event i. This term is functionally equivalent to the spatial overlap O_{sg} in equation (2), except transformed to represent actual densities across all species in the group rather than probability densities per species, and restricted to observed events rather than all events.

v_z is the species group vulnerability for species group z;

v_g is the fishery group vulnerability for fishery group g;

k_{zg} is the cryptic mortality rate for species group *z* in fishery group *g*; and

ϵ_{zg} is an error term associated with the combination of species group *z* and fishery group *g*;

3.2.2 RE-APPLYING MODELLED VULNERABILITES TO SPECIES-LEVEL IMPACT

An integrated Bayesian model fitted to fisheries observer data as in Equation (18) is the best means by which observed capture rates across all fisheries and species can be used to estimate v_z and v_g in a multi-species/multi-fishery risk setting. Subsequently the split vulnerability parameters v_z and v_g are re-combined with estimates of the cryptic mortality multiplier k_{zg} to estimate q_{sg} as in equation (12) (noting v_s = v_z for all species in group z), and combined with live releases and live release survival as in equations (10)–(11) to estimate total fishery-related deaths (hereafter FRDs).

$$I_{sgi} = v_s v_g O_{si} * \mathbb{N}_{si} \tag{19}$$

$$C_{sgi} = \frac{v_s v_g O_{si} * \mathbb{N}_{si}}{k_{zg}} \tag{20}$$

$$D_{sgi} = \frac{v_s v_g O_{si} * \mathbb{N}_{si}}{k_{zg}} * \varkappa_{sg} \tag{21}$$

In these equations impacts can be estimated per individual fishing event (including un-observed fishing events) or combined at any scale to yield spatially explicit estimates of captures and FRDs (with uncertainty) on an individual species and fishery basis, even for species and fisheries for which captures data were insufficient to inform estimates of species catchability on an individual basis. Model diagnostics should include comparisons of observed vs. expected numbers of observed captures, including on a spatially disaggregated basis (e.g. Figure 3.9 below) to inform evaluation of structural model assumptions and to assess the accuracy of spatial data layer inputs.

3.2.3 FROM IMPACT TO RISK

For protected species risk assessments, the estimation of species level impact and risk is as follows. Alternative approaches utilised in fish and benthic habitat risk assessments will be developed separately.

3.2.3.1 BIOLOGICAL POPULATION SIZE

Fishery-related deaths on an annual basis are evaluated as a proportion of the biological population size for each species, N_s. To ensure that risk scores are biologically meaningful, N_s is necessarily applied at the level of a distinct biological population at the scale of a country or region (for protected species) or a distinct stock (for non-target fish). Where and when a proportion of the biological population exists outside of the spatial domain of the risk assessment, biological population N_{si} will differ from available population $\mathbb{N}_{si}$.

3.2.3.2 IMPACT RATIO

Because individual deaths are additive, impacts can be summed across groups, yielding total FRDs at the species level:

$$D_s = \sum_{\mathrm{g}} D_{sg} \tag{22}$$

The *impact ratio U* is defined as the proportion of the total biological population killed by fishing effort each year, either at a fishery group level or collectively for all fishery groups at the species level:

$$U_{sg} = \frac{D_{sg}}{\mathrm{N}_s} \tag{23}$$

$$U_s = \sum_{\mathrm{g}} U_{sg} = \sum_{\mathrm{g}} \frac{D_{sg}}{\mathrm{N}_s} \tag{24}$$

… U_s is therefore analogous to exploitation rate *U* in fisheries.

Note that combining equations (3), (7) and (9) (where N = $\mathbb{N}$, i.e., neglecting or correcting for seasonal migrations that change available population size) implies

$$\frac{D_{sg}}{N_s} = q_{sg} O_{sg} \varkappa_{sg} = U_{sg} \tag{25}$$

Summing across all fishery groups as in equation (25) yields:

$$U_s = \sum_{g} q_{sg} O_{sg} \varkappa_{sg} \tag{26}$$

The power of this formulation is that so long as species catchability q_{sg} can be estimated by some means other than equation (3) (and adjusting for variable seasonal presence of the species in question within the spatial domain) it becomes possible to estimate impact levels (and subsequently risk), *even for species for which both population size N_s and total observable captures C_{sg} are unknown*.

Equation (26) becomes very important in the application of the SEFRA method to very rare species (because captures are too rarely observed to estimate C_{sg} with any statistical power), or to species for which no observer data is available to estimate capture rates, or to species for which population size is unknown (e.g., seabirds for which colonies are inaccessible to survey; deepwater fish; many cetaceans). Alternative means of estimating q are under development for application of the SEFRA method to deepwater fish (Sibanda et al. 2016), analogous to similar approaches applied overseas (Zhou et al. 2009, 2011). In data-poor situations relative catchability *q* between species can also be intuited from first principles and expert knowledge (with uncertainty) or estimated by analogy with more data-rich applications conducted for similar species elsewhere.

3.2.3.3 RISK RATIO

Under the SEFRA framework, 'risk' is defined as the estimated species-level fisheries impact as a proportion of a defined impact sustainability threshold, i.e.,

$$R_s = \widehat{U}_s / U_s \tag{27}$$

Because intuitively the ability of a species to sustain impacts is related to its biological productivity, the chosen threshold U_s will vary accordingly, i.e., analogous to a target exploitation rate U_{msy} for fish or to PBR approaches commonly applied to marine mammals (Wade 1998). Where impacts are generally expressed as an annual exploitation rate (i.e., fish) or a proportional spatial impact per unit time (benthic habitats) we have adopted the term 'Maximum Impact Sustainability Threshold' or MIST, first proposed in the planned implementation of the SEFRA method for fish (Roux et al. 2015).

Implicit in the choice of threshold U_s (MIST, or PST see below) is a particular population outcome corresponding to a particular level of impact; this relationship between impact and population outcome is established via simulations. Note that because under the SEFRA method output estimates of impact and risk are themselves uncertain, it is necessary that the chosen population outcome used to define the impact threshold U_s (corresponding to R_s = 1) is expressed with reference to the level of certainty with which the outcome will be achieved.

For protected species where impacts are more commonly expressed as individual deaths rather than annual exploitation rate, an alternative but mathematically equivalent formulation of equation (27) is:

$$R_s = \frac{D_s}{PST_s} \tag{28}$$

where:

D_s is *total fishery related deaths* from equation (10), and

PST_s is the *Population Sustainability Threshold* expressed as a number of individual deaths per year and defined with reference to a particular population outcome (see below).

3.2.3.4 POPULATION SUSTAINABILITY THRESHOLD (PST)

For protected species, the SEFRA method defines an impact threshold as a function of maximum population growth rate r_{max} , analogous to the PBR ('potential biological removals') formulation of Wade (1998). Wade (1998) defines PBR as:

$$PBR = \frac{1}{2} r_{max} * N_{min} * f \quad (29)$$

where:

r_{max} is the theoretical unconstrained maximum population growth rate, reflecting biological productivity;

N_{min} is a conservative point estimate (20th percentile) of total population size; and

f is a subjective 'recovery factor' defined to adjust the threshold value to reflect management goals on a per-species basis.

Early implementations of the NZSRA utilised variations on the PBR formulation in the definition of risk, but subsequently refined this approach to the extent that referring to 'PBR' in the NZSRA is now misleading. From the 2017 iteration of the NZSRA and the first MMRA, we coin the term 'Population Sustainability Threshold' or PST, defined as follows:

$$PST = \frac{1}{2} \varphi * r_{max} * N \quad (30)$$

where:

φ (greek letter phi) is an adjustment factor estimated by simulation and defined to ensure that impacts equal to PST (i.e., R = 1) correspond to a defined population stabilisation or recovery objective.

The r_{max} term is estimated from biological and demographic input parameters, the estimation of which will be specific to different taxa, e.g., marine mammals vs. seabirds (see Section 3.3, Model Inputs, below).

For seabirds, earlier iterations of the NZSRA estimated r_{max} from field estimates of adult survival S_A and age at first reproduction *A*, and applying the formulae of Niel & Lebreton (2005), but required subsequent correction arising from estimation bias inherent in this method (Richard & Abraham 2013a, 2013b). Following recent (2016) discovery of errors in simulations used to derive the bias correction parameter, an updated approach was reviewed and approved via the AEWG in 2016 whereby r_{max} is estimated by applying an allometric power relationship between body mass M and taxonomic adult survival S_{tax} (see chapter 8). r_{max} and population size *N* are in turn used to estimate a PST via equation (30).

Ideally within the SEFRA method, biological parameters used in the derivation of r_{max} should be defined as inputs to the fully integrated Bayesian model (including representation of uncertainty for each parameter) instead of estimating r_{max} outside the model and defining a single input distribution. In this way uncertainty from biological input parameters propagates through the model, and output uncertainty can be tracked back to its source including uncertain biological inputs (see Section 3.3, Model Inputs, below).

3.2.4 CONSTRAINING PARAMETER INPUTS USING BIOLOGICAL MONITORING AND OTHER AVAILABLE DATA

Under the SEFRA framework uncertainty is reflected explicitly at every stage, (i.e., using ranges or distributions for every input parameter) and propagates through interim calculations through to output estimates of risk, wherever possible via Bayesian models. A major strength of this approach is that it becomes possible to use data sources other than observed captures to constrain model input parameters or impose priors, and this information then affects subsequent estimation of vulnerability and risk via the integrated model. Where model fits are in conflict with input distributions (e.g., high population survival estimates in conflict with high estimated fisheries mortality rates) the integrated model is forced to estimate what combination of parameter estimates is most plausible and revised parameter estimates are reflected in modified posterior distributions. In this way, where logical constraints on total FRDs can be defined as a function of biological and

demographic data (e.g., adult survival S, see below) population monitoring data serve to better estimate population level risk (rather than risk scores being a function of captures data only). Appropriately, the influence of non-captures data on model outputs will be stronger for those species and fisheries for which captures data are poor relative to population or demographic data (as will be the case for example for well-monitored seabird breeding colonies). Conversely, where capture rates are better estimated than demographic parameters, model fits based on captures can inform or constrain poorly informed estimates of population parameters and/or help to direct future population research.

3.2.4.1 CONSTRAINING SEABIRD CAPTURES USING ADULT SURVIVAL

Iterative development of the NZSRA illustrates the power of this approach. In previous iterations (up to Richard & Abraham 2015) there were seabird species for which fisheries risk was estimated to be very high, primarily as a consequence of observed multiple-capture events despite very low levels of observer coverage. This resulted in high (and highly uncertain) estimates of impact and risk for these species, for which the upper bound of the estimate extended to levels that, if actualised, would cause certain population decline. Nonetheless populations of some of these same species (e.g., black petrels, Chatham albatrosses) were observed to be approximately stable, and adult survival was high, suggesting that captures were overestimated in the risk assessment.

This difficulty was overcome in the latest (2017) update of the NZSRA by incorporating biological and population input parameters affecting estimation of the *PST* (i.e., adult survival, age at reproduction, population size) within the integrated model and constraining total FRDs such that the annual death rate cannot exceed maximum annual mortality suggested by the adult survival rate, i.e., [$D < (1 - S)$]. Model fits with this constraint indicated (for Chatham albatrosses) that vulnerability to capture was lower than previously modelled, such that revised estimates of FRDs are now consistent with population trend and mark-recapture data. For black petrels, the updated model suggests that population size *N* is likely to be higher than previously estimated, and/or that live release survival is significant (live release survival was not included in previous iterations of the NZSRA). That the integrated model can use observed capture rates to better estimate population parameters, and vice versa, is a major strength of the method, and provides tangible incentive to invest in population monitoring. Before these data were combined in an integrated model, there was no clear mechanism by which seabird population time series data were used to inform seabird fisheries risk, and risk assessment outputs were in conflict with population monitoring data.

3.2.5 PST VS. PBR

A key difference from the PBR approach of Wade (1998) is that the conservative population point estimate $N_{\min}$ has been replaced with a realistic estimate of N. Because Bayesian methods allow full statistical consideration of uncertainty in the input estimate of *N* (and other input parameters), the consequences of uncertain population size are now reflected as uncertain risk estimate outputs. Because N appears in equations at multiple stages in SEFRA method, utilising a biased estimator at the outset not only affects the definition of a sustainable impact level; it also affects estimates of *available population size* and *density overlap*, hence capture rate (equations (6)–(9)) , and the estimation of *vulnerability* from observed captures (equation (18)). For this reason it is preferable to adopt realistic estimates of N (including uncertainty) in the risk assessment stage; conservatism is better incorporated in the choice of a population outcome affecting φ within the PST formulation (equation (30)) or in the risk management stage distinct from risk assessment.

For the same reason, the PST formulation eliminates the use of the recovery factor *f*. The previous incorporation of *f* into PBR in equation (29) effectively confused risk assessment and risk management within the estimation of a single term, such that it was impossible when comparing PBR scores to distinguish between a species with low biological productivity and a less ambitious recovery factor (low r_{max}, high *f*) vs. a species with higher productivity and a more ambitious recovery factor (high r_{max}, low *f*). The improved PST formulation effectively gets the *f* out of PBR so that species with comparable risk scores in equation (28) can be expected to have a comparable population outcome, irrespective of management goals. Conservatism in setting management goals is more appropriately addressed in the choice of population recovery outcome inherent in the definition of φ in equation (30); defining different management objectives for different species is best addressed outside the risk assessment. Lonergan

(2011) in his critique of PBR and related approaches makes this same argument for maintaining the separation between risk assessment and risk management.

The φ parameter in equation (30) is defined with explicit reference to a population recovery or stabilisation outcome, and is estimated by simulation, including consideration of uncertainty and the effects of environmental stochasticity. This approach was first developed by Richard & Abraham (2013a). For the NZSRA and NZMMRA, New Zealand has defined a population reference outcome as follows: 'for impacts equal to or lower than PST (i.e., $R \leq 1$), the population will recover to or stabilise at a level at or above 50% of carrying capacity, with 95% certainty (and considering the effects of environmental stochasticity)'.

Alternative population recovery outcomes are of course possible, to reflect for example different management goals or different levels of tolerance for uncertainty; these would be expected to vary especially between risk assessments devoted to different kinds of taxa (e.g., for non-target fish vs. for marine mammals). Corresponding values of φ can be explored using population simulations particular to the taxa in question.

The original seabird-focused simulations in Richard & Abraham (2013a) used this same population recovery outcome above corresponding to R=1, but with a different mathematical formulation (with an adjustment parameter called ρ), in which the ρ parameter performed more than one function simultaneously, i.e., correcting biological parameter estimation bias as well as adjusting for the population recovery outcome. The improved PST formulation in the NZMMRA and updated NZSRA is designed to separate these roles so that φ serves only as an adjustment factor reflecting the population outcome; the parameter bias correction function also previously addressed within ρ is now addressed separately via improved parameter estimation methods first reviewed by the AEWG in 2016 (see Chapter 8).

Table 3.2: Mean species-level risk estimates, disaggregated by target fisheries. Highlighted cells (increasing yellow-red) identify fisheries that are responsible for an increasing proportion of species-level risk. Target fisheries with zero risk to all species (rounded to two decimal places) are not shown; these include: albacore SLL, minor SLL, jack mackeral trawl, and grey mullet set net. Likewise, species for which mean total risk ratio rounds to zero are not shown.

Species	PBR	Inshore trawl	Flatfish trawl	Bluenose BLL	Hapuka BLL	Minor BLL	Snapper BLL	Small ling BLL	Large ling BLL	Small STN SLL	Large STN SLL	Bigeye SLL	Swordfish SLL	Hoki trawl	Hake trawl	Ling trawl	SBW trawl	Scampi trawl	Squid trawl	Middle depth trawl	Deepwater trawl	Flatfish SN	Shark SN	Minor SN	TOTAL RISK RATIO
Black petrel	437	0.20	0	0.18	0.06	0.04	0.22	0	0	0	0	0.29	0.03	0.01	0	0	0	0.01	0	0.01	0	0	0	0	1.15
Salvin's albatross	3598	0.30	0.03	0	0	0.01	0	0.09	0	0	0	0	0	0.12	0	0.01	0.01	0.08	0	0.09	0.02	0	0	0	0.78
Flesh-footed shearwater	1451	0.29	0.01	0	0.04	0.04	0.19	0	0	0	0	0.02	0.01	0.01	0	0	0	0.03	0	0.01	0	0	0	0	0.67
Westland petrel	350	0.15	0.05	0	0.01	0.01	0	0.02	0	0.04	0	0.01	0	0.07	0.01	0	0	0	0	0.03	0	0.01	0.01	0	0.48
Southern Buller's albatross	1369	0.03	0.01	0	0	0	0	0.02	0	0.04	0.01	0	0	0.14	0.01	0.01	0	0.01	0.05	0.04	0	0	0	0	0.39
Chatham Island albatross	425	0.01	0	0	0.01	0.01	0	0.20	0.01	0	0	0	0	0.02	0	0	0	0	0	0.01	0.06	0	0	0	0.36
New Zealand white-capped albatross	10915	0.15	0.05	0	0	0	0	0	0	0.01	0	0.01	0	0.04	0	0.01	0	0.01	0.03	0.03	0	0	0	0	0.35
Gibson's albatross	496	0	0	0	0	0	0	0	0	0.07	0	0.04	0.19	0	0	0	0	0	0	0	0	0	0	0	0.34
Northern Buller's albatross	1628	0.01	0	0	0	0	0	0.02	0	0.03	0	0.07	0	0.03	0	0	0	0.03	0	0.02	0	0	0	0	0.25
Antipodean albatross	364	0	0	0	0	0	0	0	0	0.05	0	0.02	0.10	0	0	0	0	0	0	0	0	0	0	0	0.20
Otago shag	285	0.01	0.13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.14
Northern giant petrel	336	0	0	0	0.02	0.01	0.01	0	0	0	0	0	0	0.03	0	0	0	0.01	0	0.01	0.01	0	0	0	0.14
Spotted shag	3710	0.02	0.06	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.09
Yellow-eyed penguin	287	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01	0.04	0.01	0.08
Campbell black-browed albatross	1980	0	0	0	0	0	0	0	0	0.02	0	0.01	0	0.01	0	0	0	0	0	0	0	0	0	0	0.08
White-chinned petrel	25626	0	0	0	0	0	0	0.02	0.01	0	0	0	0	0.01	0	0	0	0.01	0.01	0	0	0	0	0	0.05
Northern royal albatross	716	0	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.04
Foveaux shag	207	0	0.03	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.04
Grey petrel	5526	0	0	0	0	0	0	0.01	0	0.01	0	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0.04
Southern royal albatross	848	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02
Snares Cape petrel	1601	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01
Little black shag	338	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01
fishery group total -- all species		1.19	0.38	0.19	0.14	0.11	0.41	0.39	0.03	0.27	0.02	0.47	0.34	0.49	0.02	0.03	0.02	0.18	0.09	0.24	0.10	0.02	0.05	0.02	5.72

3.2.6 STANDARD RISK ASSESSMENT OUTPUTS

Data inputs and analytical pathways utilised in the current NZSRA are shown in Figure 3.1. A standard output of the NZSRA, showing risk (with uncertainty) at the species level, is shown in Figure 3.1. Note that species-level outputs are less useful to managers charged with managing risks arising from particular fisheries across multiple species simultaneously.

Table 3.2 disaggregates species-level risk by fishery group, and highlights those species for which individual fishery groups are responsible for a substantial portion of species-level risk (more than 0.1 PST). Managers concerned about the fate of a particular bird species read across the row to identify fishery groups generating risk to that species; managers responsible for a particular fishery read down a column to see what bird species are affected, and to what extent.

Because Table 3.2 disaggregates mean risk; the representation of uncertainty is lost. For this reason Table

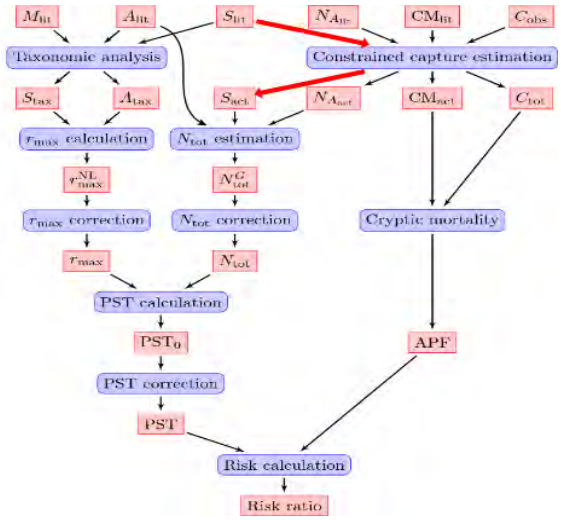


Figure 3.1: Diagram of data inputs and calculation pathways utilised in the current (2017) iteration of the NZSRA. The use of (realised) adult survival estimates to constrain captures and fishery-related deaths is highlighted.

3.2 should always be considered simultaneously with Figure 3.2 rather than in isolation. New work is underway to create a customised query tool to disaggregate and estimate impact and risk, including uncertainty, according to any

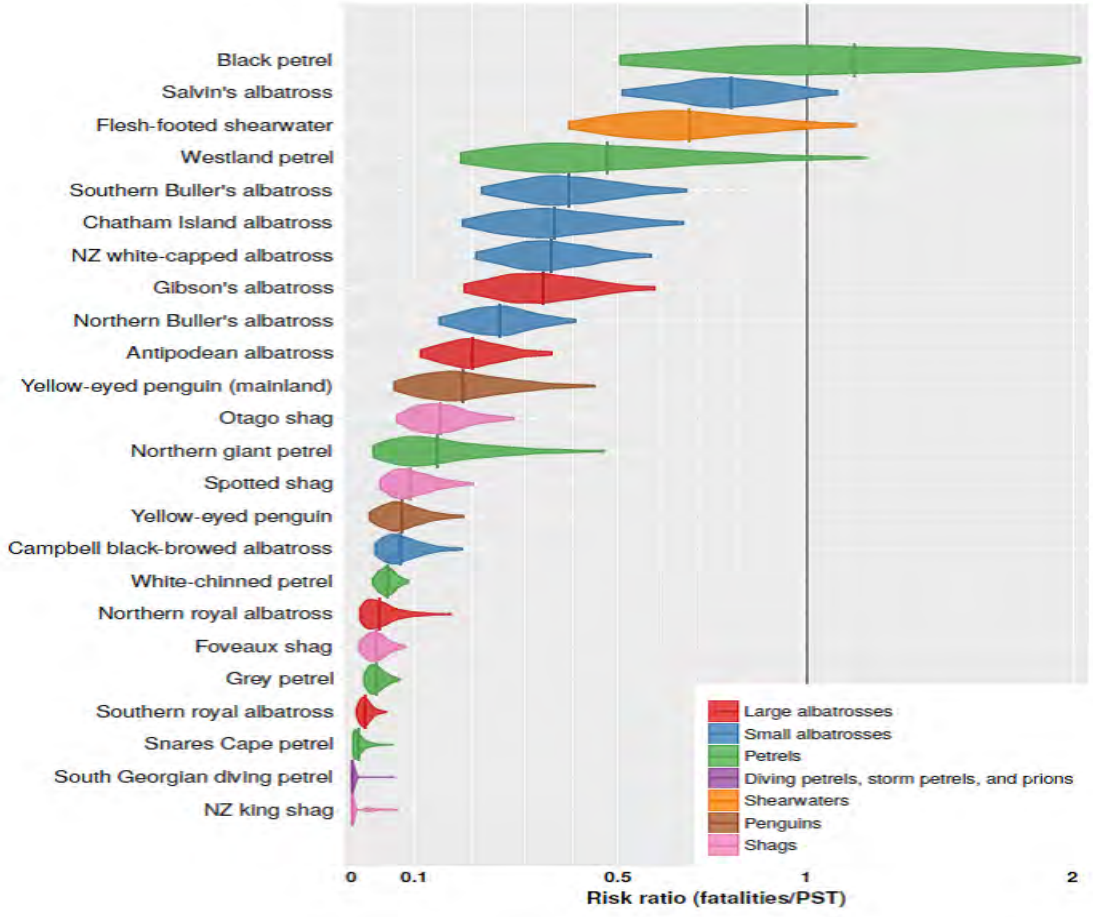


Figure 3.1: Standard species-level output of the NZSRA (from Richard et al. 2017). Species risk is shown on the x axis; the vertical line at R=1 corresponds to the level of *all human-induced mortality* that the species can sustain while still meeting the population recovery

user-defined criterion without loss of information. Since 2013 the NZSRA has included the results of sensitivities designed to track the propagation of uncertainty from input parameters through to resultant uncertainty of output estimates of risk; an example (from 2015) is shown in Figure 3.2. These figures have proven to be highly valuable to inform research prioritisation model inputs.

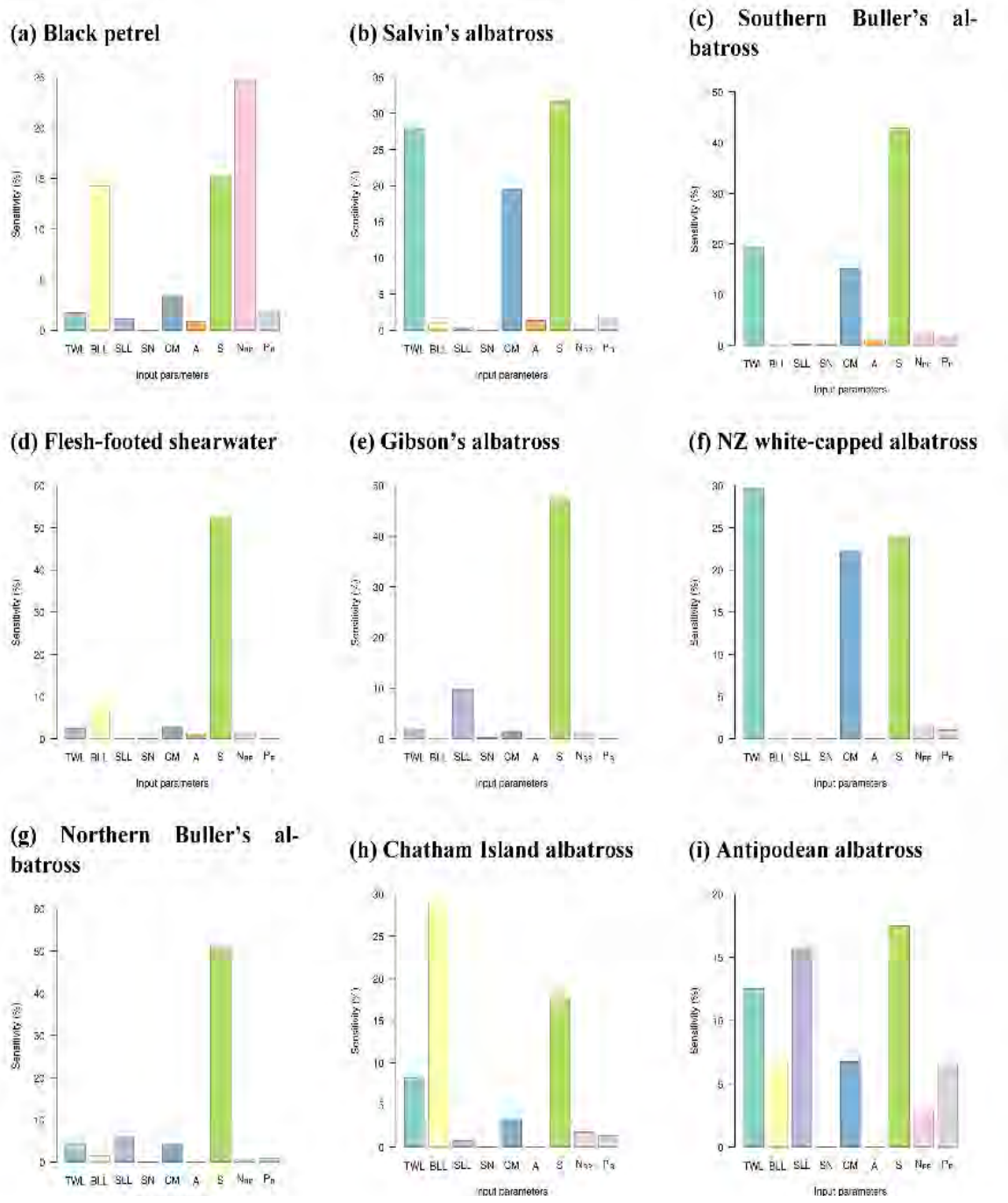


Figure 3.2: Example plot showing the propagation of uncertainty from uncertain input parameters through to output estimates of risk for at-risk species in the NZSRA. Parameters contributing to high levels of output uncertainty become a priority for future research. Note that this figure derives from an out of date version of the NZSRA (Richard et al. 2015) and is provided as an example only. Legend is as follows: TWL = vulnerability in trawl fisheries; BLL = vulnerability in bottom-longline fisheries; SLL = vulnerability in surface-longline fisheries; SN = vulnerability in set-net fisheries; CM = cryptic mortality; A = age at first reproduction; S = adult survival; N_{BP} = breeding population size; P_B = proportion breeding.

Table 3.3: Species (vulnerability) groups and cryptic mortality groups used in the 2017 iteration of the NZSRA.

Common name	Vulnerability group	ρ group	Cryptic mortality group
Gibson's albatross	Wandering albatrosses	Large albatrosses	Large albatrosses
Antipodean albatross	Wandering albatrosses	Large albatrosses	Large albatrosses
Southern royal albatross	Royal albatrosses	Large albatrosses	Large albatrosses
Northern royal albatross	Royal albatrosses	Large albatrosses	Large albatrosses
Campbell black-browed albatross	Campbell black-browed albatross	Small albatrosses	Mollymawks & giant petrel
New Zealand white-capped albatross	White-capped albatross	Small albatrosses	Mollymawks & giant petrel
Salvin's albatross	Salvin's albatross	Small albatrosses	Mollymawks & giant petrel
Chatham Island albatross	Chatham albatross	Small albatrosses	Mollymawks & giant petrel
Grey-headed albatross	Grey-headed albatross	Small albatrosses	Mollymawks & giant petrel
Southern Buller's albatross	Buller's albatrosses	Small albatrosses	Mollymawks & giant petrel
Northern Buller's albatross	Buller's albatrosses	Small albatrosses	Mollymawks & giant petrel
Light-mantled sooty albatross	Light-mantled sooty albatross	Large albatrosses	Mollymawks & giant petrel
Northern giant petrel	Giant petrel	Giant petrel	Mollymawks & giant petrel
Grey petrel	Grey petrel	Black petrel	Medium-sized seabirds
Black petrel	Black petrel	Black petrel	Medium-sized seabirds
Westland petrel	Westland petrel	Black petrel	Medium-sized seabirds
White-chinned petrel	White-chinned petrel	Black petrel	Medium-sized seabirds
Flesh-footed shearwater	Flesh-footed shearwater	Shearwaters	Medium-sized seabirds
Wedge-tailed shearwater	Shearwaters	Shearwaters	Small-sized seabirds
Buller's shearwater	Shearwaters	Shearwaters	Small-sized seabirds
Sooty shearwater	Sooty shearwater	Shearwaters	Medium-sized seabirds
Fluttering shearwater	Shearwaters	Shearwaters	Small-sized seabirds
Hutton's shearwater	Shearwaters	Shearwaters	Small-sized seabirds
Little shearwater	Shearwaters	Prions	Small-sized seabirds
Snares Cape petrel	Cape petrel	Prions	Small-sized seabirds
Fairy prion	Prions	Prions	Small-sized seabirds
Antarctic prion	Prions	Prions	Small-sized seabirds
Broad-billed prion	Prions	Prions	Small-sized seabirds
Pycroft's petrel	Pterodroma petrels	Prions	Small-sized seabirds
Cook's petrel	Pterodroma petrels	Prions	Small-sized seabirds
Chatham petrel	Pterodroma petrels	Prions	Small-sized seabirds
Mottled petrel	Pterodroma petrels	Prions	Small-sized seabirds
White-naped petrel	Pterodroma petrels	Shearwaters	Medium-sized seabirds
Kermadec petrel	Pterodroma petrels	Shearwaters	Medium-sized seabirds
Grey-faced petrel	Pterodroma petrels	Shearwaters	Medium-sized seabirds
Chatham Island taiko	Pterodroma petrels	Shearwaters	Medium-sized seabirds
White-headed petrel	Pterodroma petrels	Shearwaters	Medium-sized seabirds
Soft-plumaged petrel	Pterodroma petrels	Prions	Small-sized seabirds
Common diving petrel	Diving petrels	Diving petrels	Small-sized seabirds
South Georgian diving petrel	Diving petrels	Diving petrels	Small-sized seabirds
New Zealand white-faced storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
White-bellied storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
Black-bellied storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
Kermadec storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
New Zealand storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
Yellow-eyed penguin	Yellow-eyed penguin	Large penguins	Diving seabirds
Northern little penguin	Blue penguins	Small penguins	Diving seabirds
White-flippered little penguin	Blue penguins	Small penguins	Diving seabirds
Southern little penguin	Blue penguins	Small penguins	Diving seabirds
Chatham Island little penguin	Blue penguins	Small penguins	Diving seabirds
Eastern rockhopper penguin	Crested penguins	Small penguins	Diving seabirds
Fiordland crested penguin	Crested penguins	Small penguins	Diving seabirds
Snares crested penguin	Crested penguins	Small penguins	Diving seabirds
Erect-crested penguin	Crested penguins	Small penguins	Diving seabirds
Australasian gannet	Boobies and gannets	Shags	Diving seabirds
Masked booby	Boobies and gannets	Shags	Diving seabirds
Pied shag	Solitary shags	Shags	Diving seabirds
Little black shag	Solitary shags	Shags	Diving seabirds
New Zealand king shag	Solitary shags	Shags	Diving seabirds
Otago shag	Group foraging shags	Shags	Diving seabirds
Foveaux shag	Group foraging shags	Shags	Diving seabirds
Chatham Island shag	Group foraging shags	Shags	Diving seabirds
Bounty Island shag	Group foraging shags	Shags	Diving seabirds
Auckland Island shag	Group foraging shags	Shags	Diving seabirds
Campbell Island shag	Group foraging shags	Shags	Diving seabirds
Spotted shag	Group foraging shags	Shags	Diving seabirds
Pitt Island shag	Solitary shags	Shags	Diving seabirds
Subantarctic skua	Gulls, terns & skua	Shags	Medium-sized seabirds
Southern black-backed gull	Gulls, terns & skua	Terns	Medium-sized seabirds
Caspian tern	Gulls, terns & skua	Terns	Medium-sized seabirds
White tern	Gulls, terns & skua	Terns	Medium-sized seabirds

3.3 MODEL INPUTS

3.3.1 STRUCTURAL INPUT: SPECIES VULNERABILITY GROUPS

Conceptually, species vulnerability is the probability that an individual animal encountering a fishing event will be captured or fatally injured in that encounter. Vulnerability includes both catchability (animals captured alive or dead) and cryptic mortality. Species are assigned to *species vulnerability groups* (hereafter *species groups*) within which physical and behavioural characteristics are assumed to be similar, such that a single vulnerability score (per fishery group) can be assigned per species group. In this way observed capture rates for abundant and/or commonly observed species serve to inform the estimation of catchability and vulnerability for all species in the same group, even species which for captures are rarely or never observed. Where species groups are not used (i.e., capture rates for every species are modelled independently) statistical estimation of the species vulnerability for rarely captured species is unconstrained (such that for example in Richard et al. (2011), vulnerability and total risk scores for rare bird species varied by more than two orders of magnitude). Species vulnerability groups currently applied in the NZSRA are shown in Table 3.3. Species groups applied in the NZMMRA are shown in Table 3.4.

Assigning species to species groups should be done with care, informed by expert knowledge of species behaviour influencing fishery interactions, to ensure that superficial physiological or taxonomic similarity within the group does not conceal significant behavioural differences between species that result in real differences in vulnerability.

Group assignments should be examined with reference to model diagnostics (e.g., Figure 3.4 and Figure 3.8, below) and redefined as necessary to improve model fits. For example in Richard & Abraham (2013b) royal albatrosses and wandering albatrosses were grouped together. In 2013, visual examination of observed vs. expected capture patterns for these species revealed that the model was over-estimating capture rates for royal albatrosses and under-estimating capture rates of wandering albatrosses, evidently reflecting behavioural differences in the way these species react to fishing vessels. When the species group was subsequently split (Richard & Abraham 2015), visual fits improved markedly and the model estimated a significantly higher vulnerability for wandering albatrosses than for royal albatrosses.

Table 3.4: Species (vulnerability) groups used in the first (2017) iteration of the NZMMRA.

Species group	Common name
Whales	Antarctic blue whale
	Fin whale
	Pygmy blue whale
	Sei whale
	Humpback whale
	Southern right whale
	Sperm whale
	Bryde's whale
	Antarctic minke whale
	Pygmy right whale
	Dwarf minke whale
	Pygmy sperm whale
Blackfish	Killer whale Type A
	Long-finned pilot whale
	Short-finned pilot whale
	False killer whale
Beaked whales	Cuvier's beaked whale
	Shepherd's beaked whale
	Southern bottlenose whale
	Gray's beaked whale
	Spade-toothed whale
	Dense-beaked whale
	Andrews' beaked whale
	Hector's beaked whale
	Strap-toothed whale
Dolphins	Southern right whale dolphin
	Bottlenose dolphin
	Common dolphin
	Dusky dolphin
	Hourglass dolphin
	Hector's dolphin
	Māui dolphin
Pinnipeds	Southern elephant seal
	New Zealand sea lion
	New Zealand fur seal

3.3.2 FISHERY GROUPS

Non-target species capture rates are modelled separately within each of several broadly defined *fishing methods*. The NZSRA defines four such fishing methods: trawls, bottom longlines, surface longlines, and set nets; the NZMMRA defines also a fifth group, purse seines. All fishing effort is assigned to *fishery groups* within which the gear configuration and vessel behaviour is sufficiently consistent that species vulnerability estimates can be estimated and applied uniformly to all effort in the fishery group. Fishery groups are nested subsets of fishing methods.

Fishery group assignments should be informed by expert knowledge and based upon vessel characteristics known to affect non-target species interactions and capture rates, and defined with reference to variables universally stored in fishing effort databases (or otherwise recoverable such that all fishing events can be unambiguously assigned to groups). Variables used to distinguish between fishery groups are nearly always proxies for other underlying vessel characteristics, such that the means by which fishery groups are defined with reference to available data should utilise specific expert knowledge and should be investigated and iteratively adjusted with reference to the underlying data.

To illustrate, in the seabird risk assessment, trawl fishery groups are distinguished on the basis of vessel size and Table 3.5. Fishery group assignments utilised in the current iteration of the NZMMRA are shown in Table 3.6.

By its nature mitigation uptake is expected to reduce fishery group vulnerability; therefore vessels consistently utilising different mitigation configurations should be assigned to different fishery groups. For example, in Table 3.6, trawl fishery vessels using Sea Lion Exclusion Devices (SLEDs) are assigned to a different group from vessels not employing SLEDs.

target species (as a proxy for gear configuration), and of on-board offal processing capability (i.e., affecting seabird attraction). As with the assignment of species to species vulnerability groups, there is an inherent trade-off between increased specificity in group assignments vs. decreased statistical power arising from fewer observed captures per group, such that fishery group assignments should be made also with reference to the underlying availability of data (to ensure adequate data in each group). For this reason groups in the NZMMRA are more broadly defined than in the NZSRA (i.e., because there are fewer marine mammal captures than seabird captures).

Fishery group assignments utilised in the current iteration of the NZSRA are shown in

Where mitigation uptake is uneven or unverifiable across a fleet, and/or not recorded in a standardised format in fishing effort databases, fishery group vulnerability will be poorly estimated and the effectiveness of the mitigation to reduce species risk will not be quantifiable in risk assessment outputs. Standardised mitigation reporting (in contrast to qualitative recording e.g., in observer logbooks) and the ability to verify uptake (e.g., via electronic monitoring) will increase the utility of the SEFRA method to detect mitigation efficacy and inform risk management decisions.

Table 3.5: Fishing methods and fishery groups used in current (2017) iteration of the NZSRA.

Trawl	Small inshore < 17 m	Targeting inshore species (including flatfish), or targeting middle-depth species (principally hoki, hake, or ling) on vessels less than 17 m length
	Small inshore < 28 m	Targeting inshore species (including flatfish), or targeting middle-depth species (principally hoki, hake, or ling) on vessels more than 17 m length and less than 28 m length
	Southern blue whiting	Targeting southern blue whiting
	Scampi	Targeting scampi
	Mackerel	Targeting mackerel (primarily jack mackerel species)
	Squid	Targeting squid
	Large processor	Targeting middle-depth species, vessel longer than 28 m, processing fish on board
	Large fresher	Targeting middle-depth species, vessel longer than 28 m, with no processing on board
	Deepwater	Targeting deepwater species (principally orange roughy or oreos)
Bottom longline (BLL)	Bluenose	Targeting bluenose, and vessel less than 34 m length
	Snapper	Targeting snapper, and vessel less than 34 m length
	Ling and ribaldo	Targeting ling or ribaldo, and vessel less than 34 m length
	Other small BLL vessels	Not targeting snapper, bluenose, ling, or ribaldo, and vessel less than 34 m length
	Large vessels without IWL	Vessels over 34 m, without integrated weight line
	Large vessels with IWL	Vessels over 34 m, with integrated weight line
Surface longline (SLL)	Swordfish	Targeting swordfish, and vessel less than 45 m length
	Other small SLL vessels	Not targeting swordfish, and vessel less than 45 m length
	Large vessels	Vessel 45 m or longer
Set net	Set net	All set-net fishing

Table 3.6: Fishing methods and fishery groups used in the first (2017) iteration of the NZMMRA.

Method	Fishery	Annual effort	Observed effort
Bottom longline	BLL	37 567	65 157
Purse seine	PS	1 285	1 481
Surface longline	SLL	2 611	18 299
Set net	SN	20 557	4 823
Trawl	Pelagic trawl	2 349	17 991
	Pelagic trawl (sled)	547	1 645
	Squid trawl	1 415	20 913
	Squid trawl (sled)	799	8 533
	Inshore trawl	48 340	9 522
	Other trawl	29 100	105 764

3.3.3 SPECIES INPUTS: SPATIAL DISTRIBUTIONS

To inform the calculation of *overlap* between species and fishing effort in equations (1) and (2), the spatial distribution of each species is mapped throughout the spatial domain of the risk assessment. Species distribution layers are defined such that the value in a particular cell represents the probability that an individual animal, selected at random from the population, is present in that cell at the moment of the fishing event. For each species the value of all cells sums to 1 across the spatial domain.

Because overlap is estimated per event (rather than based on cell-aggregated summaries of fishing effort) cell size is computationally unimportant but is necessarily consistent across all species so that the resulting vulnerability estimates are likewise comparable between species.

The New Zealand seabird and marine mammal risk assessments utilise species distribution maps assembled from multiple sources, including mapped distributions from vessel-based and aerial surveys, satellite tracking data, foraging 'hotspots' delineated using expert knowledge, density gradients as a function of distance from breeding colonies, and expert-based distributions assembled via 'Delphi' workshop methods. An example species distribution, for Gibson's albatross, is shown in Figure 3.3.

To map the distributions of species for which direct observation and/or tracking is not feasible, spatial habitat models may be employed using spatially comprehensive environment data (e.g., SST, bathymetry, turbidity) as a proxy for species distribution. Successful methods of relating species distribution to underlying environmental data include the application of subjectively defined Relative Environmental Suitability models for widely distributed

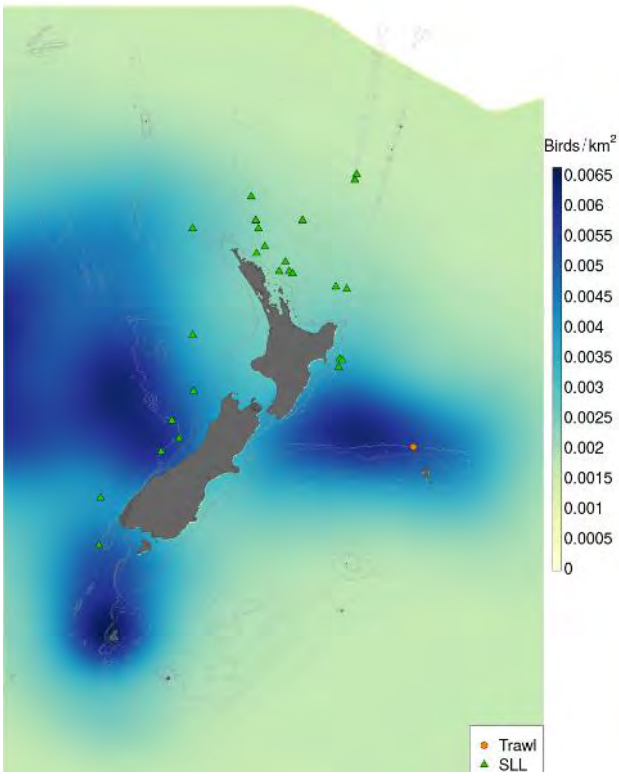


Figure 3.3: Spatial distribution layer (derived from global tracking data) for Gibson's albatross. Capture events are also shown.

marine mammals (Kaschner et al. 2006, 2011) or the use of sophisticated multivariate statistical methods such as Boosted Regression Trees or State-Space modelling, fitted to fisheries or trawl survey catch data (Leathwick et al. 2006, Pinkerton et al. 2010). Note however that by nature spatial habitat models map the full range of the *potential species habitat*, which may be substantially larger than the actual realized distribution, especially where the actual distribution reflects historical range contraction associated with population decline, or complex behavioural patterns or lifecycle movements that cannot be captured using

environmental proxies, as is often the case for marine mammals and seabirds.

In general, fish or invertebrate distributions will more often rely on environmental proxies, whereas protected species distributions will more often rely on tracking or aerial census. MPI is currently progressing work to improve estimation of marine mammal distributions as inputs to the NZMMRA (projects PRO2014-01 and PRO2017-10).

It is generally difficult to include statistical uncertainty in spatial data represented as maps; early iterations of the NZSRA either assumed that species distribution maps were precisely known, or represented uncertainty as a simple binary sensitivity between alternative maps. A superior approach is to subjectively define a normal distribution around the estimation of the overlap term O_{sg} in equation (2), with a CV reflecting the degree of confidence in the underlying spatial distributions. For example the confidence assigned to animal distributions defined from tracking studies or aerial surveys >> habitat distribution modelling >> maps derived from subjective expert knowledge. By applying spatial uncertainty to the O_{sg} term in equation (2) rather than the θ_{sg} term in equation (5), spatial uncertainty from species maps is not confounded with population size uncertainty affecting estimates of actual species density.

3.3.3.1 SEASONALLY VARIABLE DISTRIBUTIONS

Seasonally variable species distributions (e.g., for migratory species) can be used at whatever level of seasonal resolution the distribution data will support, without loss of statistical power.

The NZSRA currently applies two spatial distributions per year, i.e., breeding season and non-breeding season distributions, with the duration of each season defined individually for each species at the scale of months. In contrast, the CCSBT global risk assessment, which relied primarily on electronic tracking data to define global seabird distributions, split tracking data into four seasons (summer/autumn/winter/spring) for all seabird species alike.

Because catchability and vulnerability are estimated as a function of overlap across all fishing events simultaneously regardless of year or season, defining a higher number of seasonal distributions does not result in a loss of statistical power. The underlying assumption is that interaction rate is proportional to encounter rate regardless of season (i.e., vulnerability is constant throughout the year). However where seasonally variable animal behaviour results in changed vulnerability (e.g., if nesting seabirds target fishing boats more aggressively during the chick-rearing period) then it may be useful instead to estimate vulnerability in each season separately (i.e., using q_{zt} in place of q_z in equation (27)). Seasonally variable q should only be considered where sufficient data are available in each season to inform the estimation, and by testing the model's ability to discern seasonally variable vulnerability using model diagnostics (Figure 3.4 and Figure 3.8 below).

3.3.3.2 TRANSIENT OR SEASONALLY ABSENT SPECIES

For highly migratory species that leave the spatial domain of the risk assessment entirely (e.g., New Zealand seabirds that leave the EEZ outside the breeding season), spatial distribution layers are not modified and species overlap O from equations (1) and (2) remains unchanged. Instead, the changed encounter rate is reflected by recording the proportional change to *available population size* as in equation (5).

3.3.4 BIOLOGICAL POPULATION SIZE

Risk is necessarily estimated with reference to a biologically meaningful estimate of population size N. Applications of the SEFRA method in New Zealand to date have estimated species level risk at the scale of the New Zealand breeding population (i.e., not considering transient species and not differentiating between local sub-populations) except in particular instances where locally important sub-populations have been specifically identified, and captures can be unambiguously assigned to that local population. To illustrate, in the NZSRA, since Richard & Abraham (2013b), risk to the small mainland population of yellow-eyed penguins is assessed separately from that to the large Snares Island population, and in the NZMMRA risk is estimated separately for Māui vs. Hector's dolphins. Note however that because vulnerability is an inherent property of the species and is estimated at the species group level in equation (18), there is no loss of statistical power if impact and risk is subsequently disaggregated and applied at the scale of smaller subpopulations. This option will be applied to regional subpopulations of Hector's dolphins in the review of the Threat Management Plan (see Chapter 6), and

may be applied to other coastal marine mammal species in the next iteration of the NZMMRA.

For protected species populations, input estimates of biological population size N should utilise the most recent available estimates, e.g., derived from population census, mark-recapture, genetic mark-recapture, or other methods. Because captures are estimated with reference to the entire vulnerable population, estimates derived from breeding colony census must be scaled upwards to also include non-breeders, or, as in the NZSRA, the breeding and non-breeding populations are estimated separately and assigned their own spatial distributions, which are subsequently combined. All input distributions are defined using priors reflecting estimated uncertainty.

3.3.5 AVAILABLE POPULATION SIZE

The use of *available population size* $\mathbb{N}_i$ in equations (5)-(7) recognises that the number of individuals actually present in the spatial domain of the risk assessment at the moment of fishing event *i* may be different than the size of the biological population N against which impacts are evaluated.

The means by which available population size is reflected in the current NZSRA is by estimating, for every migratory species, the proportion of breeding and non-breeding seabirds that are present within the domain of the NZSRA in the breeding and non-breeding seasons. To illustrate, for a migratory bird species for which half of the population is absent from New Zealand waters during the non-breeding season, the available population $\mathbb{N}_i$ = 0.5 N, and expected number of captures associated with fishing events during the non-breeding is correspondingly reduced. In situations where an entire population leaves the spatial domain of the risk assessment on a seasonal basis, $\mathbb{N}_i$= 0 in that period.

Seasonal adjustments of this nature are necessary because the estimation of vulnerability occurs across all fishing events simultaneously. In the example of a migratory bird that is seasonally absent, if the $\mathbb{N}_i$ adjustment were not used, the model would nonetheless 'expect' captures on observed fishing events in the period when the bird is absent, and the effect of the recorded zero capture events would then depress the estimated vulnerability $q_z q_g$, leading to underestimation of capture rates and risk in the period when the bird is once again present.

Where animals present in New Zealand are merely a subset of a single globally distributed population (e.g., many cetacean species) the notion of a 'New Zealand population' may have no biological meaning; in these instances risk should be estimated with reference to the full global population, for which the presence of only a subset of that population in New Zealand waters at any given time is represented by estimating a permanently lower *available population size* $\mathbb{N}_i$ (i.e., $\mathbb{N}_i < N$ for all i).

Note also that in some instances it is possible to have an available population size $\mathbb{N}_i$ that is higher than the biological population N, for example if biological risk is evaluated with reference to a small local population, but observed capture rates reflect the presence of abundant transient individuals from other breeding populations outside the spatial domain of the risk assessment. This was the case for giant petrels in early iterations of the NZSRA (Waugh et al. 2009, Richard et al. 2011) in which giant petrel risk was artificially inflated because all captures were originally assumed to originate from a very small local population despite the presence of transient birds from an abundant overseas population.

Proportional adjustments in available population sizes for breeding and non-breeding populations in the current NZSRA are shown in Table 3.7. Similarly, for wide-ranging marine mammal species in the NZMMRA it is necessary to estimate what proportion of the population is present in the New Zealand domain at a given time.

Table 3.7: Biological and seasonally adjusted (i.e., non-breeding season) available population sizes applied in the 2017 iteration of the NZSRA.

Common name	Scientific name	Breeding period		Biological N	% staying in NZ
		Start	End		
Gibson's albatross	*Diomedea antipodensis gibsoni*	-	-	24 200	-
Antipodean albatross	*Diomedea antipodensis antipodensis*	-	-	17 900	-
Southern royal albatross	*Diomedea epomophora*	-	-	41 800	-
Northern royal albatross	*Diomedea sanfordi*	-	-	35 200	-
Campbell black-browed albatross	*Thalassarche impavida*	Aug	May	81 400	50
New Zealand white-capped albatross	*Thalassarche cauta steadi*	Nov	Aug	457 000	50
Salvin's albatross	*Thalassarche salvini*	Sep	Apr	139 000	10
Chatham Island albatross	*Thalassarche eremita*	Aug	May	17 700	2.5
Grey-headed albatross	*Thalassarche chrysostoma*	Sep	May	29 100	20
Southern Buller's albatross	*Thalassarche bulleri bulleri*	Dec	Aug	52 500	2.5
Northern Buller's albatross	*Thalassarche bulleri platei*	Oct	Jun	62 600	2.5
Light-mantled sooty albatross	*Phoebetria palpebrata*	Sep	Jun	32 200	20
Northern giant petrel	*Macronectes halli*	Aug	Feb	14 900	75
Grey petrel	*Procellaria cinerea*	Feb	Nov	190 000	2.5
Black petrel	*Procellaria parkinsoni*	Oct	Jul	17 900	0.5
Westland petrel	*Procellaria westlandica*	Mar	Dec	12 100	2.5
White-chinned petrel	*Procellaria aequinoctialis*	Nov	May	922 000	20
Flesh-footed shearwater	*Puffinus carneipes*	Sep	May	40 400	0.5
Wedge-tailed shearwater	*Puffinus pacificus*	Oct	May	149 000	0.5
Buller's shearwater	*Puffinus bulleri*	Sep	May	1 330 000	0.5
Sooty shearwater	*Puffinus griseus*	Nov	May	18 500 000	0.5
Fluttering shearwater	*Puffinus gavia*	Jul	Feb	781 000	80
Hutton's shearwater	*Puffinus huttoni*	Sep	Apr	363 000	2.5
Little shearwater	*Puffinus assimilis*	Apr	Nov	493 000	5
Snares Cape petrel	*Daption capense australe*	Nov	Feb	43 800	90
Fairy prion	*Pachyptila turtur*	Mar	Jan	6 420 000	15
Antarctic prion	*Pachyptila desolata*	Nov	Mar	3 170 000	15
Broad-billed prion	*Pachyptila vittata*	Feb	Jan	1 490 000	5
Pycroft's petrel	*Pterodroma pycrofti*	Oct	Apr	8 440	0
Cook's petrel	*Pterodroma cookii*	Sep	Apr	1 050 000	0.5
Chatham petrel	*Pterodroma axillaris*	Nov	Jun	868	0
Mottled petrel	*Pterodroma inexpectata*	Oct	May	1 180 000	0
White-naped petrel	*Pterodroma cervicalis*	Oct	May	178 000	0
Kermadec petrel	*Pterodroma neglecta*	-	-	20 200	-
Grey-faced petrel	*Pterodroma macroptera gouldi*	Mar	Jan	839 000	10
Chatham Island taiko	*Pterodroma magentae*	Sep	May	58	20
White-headed petrel	*Pterodroma lessonii*	Aug	May	993 000	10
Soft-plumaged petrel	*Pterodroma mollis*	Sep	May	13 300	0.5
Common diving petrel	*Pelecanoides urinatrix*	Mar	Jan	2 930 000	20
South Georgian diving petrel	*Pelecanoides georgicus*	Sep	Feb	208	0
New Zealand white-faced storm petrel	*Pelagodroma marina maoriana*	Sep	Apr	5 040 000	0.5
White-bellied storm petrel	*Fregetta grallaria grallaria*	Apr	Aug	3 550	100
Black-bellied storm petrel	*Fregetta tropica*	Oct	May	242 000	50
Kermadec storm petrel	*Pelagodroma albiclunis*	Jun	Dec	175	50
New Zealand storm petrel	*Pealeornis maoriana*	-	-	834	-
Yellow-eyed penguin	*Megadyptes antipodes*	Aug	Apr	6 540	100
Northern little penguin	*Eudyptula minor f. iredalei*	Jul	Feb	21 700	100
White-flippered little penguin	*Eudyptula minor f. albosignata*	Jul	Feb	6 740	100
Southern little penguin	*Eudyptula minor f. minor*	Jul	Feb	21 700	100
Chatham Island little penguin	*Eudyptula minor f. chathamensis*	Jul	Feb	21 700	100
Eastern rockhopper penguin	*Eudyptes chrysocome filholi*	Oct	May	203 000	5
Fiordland crested penguin	*Eudyptes pachyrhynchus*	Jul	Mar	13 300	50
Snares crested penguin	*Eudyptes robustus*	Sep	Feb	137 000	5
Erect-crested penguin	*Eudyptes sclateri*	Sep	Mar	403 000	50
Australasian gannet	*Morus serrator*	Aug	Mar	147 000	20
Masked booby	*Sula dactylatra*	-	-	775	-
Pied shag	*Phalacrocorax varius varius*	-	-	16 000	100
Little black shag	*Phalacrocorax sulcirostris*	Oct	Dec	4 060	100
New Zealand king shag	*Leucocarbo carunculatus*	Mar	Oct	629	100
Otago shag	*Leucocarbo chalconotus*	Aug	Mar	4 390	100
Foveaux shag	*Leucocarbo stewarti*	Aug	Mar	3 210	100
Chatham Island shag	*Leucocarbo onslowi*	Sep	Feb	1 190	100
Bounty Island shag	*Leucocarbo ranfurlyi*	Oct	Dec	412	100
Auckland Island shag	*Leucocarbo colensoi*	Nov	Mar	7 200	100
Campbell Island shag	*Leucocarbo campbelli*	Nov	Feb	7 090	100
Spotted shag	*Stictocarbo punctatus*	-	-	46 900	-
Pitt Island shag	*Stictocarbo featherstoni*	Sep	Feb	1 310	100
Subantarctic skua	*Catharacta antarctica lonnbergi*	Sep	Feb	1 620	50
Southern black-backed gull	*Larus dominicanus dominicanus*	Sep	Mar	6 800 000	100
Caspian tern	*Hydroprogne caspia*	Sep	Jan	3 010	100
White tern	*Gygis alba candida*	Sep	Apr	342	100

3.3.6 FISHING EFFORT DISTRIBUTIONS

Fishing effort is assigned to fishery groups as above and mapped in space. Note that the mathematical estimation of overlap in equation (1) is carried out for each individual fishing event and multiplied by the density of the species group in question at the particular location (equation (5)). Because fishing effort is not summarised spatially before calculating overlap, there is no need to consider the cell size at which fishing effort distributions are aggregated, except for display purposes.

An example fishing effort distribution is shown in Figure 3.5. The intersection of the species and effort distributions (Figure 3.3 and Figure 3.5) to estimate overlap is illustrated in Figure 3.4.

In New Zealand, most commercial fishing effort data are reported using spatially precise start and end locations per fishing event. However where fishing effort is reported only within larger statistical areas, it is necessary to assign all fishing events to specific points in space using logical assumptions (e.g., effort randomly distributed within statistical areas, or distributed as a function of proximity to land or ports, etc.). Because spatial overlap influences both the estimation of species and fishery group vulnerability from observed capture rates (equation (18)) and also the subsequent estimation of total captures including in unobserved effort (equation (16)) it is worthwhile to expend effort at the outset to define or model the distribution of fishing effort as accurately as possible.

Translating individual fishing events into mathematical estimates of overlap in equation (1) requires decisions about the units in which effort is expressed, e.g., numbers of deployments vs. length of trawls for trawl fisheries, or numbers of hooks vs. numbers of deployments for longlines. These decisions should be made with care, utilising expert knowledge of seabird-fishery interactions, and informed by exploration of the data to determine what units of effort most effectively model observed capture rates.

Standard units in which effort events are expressed in the MMRA are shown in the legend of Table 3.6. Unsurprisingly effort is expressed with reference to kilometres of net for setnets, and numbers of hooks for longline fisheries; but for trawl fisheries effort is expressed with reference to the

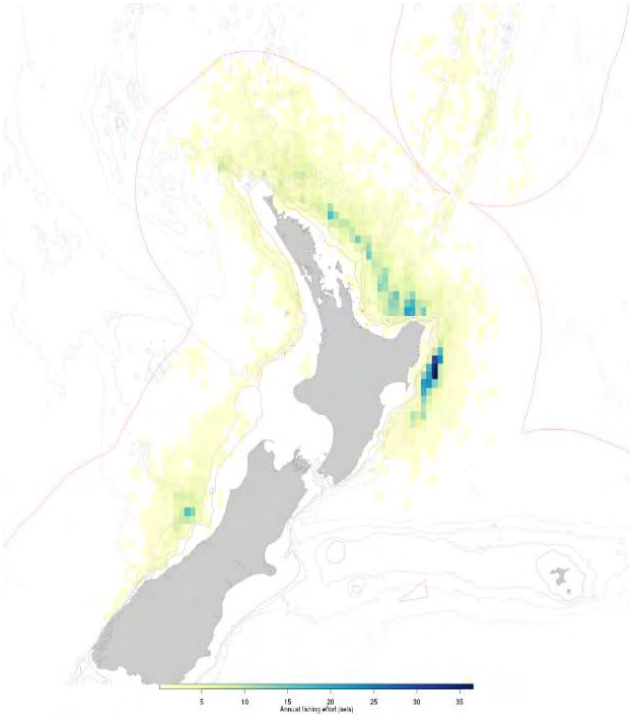


Figure 3.5: Fishing effort spatial distribution for the small (domestic) SLL fishery group not targeting swordfish, 2005–06 to 2014–15.

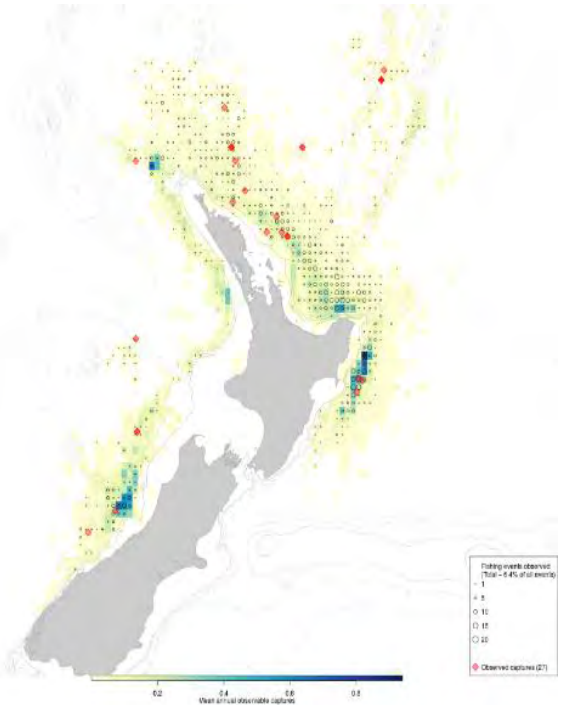


Figure 3.4: Overlap *O* of the distribution of Gibson's albatross (Figure 3.4) with the small (domestic) SLL fishery group (Figure 3.5). Black circles denote observed fishing effort. Capture events are also shown. Captures are expected to occur in space proportional to the intensity of the overlap in that location. Examination of expected vs. observed patterns of capture events in space is a primary diagnostic of model fit. This example fits poorly (i.e., captures occur disproportionately in the north in an area of lower overlap), suggesting the need to re-examine spatial inputs (species distributions) or model structural assumptions (e.g., fishery group definitions).

number of hauls only, independent of distance or duration (suggesting that most protected species captures occur at the time of the set or haul, rather than the tow). In contrast, the SEFRA method applied to fish or benthic impacts will by necessity reflect duration or length of tow for trawl fisheries, except perhaps for midwater or seamount trawl fisheries in which fishers target individual acoustic marks in a highly selective way. These decisions should be informed by experts with knowledge of the operational factors affecting vessel behaviour in the particular fisheries in question, and tested with reference to the data.

3.3.6.1 PARTIALLY OBSERVED FISHING EVENTS

The SEFRA method estimates species and fishery group catchability as a function of observed capture rates and overlap on the observed subset of the fishing effort data (equation (18)). Importantly, in some instances only a subset of a particular fishing event may be observed, effectively reducing observed fishing intensity a'_i in equation (1), (e.g., if a fisheries observer observes only a portion of a longline haul coming on board the vessel, and is off duty or occupied with other duties for the other portion). For this reason it is important that observer databases record what proportion of the event is observed, and that observed capture events distinguish between 'on duty' captures (i.e., caught during the observation period) vs. 'off duty' captures reported independently by the vessel. Whether off-duty captures are included in the estimation of catchability (hence vulnerability and risk) relies on assumptions about the reliability of vessel-reported capture data when an observer is not present to verify. Whether or not off-duty captures are used, accurate estimation is only possible if on-duty vs. off-duty captures are clearly distinguished in fishery databases (not merely in observer comments), and the observed proportion of each fishing event a'_i is recorded.

3.3.6.2 UNIDENTIFIED CAPTURES

Reliance on observed captures data creates a strong imperative to ensure that taxonomic ID by fisheries observers is accurate or subsequently verified by necropsy. However because estimation of species catchability in equation (18) occurs at the species group rather than the individual species level, taxonomic resolution below the level of species group is not required (except for example where observer data is also used to inform species distribution mapping). The taxonomic resolution and reliability of observer data should thus be considered in the stage at which species groups are being defined (i.e., there is no benefit in defining species groups at a finer level of taxonomic resolution than the observed captures can support).

3.3.7 YEARS' OBSERVED FISHING EFFORT DATA USED TO ESTIMATE CATCHABILITY

Because risk assessment approaches are designed for application to data-poor problems, there is an imperative in the estimation of catchability and vulnerability to include as much data as possible. At the same time, implicit in the assignment of fishing events to fishery groups is the underlying assumption that factors affecting capture rates by all fishing events in the same group are similar (or at least indistinguishable) within the fishery group. This assumption is violated in situations where vessels have changed their gear, or adopted mitigation measures, or otherwise changed their at-sea behaviour in ways that would be expected to change the probability of capture and/or cryptic death per encounter with non-target species.

Decisions about which years' data should be used in the estimation of species and fishery group vulnerability in equation (18) should be taken with care, with reference to available data indicative of observed capture rates, and informed by experts with relevant knowledge of fishery gear and at-sea operations and the history of changed practices affecting interactions with non-target species. Where a step-change in capture rates is likely (i.e., corresponding to new gear technology or new imposed regulations) data use should be restricted to the subset of the historical data representing current practice, or fishery data before and after the change should be assigned to different fishery groups. In the latter instance it may be possible to quantify the effect of the change on capture rates empirically, by comparing vulnerability estimates between the groups.

In the update of the NZSRA the AEWG considered as a sensitivity suggestions that deepwater fishery groups should be limited to fishing events post-2010, when new mitigation requirements were imposed and revised offal discard practices were widely implemented. For other fishery groups at present there are not sufficient data to evaluate whether or not capture rates have changed sufficiently to warrant limiting the input fishery data in this

way. The time period over which observed capture rate data is used to estimate vulnerability in the NZSRA are shown in

Table 3.5. Due to a lack of data, the current NZMMRA uses the full time period from which data are available.

3.3.7.1 TRACKING FISHERY PERFORMANCE AFFECTING VULNERABILITY OVER TIME

Because of the imperative to include as much data as possible, in the absence of an identifiable step change in fishing practice the SEFRA method is not well suited for tracking changing catchability over time (i.e., indicative of mitigation uptake or voluntarily changed at-sea practices). To detect change of this nature it is necessary to test alternate structural assumptions, i.e., running sensitivities using observer data from different time periods, and comparing the resulting estimates of catchability, vulnerability, and risk. (In contrast, changing spatio-temporal distributions of fishing effort are manifested in overlap rather than vulnerability, so are immediately apparent and easily tracked over time).

Furthermore because vulnerability estimation in equation (18) is integrated across all fishery groups simultaneously and informed by input priors that reflect information other than observed capture rates, changes in the estimated vulnerability can arise from multiple sources other than observed changes in the capture rate in the fishery group in question.

Where tracking changed performance over time in particular fisheries or subsets of fisheries is an imperative, it is necessary to develop dedicated tools for this purpose, i.e., to define particular queries and run sensitivities in which changed outputs arise only from the fishery in question while other inputs are held constant. MPI is progressing work to develop this capability (project PRO2016-06 and SEA2016-30).

3.3.8 YEARS' FISHING EFFORT DATA TO REPRESENT CURRENT EFFORT AND RISK

Once species and fishery group vulnerability have been estimated by the model described in equation (18), there is no longer an imperative to maximise the use of fishing effort data in the subsequent estimation of current impact on a species- and fishery-group-specific basis in equations (5)–(9). Instead, it is important to use the best available proxy for 'current' or expected future fishing effort. Generally the recent past is considered the best proxy for the immediate future, but where fishing effort trends are changing rapidly or future changes can be forecast (e.g., reflecting changed TACs, management boundaries or fleet composition) it may be worthwhile to apply alternative assumptions, or generate hypothetical spatial effort scenarios on a case by case basis.

As a default the NZSRA and NZMMRA use the most recent three years' fishing effort data to approximate the 'current' distribution of effort, and to estimate corresponding 'current' impact and risk.

3.3.9 CRYPTIC MORTALITY

The modelling step of the SEFRA method in equation (18) fits to data indicative of total observable captures. However biological risk is a function of deaths, not captures; the relationship between captures and FRDs is reflected in the estimation of cryptic mortality and live release survival rates in equations (10)–(11). Input parameters to inform these equations are almost always highly uncertain. Often some data may exist for the *live release rate* r_{sg}, but data to better estimate the *cryptic mortality multiplier* k_{sg} and *live release survival rate* L_{sg} are by nature difficult to obtain, generally requiring dedicated research projects. In the absence of data, it is necessary to estimate these parameters outside the model using expert knowledge, reflecting uncertainty as input priors.

Scientists and other technical experts are often reluctant to provide numerical estimates where the answers are highly uncertain, citing lack of data. But failure to explicitly consider cryptic mortality and live release survival within protected species risk assessments constitutes an implicit adoption of extreme values (0 or 1) with absolute certainty; this approach is far less defensible than applying subjective estimates with explicit priors reflecting actual uncertainty. Inclusion of highly uncertain parameters based on expert knowledge serves to illustrate for managers the real consequences of the current lack of knowledge regarding cryptic mortality, and creates positive incentives for fishers, both to modify at-sea behaviour (e.g., improved protected species handling protocols at sea to increase live release

survival) and to collect better data so that improved performance is reflected in reduced risk. Furthermore, in an integrated Bayesian multi-species model, ignoring these parameters may force the model to adopt skewed estimates of other important parameters in order to fit model constraints. For these reasons inclusion of even highly subjective parameter estimates in equations (10) and (11) is essential.

Experts who may initially profess their inability to estimate unknown parameters often find that collectively they 'know' far more than they expect, when confronted with the consequences of failing to provide an estimate (i.e., many experts are reluctant to propose a 'correct' estimate but quick to reject one that they 'know' to be 'wrong'). To capture this tendency effectively, highly uncertain subjective estimates are best elicited in a structured workshop setting, or via Delphi methods (e.g., as used in the 2016 NZMMRA).

3.3.9.1 CRYTIC MORTALITY GROUPS

Similar to species vulnerability groups, species are assigned to cryptic mortality groups, reflecting groups of species that are expected to interact with fishing gear in similar ways that will affect cryptic mortality rates. Cryptic mortality groups are more broadly defined than species vulnerability groups.

In the NZSRA, all seabird species are assigned to one of five such groups on the basis of body mass (affecting the amount of forward momentum with which they may be expected to interact with trawl warps and/or wing length that affects likelihood of warp entanglement) and also diving ability; see Table 3.3.

In the NZMMRA, cryptic mortality groups reflect body size and foraging behaviour affecting likely interactions with vessels (e.g., large toothed whales are considered separately from large baleen whales because depredation behaviour may lead to substantially increased entanglement risk in longlines). Cryptic mortality parameters are applied at the level of the five broadly defined fishing methods in Table 3.6.

3.3.9.2 INPUT PARAMETER DISAGGREGATION FOR IMPOROVED ESTIMATION OF CRYTPIC MORTAILITY

Where protected species may interact with fishing vessels in a variety of different ways, refined estimation of cryptic mortality rates is greatly aided by disaggregating the input parameters to distinguish between different types of interactions, to make maximum use of available data. The power of this approach is illustrated below with reference to the NZSRA, for which the most recent iteration estimates and applies different cryptic mortality parameters for each fishery group.

3.3.9.2.1 SEABIRDS IN TRAWL FISHERIES

In the first application of cryptic mortality within the NZSRA, Sharp et al. (2011) disaggregated the estimation of cryptic mortality multiplier k_{sg} in trawl fisheries as follows.

- Captures and/or mortality events are assumed to arise from three types of interaction:
  - Net captures
  - Surface warp strikes (bird resting or hovering at surface is overtaken and potentially entangled/drowned by a moving warp)
  - Aerial warp strikes (a flying bird strikes a warp under its own forward momentum).
- Warp captures vs. net captures are recorded separately by fisheries observers; using these data the estimated proportion of net captures can be estimated separately for each cryptic mortality group and fishery group, and applied to estimate group-specific cryptic mortality rates, as follows:
  - For net captured birds:
    - Live releases are recorded by fisheries observers; these data are used to estimate the live release rate, $r_{sg} - net$.
    - Live release survival $L_{sg} - net$ is estimated subjectively or requires dedicated research projects (e.g., banding or radio-tracking of live released birds)
    - The cryptic mortality multiplier $k_{sg} - net$ (reflecting drowned or injured birds that drop out of the net uncounted) is estimated subjectively or requires dedicated observation.
  - For warp captured birds:
    - All warp captures are assumed to arise from surface warp strikes.
    - No warp captured birds are assumed to be released alive ($r_{sg} - warp = 0$)
    - The surface strike cryptic mortality multiplier $k_{sg} - surf$ is estimated relative to observed surface captures based on

dedicated research projects (e.g., 'corpse catchers) or warp strike observational studies (e.g., Watkins et al. 2010, Abraham 2010)

- The aerial strike cryptic mortality multiplier $k_{sg} - air$ is estimated relative to surface captures, applying surface: aerial warp strike ratios and subjective estimates of the fate of aerial warp strikes from dedicated observational studies elsewhere (Watkins et al. 2008). These could be productively updated to also include use of more recent data (e.g., Parker et al. 2013).

The sequence by which disaggregated cryptic mortality parameters for trawl fisheries are combined to generate a total fisheries related deaths multiplier $\varkappa_{sg}$ as in equation (11) is displayed below in Figure 3.6.

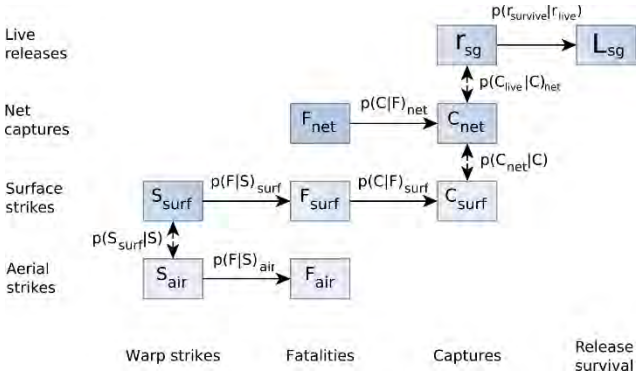


Figure 3.6: Transition probabilities by which the total fisheries-related deaths multiplier $\varkappa_{sg}$ (including live release survival as in equation (10)) is estimated for seabirds in trawl fishery groups.

Cryptic mortality rates consistent with this framework (but without incorporating live releases or distinguishing between different trawl fishery groups) were adopted in the 2013 iteration of the NZSRA (Richard & Abraham 2013b). The full framework was adopted in the 2017 iteration (summarised in Chapter 8). Importantly, the disaggregated cryptic mortality parameters in equation (10) including priors to represent uncertainty are incorporated as separate inputs into the integrated Bayesian multi-species risk model, rather than estimated outside the model and summarised as a single multiplier $\varkappa_{sg}$ from equation (11). In this way posteriors arising from the fitted model will help to refine poorly estimated cryptic mortality or live release parameters and/or to indicate where dedicated research projects may be useful to reduce uncertainty.

Utilisation of a 'corpse catcher' on trawl warps may provide empirical data to better estimate the rate at which fatal surface warp strikes result in an observed capture (i.e., $p(C|D_{surf})$ in Figure 3.6).

3.3.9.2.2 SEABIRDS IN LONGLINE FISHERIES

From the 2013 iteration the NZSRA has applied a total fisheries related deaths multiplier $\varkappa_{sg}$ for all longline fisheries, based on a single dedicated observational study in surface longline fisheries (Brothers et al. 2010). This approach can be substantively improved, e.g., by re-examining the Brothers et al. (2010) dataset to distinguish between species cryptic mortality groups, and by applying distinct assumptions regarding the fate of birds captured on the set vs. the soak vs. the haul (i.e., $k_{sg} - set$ will be higher than $k_{sg} - haul$, and live releases would be applied to haul-captured birds only). Furthermore, the use of these data primarily from global high seas SLL fisheries to estimate cryptic mortality in domestic SLL fishery groups, and the extension of these results also to BLL fishery groups, is untested.

From the 2017 iteration the NZSRA incorporates live release rate (separately for BLL vs. SLL, using New Zealand-specific data) and live release survival (subjectively estimated with high uncertainty). A dedicated research project is in the planning stages using dead geese and ducks as proxies for large and medium seabirds caught on the set, to better estimate $k_{sg} - set$.

Pierre et al. (2015) make specific further recommendations for improvement of cryptic mortality parameter estimation.

3.3.9.3 ESTIMATING CHANGING CRYPTIC MORTALITY OVER TIME

Because cryptic mortality multipliers have a direct and potentially dramatic effect on total FRDs in equations (10) and (11), but are not necessarily reflected in observed capture events by which species vulnerability is estimated in equation (18), it is plausible that changed fishery practices affecting cryptic mortality and/or live release rates may occur without any corresponding change in observed capture rates, hence vulnerability and risk. If such changes are likely then the factors underlying estimation of cryptic mortality need to be examined in a temporally explicit way, so that constant capture rates don't potentially mask substantially changing death rates.

To illustrate, in trawl fisheries, seabird net captures will accrue a fairly low fisheries-related deaths multiplier (likely less than 2) because relatively few diving birds are thought to drown but fall out of the net uncounted ($k_{sg} - net$ is low), and a substantial proportion of flying birds entrapped by the meshes on the outside of the net are released alive and may survive ($r_{sg} - net$ and $L_{sg} - net$ are non-zero). In contrast, warp captures may accrue a very high cryptic mortality rate because: i) surface struck birds dragged underwater and drowned on the warps are only recovered if their bodies are subsequently impaled on a sprag or otherwise entangled in the gear; ii) aerial warp strikes may result in fatal injuries such broken wings, with no mechanism for body recovery leading to a recorded capture; and iii) there are no warp captured birds released alive. For these reasons a capture on the warp implies a higher number of actual deaths, hence greater risk, relative to a capture in the net.

It is therefore possible that changes to seabird mitigation and offal discard practices over time that have the effect of Table 3.5. This will have the likely effect of reducing FRD multipliers for those (well-observed) fisheries and species for which the proportion of net captures has increased relative to warp captures, and increasing the uncertainty associated with FRD multipliers for other poorly observed fishery groups.

3.3.10 DEMOGRAPHIC AND BIOLOGICAL PARAMETERS

3.3.10.1 SPECIES BIOLOGICAL INPUTS

Biological parameters are derived from available data or published proxies and defined as input distributions reflecting uncertainty. The SEFRA method applied to protected species requires sufficient biological parameter inputs to inform the estimation of $r_{max,}$, for use in equation (30). In the 2017 iteration of the NZSRA these include age at reproduction and adult survival, both of which are in turn derived from allometric relationships with body mass (Chapter 8). In contrast the NZMMRA uses published literature values for $r_{max,}$; other future applications of the SEFRA framework (e.g., for non-target fish or benthic invertebrates) will use alternative means of representing intrinsic species productivity to derive a MIST as in equation (29).

shifting captures from the warp to the net could occur with little to no observable change in estimated capture rate and vulnerability, effectively disguising a substantial reduction in total FRDs and species risk if changes to cryptic mortality and total fisheries related deaths multipliers were considered. This effect may have occurred in some New Zealand deepwater fisheries, for which there is an increasing trend in the proportions of net captures and of live released birds since changed mitigation and offal management practices began to be adopted from around 2005 (Figure 3.7). It is likely that this trend reflects a shift in the species composition of captured birds – away from mollymawk species primarily caught on the warp, and toward medium sized and diving birds, more often caught in the net. The 2017 iteration of the NZSRA applies the observed ratio of net: warp captures for different fishery groups individually, and estimates group-specific fisheries-related deaths multipliers at the level of each combination of cryptic mortality species group x fishery group in Table 3.3 and

A similar modification should be considered in future to distinguish between SLL and BLL fishery groups based on the proportion of captures on the set vs. on the haul.

3.3.10.2 POPULATION DEMOGRAPHIC INPUTS

As described above, population monitoring data may be used to define constraints on total fisheries-related deaths within an integrated model (as in the 2016 NZSRA). In this way the SEFRA method allows utilisation of all available biological, demographic, and fisheries observer data to inform estimates of fisheries impact and risk simultaneously across all fishery groups and species groups.

It is important however to distinguish between the *taxonomic / ideal* biological parameters affecting species productivity and the estimation of $r_{max,}$ (above) vs. *actual/ realised* parameters specific to the impacted population in question. The former inputs represent intrinsic characteristics of the species and may legitimately be sourced from published data from overseas populations, or derived from allometric and life history relationships for the species in question (as in the 2016 NZSRA), or estimated by analogy with similar proxy species. In contrast,

demographic parameters used to constrain fisheries impacts must necessarily come from direct observations of the particular impacted population, and must be both reliable and current (i.e., reflective of the same time period over which fisheries effort data are included in the risk assessment). To illustrate, in the 2016 NZSRA, adult survival S appears in both calculation pathways of **Error! Reference source not found.**, informing the estimation of r_{max} via the left-hand path and constraining total fishery-related deaths via the right-hand path. This model distinguishes between the 'taxonomic' (un-impacted ideal) adult survival S_{tax} affecting estimation of r_{max} on the left, vs. the 'actual/realised' adult survival S_{act} for the impacted population in question, to constrain impact estimates on the right. Using demographic monitoring data to constrain impacts within the Bayesian model is a powerful innovation but should applied cautiously and only using quality data. Adoption of this innovation within the SEFRA method creates powerful incentive to fund and deliver population monitoring research to better inform fisheries risk assessment.

Populations for which adult survival is used to constrain total fishery-related deaths in the 2017 NZSRA, and the source of demographic parameter estimates used to define this constraint, are shown in Table 3.8.

3.3.11 MODEL DIAGNOSTICS

A primary means of testing spatial parameter inputs and structural assumptions and evaluating model fit is to examine spatial patterns of expected vs. observed captures on a species- and fishery-group-specific basis, as in Figure 3.4. These maps should be produced and evaluated routinely for every combination of species group x fishery group that produces substantial risk for any at-risk species (e.g., highlighted in Table 3.2). Where spatial fits are good, observed captures should show the same spatial pattern as the underlying observed overlap. Poor spatial fits should prompt further investigation either of spatial data inputs (i.e., animal distribution layers) or structural assumptions (e.g., species and fishery group definitions, seasonal variation in available population size), which may be iteratively adjusted and re-evaluated until spatial fits improve.

Similarly, expected vs. observed capture estimates should be evaluated across all fishery group x species group combinations simultaneously, as in Figure 3.8. Outliers prompt further investigations.

Table 3.8: Realised adult survival S_{actual}, used to constrain total fishery related deaths in the integrated model of the 2017 NZSRA. Annual fishery-related deaths are constrained to be less than 1 minus adult survival (D < (1-S)). This is a precautionary constraint, allowing that all deaths are attributable to fisheries (i.e., neglecting natural mortality).

Species	Prior Mean	Prior 95% c.i.	Posterior Mean	Posterior 95% c.i.
Gibson's albatross	0.962	0.939–0.984	0.962	0.932–0.983
Antipodean albatross	0.956	0.941–0.968	0.957	0.942–0.969
Southern royal albatross	0.949	0.931–0.963	0.949	0.932–0.964
Northern royal albatross	0.938	0.910–0.967	0.938	0.898–0.967
Campbell black-browed albatross	0.945	0.930–0.957	0.945	0.930–0.958
NZ white-capped albatross	0.959	0.935–0.975	0.959	0.937–0.976
Salvin's albatross	0.966	0.941–0.982	0.960	0.939–0.974
Chatham Island albatross	0.966	0.940–0.982	0.965	0.942–0.982
Grey-headed albatross	0.952	0.932–0.968	0.952	0.932–0.968
Southern Buller's albatross	0.955	0.931–0.979	0.955	0.923–0.977
Northern Buller's albatross	0.955	0.931–0.979	0.955	0.923–0.978
Light-mantled sooty albatross	0.970	0.961–0.980	0.970	0.958–0.980
Northern giant petrel	0.887	0.812–0.960	0.887	0.781–0.959
Grey petrel	0.935	0.902–0.968	0.934	0.888–0.969
Black petrel	0.927	0.899–0.947	0.926	0.901–0.947
Westland petrel	0.947	0.919–0.974	0.946	0.910–0.973
White-chinned petrel	0.935	0.902–0.968	0.935	0.891–0.969
Flesh-footed shearwater	0.935	0.931–0.940	0.935	0.930–0.940
Wedge-tailed shearwater	0.924	0.891–0.956	0.924	0.880–0.959
Buller's shearwater	0.915	0.841–0.963	0.915	0.840–0.967
Sooty shearwater	0.918	0.863–0.976	0.919	0.836–0.972
Fluttering shearwater	0.923	0.891–0.956	0.923	0.877–0.958
Hutton's shearwater	0.923	0.891–0.956	0.923	0.881–0.957
Little shearwater	0.923	0.891–0.956	0.924	0.882–0.957
Snares Cape petrel	0.855	0.776–0.935	0.855	0.744–0.938
Fairy prion	0.837	0.771–0.889	0.837	0.774–0.892
Antarctic prion	0.837	0.769–0.891	0.838	0.772–0.893
Broad-billed prion	0.838	0.773–0.891	0.838	0.774–0.893
Pycroft's petrel	0.933	0.846–0.978	0.934	0.852–0.984
Cook's petrel	0.932	0.843–0.978	0.932	0.847–0.983
Chatham petrel	0.933	0.849–0.979	0.933	0.854–0.983
Mottled petrel	0.933	0.848–0.978	0.933	0.857–0.982
White-naped petrel	0.933	0.847–0.978	0.933	0.850–0.983
Kerm. petrel	0.933	0.850–0.978	0.933	0.849–0.982
Grey-faced petrel	0.933	0.844–0.978	0.934	0.854–0.983
Chatham Island taiko	0.932	0.848–0.978	0.933	0.854–0.982
White-headed petrel	0.933	0.846–0.978	0.934	0.854–0.984
Soft-plumaged petrel	0.933	0.845–0.978	0.933	0.853–0.983
Common diving petrel	0.811	0.753–0.867	0.811	0.739–0.875
South Georgian diving petrel	0.810	0.753–0.867	0.809	0.737–0.874
NZ white-faced storm petrel	0.896	0.826–0.945	0.896	0.830–0.948
White-bellied storm petrel	0.896	0.823–0.946	0.895	0.824–0.951
Black-bellied storm petrel	0.896	0.825–0.946	0.897	0.832–0.948
Kerm. storm petrel	0.896	0.821–0.946	0.896	0.824–0.949
NZ storm petrel	0.895	0.822–0.945	0.895	0.824–0.950
Yellow-eyed penguin	0.866	0.799–0.919	0.866	0.798–0.920
Northern little penguin	0.829	0.788–0.865	0.829	0.789–0.866
White-flippered little penguin	0.829	0.786–0.865	0.829	0.789–0.867
Southern little penguin	0.829	0.787–0.865	0.829	0.787–0.867
Chatham Island little penguin	0.829	0.786–0.867	0.830	0.786–0.867
Eastern rockhopper penguin	0.840	0.816–0.860	0.840	0.818–0.862
Fiordland crested penguin	0.840	0.818–0.861	0.840	0.817–0.860
Snares crested penguin	0.840	0.818–0.860	0.840	0.818–0.860
Erect-crested penguin	0.840	0.816–0.860	0.840	0.818–0.861
Australasian gannet	0.934	0.850–0.978	0.934	0.852–0.983
Masked booby	0.848	0.781–0.900	0.849	0.785–0.903
Pied shag	0.878	0.860–0.896	0.878	0.856–0.899
Little black shag	0.878	0.860–0.896	0.878	0.856–0.899
NZ king shag	0.878	0.860–0.896	0.878	0.856–0.899
Otago shag	0.878	0.860–0.896	0.878	0.855–0.898
Foveaux shag	0.878	0.860–0.896	0.878	0.855–0.899
Chatham Island shag	0.878	0.860–0.896	0.878	0.856–0.899
Bounty Island shag	0.878	0.860–0.896	0.878	0.856–0.898
Auckland Island shag	0.878	0.860–0.896	0.878	0.856–0.899
Campbell Island shag	0.878	0.860–0.896	0.878	0.855–0.899
Spotted shag	0.878	0.860–0.896	0.879	0.856–0.900
Pitt Island shag	0.878	0.860–0.896	0.878	0.855–0.899
Subantarctic skua	0.941	0.911–0.969	0.941	0.903–0.970
Southern black-backed gull	0.808	0.743–0.861	0.808	0.744–0.864
Caspian tern	0.877	0.820–0.933	0.878	0.802–0.937
White tern	0.805	0.781–0.829	0.805	0.776–0.832

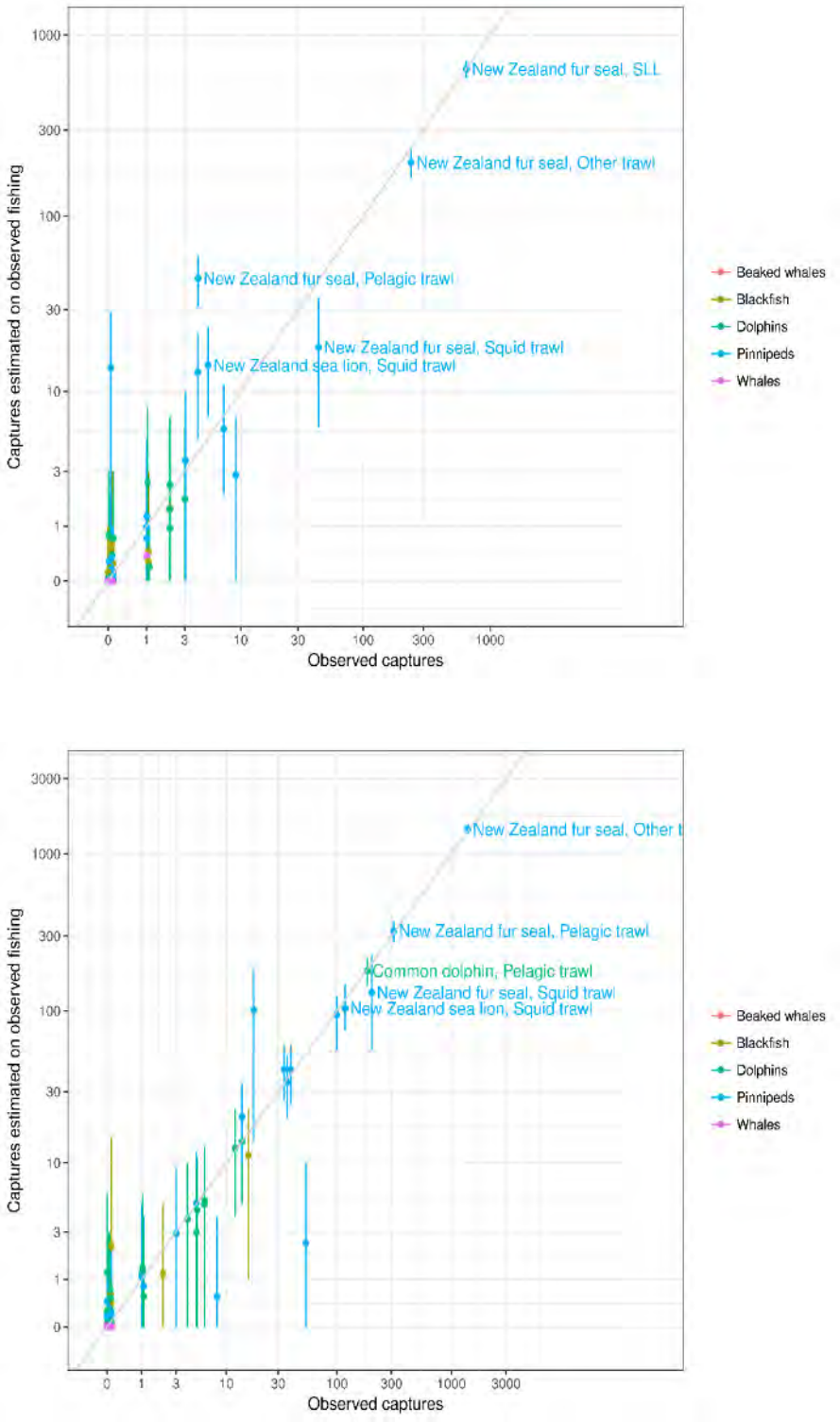


Figure 3.7: Example model diagnostics plot showing observed vs. expected numbers of live captures (top) and dead captures (bottom) for each fishery and species group combination in the 2016 NZMMRA.

3.4 ALTERNATIVE IMPLEMENTATIONS OF THE SEFRA FRAMEWORK

Alternate applications of the SEFRA method are currently planned or in development. To the extent possible, these will be developed to be conceptually and terminologically consistent with the framework described above, noting however that every individual risk assessment will be customised to address the particular nature of the specific problem and to make maximum use of the available data. For these reasons individual risk assessments will develop and apply different specific methodologies as required on a case by case basis.

3.4.1 SPECIES-SPECIFIC SEABIRD AND MARINE MAMMAL ASSESSMENTS

Where the multi-species marine mammal and seabird risk assessments indicate that fisheries risk is likely to be substantial for particular species of interest, separate species-specific implementations may be warranted to enable a more thorough understanding of available data. Species-specific implementations are already in progress for Māui and Hector's dolphins (SEA2016-30 and PRO2017-12) and for New Zealand sea lions in the Subantarctic Islands (PRO2017-10). New projects are under consideration also for New Zealand fur seals and for sea lions at newly established colonies on the New Zealand mainland.

Focusing on a particular species allows structural decisions to be tailored appropriately (i.e., using fishery group definitions or seasonally variable spatial distributions that are tailored to reflect interactions with only the species of interest). These projects will also allow disaggregation of species-level risk outputs to examine risk at a subpopulation level, and to examine sensitivities or evaluate risk management options (e.g., spatial management vs. mitigation vs. effort transition between fishery groups).

Single species SEFRA models also allow consideration of additional data reflecting covariates that may be of particular importance to some species but not to others. To illustrate, project PRO2016-02 will expand on the SEFRA framework to build a multivariate model predicting captures of black petrels and flesh-footed shearwaters by longline fisheries. As in the basic SEFRA method, capture rates are primarily a function of encounter rate, estimated via spatial overlap between species and fisheries. But the expanded model will also incorporate additional covariates thought to particularly affect black petrel and flesh-footed shearwater interactions with fisheries, e.g., moon phase, time of day, and mitigation uptake. In a multi-species model, the effects of these covariates would be diluted and likely impossible to discern. The outputs of PRO2016-02 are expected to provide insight into factors most responsible for driving fisheries captures, to inform the design of risk management options for these important species.

3.4.2 GLOBAL SEABIRD RISK ASSESSMENT

A global (southern hemisphere) seabird risk assessment is in progress to estimate out-of-zone risk to globally distributed New Zealand species. The methodological framework is as described above; available global seabird distributions are as utilised in Waugh et al. (2012). A primary challenge of this work is the poor quality of available observed captures data required to characterise global fishing effort and define meaningful fishery groups (reflecting different fishing behaviour and different levels of mitigation uptake between fleets) and thereby estimate fishery group vulnerability v_g. Species group vulnerability v_z can usefully be applied by proxy from the same or similar species in the NZSRA.

3.4.3 PELAGIC PROTECTED FISH SPECIES

For large, solitary, rare and/or protected fish species generally captured in single-capture events (e.g., pelagic sharks) it is likely that the most effective approach will apply a nearly identical mathematical formulation to that described above for seabirds and marine mammals, so long as population abundance data are available. Genetic mark-recapture methods or genetic half-sibling analyses may prove useful to obtain an estimate of absolute population size.

The primary challenge of a pelagic shark risk assessment under this approach will be to adequately represent highly dynamic spatial distributions in time; this may be achievable by applying sophisticated multi-variate habitat models (e.g., Leathwick et al. 2006, Pinkerton et al. 2010) parameterised using habitat affinity data from satellite tracked individuals, to define seasonal distributions and adjust available population size on a seasonal basis to reflect large-scale movements of pelagic fish species.

Where adequate population data are lacking and only fisheries-dependent data are available to model spatial distributions (e.g., many pelagic sharks), an alternative approach such as that developed by Fu et al. (2016) may be applied.

It is likely that any pelagic protected fish risk assessment could also be usefully extended to marine reptiles (turtles).

3.4.4 NON-TARGET FISH (TRAWL FISHERIES)

An application of the SEFRA method is currently under consideration for non-target fish species captured as bycatch in deepwater trawl fisheries, and for low information inshore fish stocks. Application of the method framework to non-target fish would follow the conceptual framework of the SEFRA method described above, but with substantial modifications of the analytical pathways outlined in equations (1)–(30), reflecting differences in data availability to inform input parameterisation. Application of the SEFRA method to protected species vs. bulk-capture bycatch species follows a similar estimation formulation as in equation 18, but the (relative) knowns and unknowns are reversed. For protected species such as seabirds and marine mammals, population size is generally known with some degree of precision (e.g., from genetic methods, breeding colony census) but capture events are sufficiently rare as to make estimation of catchability and/or vulnerability challenging; thus N and O are used to estimate q. In contrast, for non-target fish species, N is unknown but captures data are generally much richer; thus population size must be estimated from catchability q, which must in turn be estimated by other means (e.g., Zhou et al. 2009, 2011, Sibanda et al. 2016).

Because fishing gear is designed to retain fish, cryptic mortality is unlikely to be as important for bulk captured fish as for protected species, perhaps rendering the distinction between vulnerability and catchability unnecessary and eliminating the need for cryptic mortality multipliers (except for example to reflect small fish escaping through trawl meshes).

At least in trawl fisheries, because captures arise from passive interaction with gear rather than active behavioural attraction to fishing gear (as is the case with seabirds) estimation of q will by necessity include parameters for swept area and probably also a parameter for vertical availability in the water column, distinct from catchability parameters representing capture efficiency within the swept area.

Because fish are actively targeted, and because fish capture and retention in trawls is determined by both species-specific morphological and behavioural characteristics and fishery-group specific gear performance and efficiency, the structural assumptions behind the disaggregation of the vulnerability / catchability parameter into its species-group-specific and fishery-group-specific components is violated; catchability will by necessity be estimated per fishery group x species group combination (q_{sg} not $q_s q_g$).

Because schooling fish are captured in bulk, it will likely be necessary to estimate catchability as the product of two capture estimation models, one for probability of capture per fishing event and a separate model for abundance in those events in which the species is captured.

All of these modifications are under consideration by MPI contracted scientists; preliminary progress is described in Roux et al. (2015) and Sibanda et al. (2016). Subsequent extension to non-target inshore fish will be considered as one available method of the Low Information Stocks Project (LISP), subject to limitations on the ability to accurately estimate spatial distributions.

3.4.5 BENTHIC INVERTEBRATES AND/OR STRUCTURAL HABITATS

The SEFRA method is analogous to and fully compatible with spatially explicit benthic impact assessment methods for example as previously described in Sharp et al. (2009) and developed further by Mormede & Dunn (2012). The primary obstacle to full implementation of the SEFRA method for benthic invertebrates is the inherent difficulty of modelling benthic invertebrate spatial distributions given the sparse and scale-dependent nature of available environmental data to inform habitat models, and poor captures data with which to estimate the relationship between habitat and biology. For this reason the initial implementation of the impact assessment in Sharp et al. (2009) estimated impacts per spatial cell but without reference to the taxonomic composition of the benthic community; hence without an effective 'population size' there was no means of defining an impact threshold analogous to the MIST of Roux et al. (2015). Availability of improved high-resolution bathymetric and oceanographic spatial data layers to inform spatial habitat models may

make full implementation of SEFRA method increasingly feasible for benthic invertebrate taxa.

Because fishing gear is not designed to retain benthic invertebrates, and damage to benthic habitats occurs regardless of to what extent benthic material is retained, modification of the SEFRA method for bottom fishing impacts will focus exclusively on vulnerability rather than catchability, using swept-area methods, thus eliminating any need to consider cryptic mortality. Growth and recovery factors analogous to the use of r_{max} can be used to model taxon-specific population responses to different spatially explicit impacts, and nett effects on multi-species composition (as in Mormede & Dunn 2012, Pitcher et al. 2016), to inform some objective basis to define a maximum impact threshold, analogous to MIST for non-target fish. Alternatively, impact can be mapped spatially using the overlap approach with traits-based vulnerability estimation (Sharp et al. 2009, Roux et al. 2016).

3.5 REFERENCES

Abraham, E R (2010) Warp strike in New Zealand trawl fisheries, 2004–05 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 60*. 29 p.

Abraham, E R; Berkenbusch, K; Neubauer, P; Richard, Y (2017) Assessment of the risk to New Zealand marine mammals from commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 189*. 123 p.

Abraham, E R; Richard, Y (2017) Summary of the capture of seabirds in New Zealand commercial fisheries, 2002–03 to 2013–14. *New Zealand Aquatic Environment and Biodiversity Report No. 184*. 88 p.

Brothers, N; Duckworth, A R; Safina, C; Gilman, E L (2010) Seabird bycatch in pelagic longline fisheries is grossly underestimated when using only haul data. *PLoS ONE 5*: e12491. doi:10.1371/journal.pone.0012491.

Campbell, M L; Gallagher, C (2007) Assessing the relative effects of fishing on the New Zealand marine environment through risk analysis. *ICES Journal of Marine Science 64*: 256–270.

Crawford, C (2003) Qualitative risk assessment of the effects of shellfish farming on the environment in Tasmania, Australia. *Ocean and Coastal Management 46*: 47–58.

Currey, R J C; Boren, L J; Sharp, B R; Peterson, D (2013) A risk assessment of threats to Maui's dolphins. Ministry for Primary Industries and Department of Conservation, Wellington, New Zealand. SC/65a/Forinfo38. International Whaling Commission Scientific Committee, Jeju Island, South Korea. 51 p.

Fu, D; Roux, M J; Clarke, S C; Francis, M; Dunn, A; Hoyle, S (2016) Pacific-wide sustainability risk assessment of bigeye thresher shark (*Alopias superciliosus*) Final Report. *NIWA Client Report 2016089WN*, for project WCP16301. 98 p. Retrieved from https://cites.org/sites/default/files/eng/cop/17/InfDocs/E-CoP17-Inf-63.pdf.

Hobday, A J; Smith, A D M; Webb, H; Daley, R; Wayte, S; Bulman, C; Dowdney, J; Williams, A; Sporcic, M; Dambacher, J; Fuller, M; Walker, T (2007) Ecological Risk Assessment for the Effects of Fishing: Methodology. Report R04/1072 for the Australian Fisheries Management Authority, Canberra. 174 p.

Kaplan, S (1997) The words of risk analysis. *Risk Analysis 17 (4)*: 407–417.

Kaschner, K; Tittensor, D P; Ready, J; Gerrodette, T; Worm, B (2011) Current and future patterns of global marine mammal biodiversity. *PLoS One 6*: e19653.

Kaschner, K; Watson, R; Trites, A; Pauly, D (2006) Mapping world-wide distributions of marine mammal species using a relative environmental suitability (RES) model. *Marine Ecology Progress Series 316*: 2–3.

Leathwick J; Elith, J; Francis, M; Hastie, T; Taylor, P (2006) Variation in demersal fish species richness in the oceans surrounding New Zealand: an analysis using boosted regression trees. *Marine Ecology Progress Series 321*: 267–281.

Lonergan, M (2011) Potential biological removal and other currently used management rules for marine mammal populations: a comparison. *Marine Policy 35*: 584–589.

Lonergan, M E; Phillips, R A; Thomson, R B; Zhou, S (2017) Independent review of New Zealand's Spatially Explicit Fisheries Risk Assessment approach – 2017. *New Zealand Fisheries Science Review 2017/2*. 36 p.

Ministry for Primary Industries (2013) National Plan of Action – 2013 to reduce the incidental catch of seabirds in New Zealand fisheries. Ministry for Primary Industries, Wellington, New Zealand.

Mormede, S; Dunn, A (2012) An initial development of spatially explicit population models of benthic impacts to inform Ecological Risk Assessments in New Zealand deepwater fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 106.* 20 p.

Niel, C; Lebreton, J D (2005) Using demographic invariants to detect overharvested bird populations from incomplete data. *Conservation Biology 19*: 826–835.

Parker, G; Brickle, P; Crofts, S; Pompert, J; Wolfaardt, A (2013) Research into undetected seabird mortality in a demersal trawl fishery. SBWG5 Doc 07. Fifth Meeting of the Seabird Bycatch Working Group, La Rochelle, France, 1–3 May 2013. Agreement on the Conservation of Albatrosses and Petrels, Hobart.

Pierre, J P; Richard, Y; Abraham, E R (2015) Assessment of cryptic seabird mortality due to trawl warps and longlines. Final report

prepared for the Department of Conservation, Conservation Services Programme project INT2013-05. 51 p.

Pinkerton, M H; Smith, A N H; Raymond, B; Hosie, G W; Sharp, B R; Leathwick, J R; Bradford-Grieve, J M (2010) Surface distribution of *Oithona similis* in the Southern Ocean: predictions using Boosted Regression Trees. *Deep Sea Research I 57 (4)*: 469–485.

Pitcher, C R; Ellis, N; Jennings, S; Hiddink, J G; Mazor, T; Kaiser, M J, et al. (2016) Estimating the sustainability of towed fishing-gear impacts on seabed habitats: a simple quantitative risk assessment method applicable to data-limited fisheries. *Methods in Ecology and Evolution 8 (4)*: 472–480. doi:10.1111/2041-210X.12705.

Richard, Y; Abraham, E R (2013a) Application of Potential Biological Removal methods to seabird populations. *New Zealand Aquatic Environment and Biodiversity Report No. 108*. 30 p.

Richard, Y; Abraham, E R (2013b) Risk of commercial fisheries to New Zealand seabird populations. *New Zealand Aquatic Environment and Biodiversity Report No. 109*. 58 p.

Richard, Y; Abraham, E R (2015) Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006–07 to 2012–13. *New Zealand Aquatic Environment and Biodiversity Report No. 162*. 89 p.

Richard, Y; Abraham, E R; Berkenbusch, K (2017) Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006–07 to 2014–15. *New Zealand Aquatic Environment and Biodiversity Report No. 191*. 112 p. plus Supplement (148 p).

Richard, Y; Abraham, E R; Filippi, D (2011) Assessment of the risk to seabird populations from New Zealand commercial fisheries. Final Research Report for Ministry of Fisheries projects IPA2009/19 and IPA2009/20. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Roux, M-J; Edwards, C T T; Dunn, A; Clark, M R; Doonan, I (2015) A sustainability assessment framework for fish by-catch in New Zealand deepwater fisheries. ICES Annual Science Conference. 21–25 September 2015, Copenhagen, Denmark. ASC Extended Abstracts. Retrieved from http://www.ices.dk/sites/pub/ASCExtendedAbstracts/Shared Documents/A - Advancement of stock assessment methods for sustainable fisheries/A1915.pdf.

Roux, M J; Dunn, A; Baird, S; Clark, M R; Bowden, D A; Fu, D (2016) MPI Project BEN2014-01: Risk Assessment for Benthic Habitats, Biodiversity and Production. Presentation to the New Zealand Ministry for Primary Industries Aquatic Environment Working Group (AEWG), 24 August 2016, Wellington.

Sharp, B R; Parker, S J; Smith, N (2009) An impact assessment framework for bottom fishing methods in the CCAMLR area. *CCAMLR Science 16*: 195–210.

Sharp, B R; Waugh, S M; Walker, N A (2011) A risk assessment framework for incidental seabird mortality associated with New Zealand fishing in the New Zealand EEZ. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Sibanda, N; Edwards, C T T; Roux, M-J; Dunn, A (2016) Estimating gear efficiency for non-target species. Final Research Report for the Ministry of Primary Industries. 29 p.

Wade, P R (1998) Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Marine Mammal Science 14*: 1–37.

Watkins, B P; Peterson, S L; Ryan, P G (2008) Interactions between seabirds and deep-water hake trawl gear: an assessment of impacts in South African waters. *Animal Conservation 11*: 247–254.

Waugh, S; Filippi, D; Abraham, E (2009) Ecological risk assessment for seabirds in New Zealand fisheries. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Waugh, S M l; Filippi, D P; Sharp, B R; Weimerskirch, H (2012) Ecological risk assessment of global fisheries for southern bluefin tuna. CCSBT ERS/1203/09. Paper submitted to the Ninth Meeting of the Ecologically Related Species Working Group. 27–30 March 2012, Tokyo, Japan.

Waugh, S; Filippi, D; Walker, N A; Kirby, D S (2008) Updated preliminary results of an ecological risk assessment for seabirds and marine mammals with risk of fisheries interactions. Report to the Scientific Committee for CCAMLR, WG-FSA-08/51.

Zhou, S; Griffiths, S P; Miller, M (2009) Sustainability assessment for fishing effects (SAFE) on highly diverse and data-limited fish bycatch in a tropical prawn trawl fishery. *Marine and Freshwater Research 60 (6)*: 563–570.

Zhou, S; Smith, A D; Fuller, M (2011) Quantitative ecological risk assessment for fishing effects on diverse data-poor non-target species in a multi-sector and multi-gear fishery. *Fisheries Research 112 (3)*: 168–178.

THEME 1: PROTECTED SPECIES

4 NEW ZEALAND SEA LION (*PHOCARCTOS HOOKERI*)

Status of chapter	This chapter has been partially updated for AEBAR 2018. Updated figures and tables have been identified in their captions.
Scope of chapter	This chapter briefly describes: the biology of New Zealand sea lion/rāpoka (*Phocarctos hookeri*) the nature and extent of potential interactions with fisheries; management of fisheries interactions; means of estimating fisheries impacts and population-level risk; and remaining sources of uncertainty, to guide future work.
Area	Southern parts of the New Zealand EEZ and Territorial Sea, near the Subantarctic Islands and around Stewart Island and South Island.
Focal localities	Areas with potential for significant fisheries interactions include the Auckland Islands Shelf, the Campbell Plateau, Stewart Island, and the southern and south-eastern coasts of the South Island.
Key issues	Improved understanding of the effects of fishing in the context of non-fishery threats and environmental variability; improved understanding of spatio-temporal distributions affecting interaction rates with fishing effort; improved estimation of cryptic mortality in trawls employing SLEDs; improved understanding of the risk factors and population consequences of *Klebsiella pneumoniae*-infection and death of pups; improved understanding of the causes and population consequences of nutritional stress for the Auckland Islands and Campbell Island colonies; improved understanding of factors affecting newly established South Island and Stewart Island breeding populations.
Emerging issues	Improved means of estimating incidental captures and risk in poorly observed inshore fisheries potentially interacting with South Island and Stewart Island colonies. Management of public interactions with recovering South Island and Stewart Island populations.
FNZ research (current)	PRO2017-10: *Analysis of New Zealand sea lion tracking data to estimate overlap with fisheries;* PRO2017-08C *Factors affecting New Zealand sea lion pup survival*; SEA2015-10: *Sea lion prey survey*; SEA2014-12: *New Zealand sea lion stable isotope analysis*; PMM2018-11: *Update Auckland Islands NZ sea lion demographic population model;* PMM2018-09: *Desktop estimation of New Zealand sea lion cryptic mortality in trawls using SLEDs*; PMM2018-05B *Estimate spatial distributions for South Island NZSL to assess potential fisheries overlap and risk (including aquaculture). ZBD2018-05: Environmental variability, regime shifts, and ecosystem function in the Sub-Antarctic*.
NZ government research (current)	DOC Marine Conservation Services Programme (CSP): INT2014-01 *Observing commercial fisheries*; INT2015-02 *Identification of marine mammals, turtles and protected fish captured in New Zealand fisheries*; MIT2014-01 *Protected species bycatch newsletter.* POP2015-05 *New Zealand Sea Lion: Auckland Islands population project.* NIWA Research: SA123098 *Multispecies modelling to evaluate the potential drivers of decline in New Zealand sea lions*; TMMA103 *Conservation of New Zealand's threatened iconic marine megafauna.* Joint funding with Deepwater Group: *New Zealand sea lion population project (Campbell Islands).*
Related chapters/issues	Chapter 5: New Zealand fur seals.

4.1 CONTEXT

Management of fisheries impacts on New Zealand sea lions is legislated under the Marine Mammals Protection Act (MMPA) 1978 and the Fisheries Act (FA) 1996.

The Minister of Conservation gazetted the New Zealand sea lion as a threatened species in 1997. All marine mammal species are designated as protected species under s.2 (1) of the FA. In 2005, the Minister of Conservation approved the Conservation General Policy, which specifies in Policy 4.4 (f) that '*Protected marine species should be managed for their*

long-term viability and recovery throughout their natural range.' DOC's Regional Conservation Management Strategies outline specific policies and objectives for protected marine species at a regional level. New Zealand's Subantarctic islands, including Auckland and Campbell islands, were inscribed as a World Heritage area in 1998.

Fisheries New Zealand manages fishing-related mortality of New Zealand sea lions under s.15 (2) of the FA. Under that section, the Minister of Fisheries '*may take such measures as he or she considers are necessary to avoid, remedy, or mitigate the effect of fishing-related mortality on any protected species, and such measures may include setting a limit on fishing-related mortality.*'

The relevant National Fisheries Plan for the management of incidental captures of New Zealand sea lions is the National Fisheries Plan for Deepwater and Middle-depth Fisheries Part 1A (the National Deepwater Plan). Under the National Deepwater Plan, the objective most relevant for management of New Zealand sea lions is Environmental Outcome 8 : *Manage deepwater and middle-depth fisheries to avoid, remedy or mitigate the adverse effects of these fisheries on the long-term viability of endangered, threatened and protected species*.

Specific objectives for the management of incidental captures of New Zealand sea lions are outlined in the fishery-specific chapters of the National Deepwater Plan for the fisheries with which New Zealand sea lions are most likely to interact. These fisheries include sub-Antarctic trawl fisheries for arrow squid, southern blue whiting, and scampi. The chapters in the National Deepwater Plan (Part 1B) for arrow squid, southern blue whiting and scampi are being reviewed and updated in 2019.

The New Zealand sea lion population is monitored based on pup counts at the main breeding colonies, the largest of which are on the Auckland Islands. The number of sea lion pups born at the Auckland Islands declined nearly 50% between 1998 and 2009, and appears to have stabilised thereafter. In 2014, following the third-lowest pup count on record, the Minister of Conservation and the Minister for Primary Industries requested that DOC and MPI work to develop a New Zealand sea lion/rāpoka Threat Management Plan (TMP). The process to develop the TMP involved a number of workstreams, including: a workshop to understand causes of pup mortality for sea lions at the Auckland Islands; two multi-day workshops, attended by a panel of independent experts, to inform a multi-threat risk assessment (Roberts 2015, Debski & Walker 2016); and inaugural meetings of the New Zealand sea lion/rāpoka Forum and Advisory Groups in early 2017. The TMP was finalised in 2017 (DOC & MPI 2017).

The TMP reflects the demographic population model and multi-threat risk assessment described in Roberts & Doonan (2016), and recognises that no single identified threat in isolation is responsible for the observed population decline, such that population recovery would benefit from mitigation of multiple threats at the four main breeding sites (DOC & MPI 2017). The TMP commits to two objectives:

1) Halt the decline of the New Zealand sea lion population within 5 years.
2) Ensure the New Zealand sea lion population is stable or increasing within 20 years, with the ultimate goal of achieving 'Not Threatened' status.

The TMP proposes a work programme toward achievement of the plan's objectives, to be reviewed every five years. An overview of the TMP and identified workstreams, including research priorities, are reproduced in Figures 4.1 and 4.2.



Figure 4.1: Threat management and population recovery objectives specific to four different New Zealand sea lion breeding populations, from the New Zealand sea lion Threat Management Plan (DOC & MPI 2017).



Figure 4.2: Workstreams identified in the New Zealand sea lion Threat Management Plan (DOC & MPI 2017)

4.2 BIOLOGY

4.2.1 TAXONOMY

The New Zealand sea lion (*Phocarctos hookeri*, Gray 1844) is one of only two species of otariid (eared seals, including fur seals and sea lions) native to New Zealand, the other being the New Zealand fur seal (*Arctocephalus forsteri*, Lesson 1828). The New Zealand sea lion is New Zealand's only endemic pinniped.

4.2.2 DISTRIBUTION

Before human habitation, New Zealand sea lions ranged around the North and South Islands of New Zealand and the Chatham Islands (Rawlence et al. 2016). Pre-European remains of New Zealand sea lions have been identified from at least 47 archaeological sites, ranging from Stewart Island to North Cape, with most occurring in the southern half of the South Island (Smith 1989, 2011, Childerhouse & Gales 1998, Gill 1998). Analysis of Holocene remains indicated that breeding sea lions once occurred around north-west Nelson, and that South Island and Chatham Island subpopulations were genetically distinct from contemporary New Zealand sea lions. These subpopulations became extinct shortly after the arrival of Polynesian settlers (Collins et al. 2014a, 2014b, Rawlence et al. 2016). Subsistence hunting on the South Island and subsequent commercial harvest from outlying islands of New Zealand sea lions for skins and oil resulted in population decline and contraction of the species' range (Gales 1995, Childerhouse & Gales 1998, Nagaoka 2001, 2006). Currently, most New Zealand sea lions are found in the New Zealand Subantarctic, with individuals ranging to the New Zealand South Island and Macquarie Island. New Zealand sea lion breeding colonies[1] are highly localised, with most pups being born at two main breeding areas, the Auckland Islands and Campbell Island (Wilkinson et al. 2003, Chilvers 2008). At the Auckland Islands, there are three breeding colonies: Enderby Island (at Sandy Bay); Dundas Island; and Figure of Eight Island. On Campbell Island there is one breeding colony at Davis Point, another colony at Paradise Point, plus a small number of non-colonial breeders (Wilkinson et al. 2003, Chilvers 2008, Maloney et al. 2009, Maloney et al. 2012). Breeding on the Auckland Islands represents 68–79% of the pup production for the species, with the remaining 21–32% occurring on Campbell Island (based on concurrent pup counts in 2008, 2010, 2015 and 2018; see Section 0). Numbers of breeding sea lions at the new Stewart Island colony and at haul-out sites on South Island are comparatively low, but may be expected to increase steadily if these recolonization events are successful.

Intermittent sea lion pup sightings have been reported at Port Pegasus, Stewart Island since the 1990's. In 2011 a pup survey and tagging programme was initiated, with 16 pups tagged. Breeding success at the Stewart Island location has increased steadily since that time, with 55 pups tagged in 2018 in the standard survey area, plus another 7 pups tagged outside the survey area (Boren 2018). After 5 consecutive years with more than 35 pups being counted, in 2018 the Stewart Island population was officially recognised as the third New Zealand sea lion breeding colony.

Successful sea lion breeding has also been observed on the Otago Coast, South Island, beginning with a single female that arrived in 1992 and gave birth in 1993 (McConkey et al. 2002). Pup production at this location increased slowly, to 7 pups in 2013, followed by a more rapid increase, to 18 pups in 2018[2] (Figure 4.3).

On land, New Zealand sea lions are able to travel long distances and ascend hills. They are found in a variety of habitats including grass fields, bedrock, and dense bush and forest; breeding colonies are primarily on sandy beaches (Gales 1995, Augé et al. 2012). In early summer, breeding sea lions are spatially constrained in the vicinity of colony locations. Following the end of the females' oestrus cycle in late January, adult and sub-adult males disperse throughout the species' range, whereas the dispersal of females (both breeding and non-breeding) is more restricted both during and subsequent to the breeding

[1] DOC (2009) defines colonies as 'haul-out sites where 35 pups or more are born each year for a period of 5 years or more.' Haul-out sites are defined as 'terrestrial sites where New Zealand sea lions occur but where pups are not born, or where fewer than 35 pups are born per year over 5 consecutive years.'

[2] Jim Fyfe, Otago pup numbers 2018, https://www.doc.govt.nz/globalassets/documents/conservation/marine-and-coastal/marine-conservation-services/reports/otago-pup-numbers-2018-jim-fyfe.pdf.

season (Marlow 1975, Robertson et al. 2006, Chilvers & Wilkinson 2008).

4.2.3 FORAGING ECOLOGY

Foraging studies have been conducted on lactating female New Zealand sea lions from Enderby Island (Chilvers et al. 2005b, 2006, 2013, Chilvers & Wilkinson 2009), as well as at the Auckland Islands, Stewart Island, and the Otago Peninsula (see Augé et al. 2011a, 2014, Chilvers et al. 2011). Leung et al. (2012, 2013b, 2014b) investigated foraging by juvenile New Zealand sea lions at Enderby Island, Auckland Islands in contrast with juvenile animals at Otago Peninsula (Leung et al. 2013a), and in mother-yearling pairs at Enderby Island (Leung et al. 2014a). A comprehensive analysis of spatial foraging patters of Auckland Islands se alions is currently in progress under MPI contract PRO2017-10, using all available satellite telemetry data to characterise diving behaviour and estimate overlap with fisheries. This research is scheduled to be completed in early 2019. Similar analyses of satellite-tracked individuals from Campbell Island is in preparation (M.-A. Lea, pers. comm.).

Previous analyses of sea lion foraging indicate that females from Enderby Island forage primarily within the Auckland Islands continental shelf and its northern edge, and that individuals show strong foraging site fidelity both within and across years. Satellite tagging data from lactating females at the Auckland Islands shows that the mean return distance travelled per foraging trip is 423 ± 43 km (n = 26), which is greater than that recorded for any other sea lion species (Chilvers et al. 2005b). While foraging, about half of the time was spent submerged, with a mean dive depth of 130 ± 5 m (max. 597 m) and a mean dive duration of 4 ± 1 minutes (max. 14.5 minutes; Chilvers et al. 2006). Both juvenile (2–5 years old) female and male sea lions foraged to the north of the Auckland Islands, but mean distance travelled per foraging trip was shorter in females (99 ± 12 km, n = 19) compared to males (184 ± 25 km, n = 12), and the mean maximum distance from the colony for males (93 ± 10 km) was about twice that for females (51 ± 5 km; Leung et al. 2012). A study of seven dependent yearling New Zealand sea lions (Leung et al. 2013b) found that dive depth was negatively related with animal mass (lighter sea lions dived to greater depths), but in juvenile (2–5 years old) New Zealand sea lions, diving ability (dive depth, dive duration and bottom time per dive) improved with both mass and age, and five-year-old male New Zealand sea lions had similar dive capability to adult females (Leung et al. 2014b). New Zealand sea lions, like most pinnipeds, may use their whiskers to help them capture prey at depths where light does not penetrate (Marshall 2008, Hankel et al. 2010). Leung et al. (2014a) found no evidence that yearling New Zealand sea lions were developing foraging skills through observational learning of maternal behaviours in a study of seven mother-yearling partnerships at Enderby Island.

A recent review of studies conducted on female New Zealand sea lions suggests a continuum of foraging behaviour between benthic foraging vs meso-pelagic foraging modes (Roberts et al. 2018). An earlier study suggested that individual animals may tend to specialize in one or the other foraging mode (Chilvers & Wilkinson 2009). In that study benthic divers had fairly consistent dive profiles, reaching similar depths (120 m on average) on consecutive dives in relatively shallow water, presumably to feed on benthic prey. Meso-pelagic divers, by contrast, exhibited more varied dive profiles, undertaking both deep (over 200 m) and shallow (less than 50 m) dives over deeper water. Benthic divers tended to forage further from their breeding colonies, making their way to the north-eastern limits of Auckland Islands' shelf, whereas meso-pelagic divers tended to forage along the north-western edge of the shelf over depths of approximately 3000 m (Chilvers & Wilkinson 2009). Meynier et al. (2014), employing fatty acid (FA) analyses of blubber samples, found that FA profiles were different in primarily benthic-diving vs primarily meso-pelagic-diving lactating New Zealand sea lions, suggesting a different utilisation of prey resources such that, while prey species taken were similar across both dive modes, the proportion of particular prey differed between the two modes. In addition, Chilvers (2017a) found that the composition of stable isotopes obtained from both blood serum and whiskers differed between benthic versus mesopelagic foraging sea lions at the Auckland Islands. Further, Meynier et al. (2014) found that the body condition index (BCI: the residual between the measured and predicted body mass from the mass-length regression provided by Childerhouse et al. (2010a) was significantly greater in meso-pelagic divers compared to benthic divers.

The differences in dive profiles have further implications for the animals' estimated aerobic dive limits (ADL; Gales & Mattlin 1997, Chilvers et al. 2006), defined as the maximum amount of time that can be spent underwater without increasing blood lactate concentrations (a byproduct of anaerobic metabolism). If animals exceed their ADL and accumulate lactate, they must surface and go through a recovery period in order to aerobically metabolise the

lactate before they can undertake subsequent dives. Chilvers et al. (2006) estimated that lactating female New Zealand sea lions at the Auckland Islands exceed their ADL on 69% of all dives, a much higher proportion than most other otariids (which exceed their ADL for only 4–10% of dives; Chilvers et al. 2006). Auckland Islands sea lions that exhibit benthic diving profiles are estimated to exceed their ADL on 82% of dives, compared with 51% for meso-pelagic divers (Chilvers 2008).

Chilvers et al. (2006) and Chilvers & Wilkinson (2009) suggested that the long, deep diving behaviour, the propensity to exceed their estimated ADL, and differences in physical condition and age at first reproduction from animals at Otago together indicate that females from the Auckland Islands may be foraging at or near their physiological limits. However, Bowen (2012) suggested a lack of relationship between surface time and anaerobic diving would seem to indicate that ADL has been underestimated. Further, given a number of studies of diving behaviour were conducted during early lactation when the demands of offspring are less than they would be later in lactation, Bowen (2012) considered it unlikely that females are operating at or near a physiological limit.

Adult females at Otago are generally heavier for a given age, breed earlier, undertake shorter foraging trips, and have shallower dive profiles compared with females from the Auckland Islands (Table 4.1). These observed differences may reflect differences in habitat (including prey availability) between the Auckland Islands and the Otago Peninsula, or a founder effect, or a combination of these or other factors. Similarly, Leung et al. (2013a) compared foraging characteristics in juvenile (2–3 years old) female New Zealand sea lions at Enderby Island and Otago Peninsula. Overall, females at Otago were heavier (3 year old mean 96 kg) than females at Enderby (3 year old mean 72 kg), and exhibited shorter mean foraging trip distance (19 km at Otago, 103 km at Enderby), shallower mean dive depth (15 m at Otago, 69 m at Enderby) and shorter mean dive duration (1.8 min at Otago, 3.2 min at Enderby). Leung et al. (2013a) concluded that the Auckland Islands are a less optimal habitat compared to Otago. New evidence from satellite tracked individuals at the Campbell Islands (M-A Lea, pers. comm.) and from analysis of sea lion prey including a dedicated ocean survey (Roberts et al. 2018) suggests that sea lions at the Subantarctic Islands may suffer from periods of low prey availability and may be forced to forage at the limits of their physiological capabilities. This would make these populations particularly susceptible to environmental variability affecting availability of preferred prey. Roberts and Doonan (2016) identify nutritional stress as a potentially significant threat to sea lions.

The foraging of lactating females at Port Pegasus, Stewart Island was recently characterised by Chilvers (2017b), describing their foraging characteristics as intermediate between Auckland Islands and Otago Peninsula females with respect to dive depth, dive duration and body mass. Satellite telemetry data indicated that nearly all foraging was within 50 km of the tagging site at Port Pegasus (Roberts 2017a).

Table 4.1: Comparison of selected characteristics between adult female New Zealand sea lions from the Auckland Islands and those from the Otago Peninsula (Augé et al. 2011a, 2011b, Chilvers et al. 2006, Roberts & Doonan 2016). Data are means ± s.e. (where available).

Characteristic	Auckland Islands	Otago
Reproduction at age 4	19% of females (95 % CI = 16 –23 %)	> 85% of females
Average mass at 8–13 years of age	112 kg	152 kg
Foraging distance from shore	102.0 ± 7.7 km (max = 175 km)	4.7 ± 1.6 km (max = 25 km)
Time spent foraging at sea	66.2 ± 4.2 hrs	11.8 ± 1.5 hrs
Dive depth	129.4 ± 5.3 m (max = 597 m)	20.2 ± 24.5 m (max = 389 m)
Dives estimated to exceed ADL	68.7 ± 4.4%	7.1 ± 8.1%

New Zealand sea lions are generalist predators with a varied diet that includes marine mammal prey (New Zealand fur seal *Arctocephalus forsteri*), seabirds (yellow-eyed penguin *Megadyptes antipodes*, blue penguin *Eudyptula minor*, Southern rockhopper penguin *Eudyptes chrysocome*, southern royal albatross *Diomedea epomophora*), elasmobranchs (rough skate *Raja nasuta*), teleost fish (e.g., opalfish *Hemerocoetes* spp., hoki *Macruronus novaezelandiae*, red cod *Pseudophycis bachus*, jack mackerel *Trachurus* spp., barracouta *Thyrsites atun,* southen blue whiting *Micromesistius australis*), cephalopods (e.g., octopus *Enteroctopus zelandicus* and *Macroctopus maorum*, squid *Nototodarus sloanii*), crustaceans (e.g., lobster krill *Munida gregaria*, and other invertebrates, e.g., salps) (Cawthorn et al. 1985, Moore & Moffat 1992, Bradshaw et al. 1998, Childerhouse et al. 2001, Lalas et al. 2007, Moore et al. 2008, Meynier et al. 2009, Augé et al. 2012, Lalas et al. 2014, Lalas & Webster 2014, Morrison et al. 2017). The three main methods used to assess New Zealand sea lion diets involve analyses of stomach contents, scats and regurgitate, and the fatty acid composition of blubber (Meynier et al. 2008). Stomach contents of bycaught animals tend to be biased towards the target species of the fishery concerned (e.g., squid in the Auckland Islands squid fishery), whereas scats and regurgitates are biased towards less digestible prey (Meynier et al. 2008). Stomach, scat and regurgitate approaches tend to reflect only recent prey (Meynier et al. 2008). By contrast, analysis of the fatty acid composition of blubber provides a longer-term perspective on diets ranging from weeks to months (although individual prey species are not identifiable). Fatty acid analysis suggests that the diet of female New Zealand sea lions at the Auckland Islands tends to include proportionally more arrow squid and hoki and proportionally fewer red cod than for male New Zealand sea lions, while lactating and non-lactating females do not differ in their diet (Meynier et al. 2008, Meynier 2010). Within a sample of lactating female New Zealand sea lions, Meynier et al. (2014) used fatty acid analyses to show that the diet of benthic-diving and meso-pelagic-diving animals consisted of similar prey, though different mass contributions for each prey species.

Previous assessments have identified considerable spatial (comparing colonies) and temporal (inter-annual and seasonal) variation in the diet composition of New Zealand sea lions. For instance, jack mackerel and baracoutta were identified as the main prey of the Otago Peninsula population (Augé et al. 2012), though were less prevalent in winter and spring when inshore species dominated diet composition (Lalas 1997) and were infrequent prey of the Auckland Islands population (Childerhouse et al. 2001, Stewart-Sinclair 2013). A long-term diet assessment of the Sandy Bay colony at the Auckland Islands (1994–95 to 2012–13) identified a decrease in the occurrence of large-sized prey (e.g., *Enteroctopus zealandicus*) and an increasing trend in small-sized prey (e.g., opalfish, rattails and *Octopus* spp.) (Childerhouse et al. 2001, Stewart-Sinclair 2013).

Teeth from individual sea lions at the Auckland Islands that were archived at Massey University and Te Papa were used to estimate trophic histories over an extended historical period. Graham (in prep) analysed 292 samples from the annual growth bands found in 17 sea lion teeth. This dataset represents the nitrogen and carbon isotopic histories of 13 female sea lions dating from 1935 to 2005, along with the histories of 4 males that were analysed initially to refine methods and techniques. Nitrogen isotope (δ15N) data indicate an animal's trophic ecology and changes in their foraging strategies. It was found that the male sea lions consistently forage at a higher trophic level than the females. The δ15N values of the remaining 13 females shows 3 distinct features. At a broad scale, there is considerable variation between individuals, suggesting variable foraging strategies. A maternal or lactation signal was observed in almost all teeth samples. This signal occurs because as the pup consumes the mother's milk its isotope value will be one trophic level higher than its mother. In general, the lactation signal declined for most of the individuals in the first year, and by year two it was only present in two individuals born in 1943 and 1994. An increase in trophic level occurs after age five, which coincides with the age at first breeding, but again there is inter-individual variation. The δ13C dataset for female New Zealand sea lions shows an overall decreasing temporal trend, with notable decreases pre-1960 and post-1990. Changes in primary productivity affect the δ13C values at the base of the food web and this signal has been shown to propagate up the food web. Overall, in periods of higher productivity the δ13C values increase (Laws et al. 1995, Schell et al. 1998, Graham et al. 2010). This suggests that during the 1940–60s and late 1990–early 2000s there was either a) a decrease in productivity around the Auckland Islands where the female sea lions forage (i.e., shift in ocean conditions) or b) the females shifted their main foraging strategy (e.g., benthic vs. mesopelagic related to available prey). More samples will be required, especially in the earlier time period to resolve the timing of these isotopic changes as it relates to estimated changes in fish stocks. In

addition, examining other species in the region during this time period will enable larger-scale assessments and how they might relate to shifts in ocean conditions.

4.2.4 REPRODUCTIVE BIOLOGY

New Zealand sea lions exhibit marked sexual dimorphism, with adult males being larger and darker in colour than adult females (Walker & Ling 1981, Cawthorn et al. 1985). Cawthorn et al. (1985) and Dickie (1999) estimated the maximum age of males and females to be 21 and 23 years, respectively; Childerhouse et al. (2010b) reported a maximum estimated age for females of 28 years. Females can become sexually mature as early as age two and may give birth the following year. However, at the Auckland Islands most females do not breed until they are six years old (Childerhouse et al. 2010b); at Otago Peninsula most females breed by age four (Roberts & Doonan 2016). Males generally reach sexual maturity at four years of age, but because of their polygynous colonial breeding strategy (i.e., males actively defend territories and mate with multiple females within a harem) they are only able to successfully breed at 7–9 years old, once they have attained sufficient physical size to compete successfully with other males (Marlow 1975, Cawthorn et al. 1985). At the Auckland Islands, the reproductive rate in females increases rapidly between the ages of 3 and 7, reaching a plateau until the age of approximately 15 and declining rapidly thereafter, with the maximum recorded age at reproduction being 26 years (Breen et al. 2016, Childerhouse et al. 2010a, Chilvers et al. 2010). Chilvers et al. (2010) estimated from tagged sea lions that the median lifetime reproductive output of a female New Zealand sea lion at the Auckland Islands was 4.4 pups, and 27% of all females that survive to age 3 never breed. Analysis of tag-resighting data from female New Zealand sea lions on Enderby Island indicates the average probability of breeding is approximately 0.30–0.35 for prime-age females that did not breed in the previous year (ranges reflect variation relating to the definition of breeders) and 0.65–0.68 for prime-age females that did breed in the previous year (MacKenzie 2011).

New Zealand sea lions are philopatric (i.e., they return to breed at the same location where they were born, although more so for females than males). Breeding is highly synchronised and starts in late November when adult males establish territories (Robertson et al. 2006, Chilvers & Wilkinson 2008). Pregnant and non-pregnant females appear at the breeding colonies in December and early January, with pregnant females giving birth to a single pup in late December before entering oestrus 7–10 days later and mating again (Marlow 1975). Twin births and the fostering of pups in New Zealand sea lions are rare (Childerhouse & Gales 2001). Shortly after the breeding season ends in mid-January, the harems break up with the males dispersing offshore and females often moving away from the rookeries with their pups (Marlow 1975, Cawthorn et al. 1985).

Pups' birth weight is 8–12 kg and is highly variable between years; parental care is restricted to females (Walker & Ling 1981, Cawthorn et al. 1985, Chilvers et al. 2006). Females remain ashore for about 10 days after giving birth before alternating between foraging trips lasting approximately two days out at sea and returning for about one day to suckle their pups (Gales & Mattlin 1997, Chilvers et al. 2005b). New Zealand sea lion pup growth rates at the Auckland Islands are lower than those reported for other sea lion species and may be linked to a relatively low concentration of lipids in the females' milk during early lactation (Chilvers 2008, Riet-Sapriza et al. 2012). Riet-Sapriza et al. (2012) also found that there was a temporal (year and month) effect on milk quality, reflecting individual sea lion characteristics and environmental factors, and that maternal body condition was positively correlated with milk lipid concentration, energy content and milk protein concentration: lactating females in good condition produced more energy-rich milk than did relatively lean females. Pups are weaned after about 10–12 months (Marlow 1975, Gales & Mattlin 1997).

4.2.5 POPULATION BIOLOGY

For New Zealand sea lions, the overall size of the population is indexed using estimates of the number of pups that are born each year (Chilvers et al. 2007). Since 1995, the Department of Conservation (DOC) has conducted mark-recapture counts at each of the main breeding colonies at the Auckland Islands to estimate annual pup production (i.e., the total number of pups born each year, including dead and live animals; Robertson & Chilvers 2011). Pup censuses have been less frequent for other colonies, including the large population at Campbell Island (Maloney et al. 2012). For the Auckland Islands population, the data show a decline in pup production from a peak of 3,021 in 1997–98 to a low of 1,501 ± 16 pups in 2008–09 (Chilvers & Wilkinson 2011, Robertson & Chilvers 2011; Figure 4.3 and Table 4.2), with the largest single-year decline (31%) occurring between the 2008 and 2009 counts.

Since 2009 the breeding population at the Auckland Islands appears to have stabilised and may be recovering, having shown a year-on-year increase in six of nine years, but future trends are uncertain. The most recent estimate of pup production for the Auckland Islands population was 1,792 pups in 2018 of which 332 were counted at Sandy Bay and 1,397 were counted at Dundas Island, using the mark-recapture method (Table 4.2 and Figure 4.3). Using demographic models to predict future population trends, the future trajectory is highly dependent on what period of time is used to estimate future demographic rates in the forward projections. The population models used to inform the TMP (Roberts & Doonan 2016) used demographic rates from the past 20 years, which includes the period of steepest population decline; these models suggest that the total Auckland Island population will continue to decline (see Section 4.4.3.4, Roberts & Doonan 2016). However, projections based on past demographic rates from a 30-year or 10-year window would be expected to produce different trajectories. Because it is not possible to anticipate what environmental or demographic conditions are likely to prevail in the future, uncertainty of this nature is best addressed with model sensitivities.

Total New Zealand sea lion population size (including pups) at the Auckland Islands has been estimated using Bayesian population models (Breen et al. 2003, 2016, Breen & Kim 2006a, 2006b, Roberts & Doonan 2016). Although other abundance estimates are available (e.g. Gales & Fletcher 1999), for the Auckland Islands population, estimates derived from the integrated models are preferred because they take into account a variety of age-specific factors (breeding, survival, maturity, incidental fisheries captures), as well as data on the resighting of tagged animals and pup production estimates (Table 4.3).

For the Campbell Island population, the latest estimate of pup production was 734 pups in 2018 (Boren 2018). Estimates of pup production at Campbell Island increased sharply in the period from 1990 to 2010 (i.e., including during the period of steepest decline at the Auckland Islands) but there has been some variation in the timing and methodology of these surveys. The later surveys in 2003, 2008, 2010 and 2015 were considered to be of sufficient quality to inform a simple population estimate (Roberts & Doonan 2016) and a comparable methodology was used to estimate pup production in 2018. Early pup mortality (i.e. in the first few months of life) at Campbell Island has been relatively high in all recent census years, including: 1998 (31%), 2003 (36%), 2008 (40%), 2010 (55%), 2015 (58%, the highest recorded at any New Zealand sea lion breeding site) and 2018 (23%) (see Childerhouse et al. 2005, 2015a, Boren 2018, Maloney et al. 2009, 2012, McNally et al. 2001).

For the Otago coast, annual pup production has ranged from 0 to 18 pups since the 1995 breeding season, with 18 recorded in 2018 (Figure 4.3). Sea lions at Otago are of special interest because they highlight the potential for establishing new breeding colonies, i.e. because the Otago coast breeding population originated with a single pregnant female (McConkey et al. 2002). The TMP identifies that the viability of new colony locations on the New Zealand South Island is of particular importance for the restoration of New Zealand sea lions to non-threatened status.

Sea lions have also established at Stewart Island, and pup census estimates have been made since 2011, about 3–4 months after the probable pupping period. Stewart Island pup counts have increased from 16 pups in 2011 to 55 pups in 2018 (Chilvers 2014, Roberts & Doonan 2016; Figure 4.3). From 2018, pup counts have exceeded 35 individuals for five consecutive years, formally qualifying Stewart Island as a new breeding colony (DOC & MPI 2017).

Established anthropogenic sources of mortality in New Zealand sea lion include: historic subsistence hunting and commercial harvest (Gales 1995, Childerhouse & Gales 1998); pup entrapment in rabbit burrows prior to rabbit eradication from Enderby Island in 1993 (Gales & Fletcher 1999); human disturbance, including attacks by dogs, vehicle strikes and deliberate shooting on South Island New Zealand (Gales 1995); and incidental captures in fisheries (see Section 4.4).

Table 4.2: Pup census estimates for all known breeding populations of New Zealand sea lions since 1994–95. Years with no census estimates were left blank (i.e., blanks do not necessarily indicate that no pups were born at that location in that year). [Updated for AEBAR 2018].

Pupping season	Annual pup census estimate					
	Auckland Islands			Campbell Island	Otago coast	Stewart Island
	Dundas	Sandy Bay	All			
1995	1 837	467	2 518		0	
1996	2 017	455	2 685		1	
1997	2 260	509	2 975		0	
1998	2 373	477	3 021		2	
1999	2 186	513	2 867		1	
2000	2 163	506	2 856		1	
2001	2 148	562	2 859		3	
2002	1 756	403	2 282		3	
2003	1 891	488	2 516	385	3	
2004	1 869	507	2 515		3	
2005	1 587	441	2 148		4	
2006	1 581	422	2 089		7	
2007	1 693	437	2 224		4	
2008	1 635	448	2 175	583	6	
2009	1 132	301	1 501		5	
2010	1 369	385	1 814	681	6	
2011	1 089	378	1 550		6	16
2012	1 248	361	1 684		6	25
2013	1 491	374	1 940		6	26
2014	1 213	290	1 575		4	32
2015	1 230	286	1 576	696	8	36
2016	1 347	321	1 727		15	31
2017	1 549	349	1 965		16	41
2018	1 397	332	1 792	734	18	55

Table 4.3: Pup production and population estimates of New Zealand sea lions from the Auckland Islands. Pup production data are direct counts or mark-recapture estimates from Chilvers et al. (2007), Robertson and Chilvers (2011), Chilvers (2012a), and Childerhouse et al. (2014, 2015b, 2016), noting that counts of dead pups began later in 2013 and 2014 and this is likely to have led to a negative bias in estimates for these years. Standard errors apply only to the portion of pup production estimated using mark-recapture methods. Mature female population estimates are from the base case model in Roberts & Doonan (2016). Total population size estimates are from the model by Breen et al. (2016). Year refers to the second calendar year of a breeding season (e.g., 2010 refers to the 2009–10 season). [Table continued on next page]. [Updated for AEBAR 2018].

Year	Pup production estimate		Mature female population size		Total population size	
	Mean	Standard error (for mark recapture estimates)*	Median	95% confidence interval	Median	90% confidence interval
1995	2 518	21	3367	3010 - 3784	15 675	14 732–16 757
1996	2 685	22	3617	3274 - 4017	16 226	15 238–17 318
1997	2 975	26	3863	3537 - 4239	16 693	15 656–17 829
1998	3 021	94	4049	3743 - 4402	16 911	15 786–18 128
1999	2 867	33	4158	3876 - 4483	15 091	13 932–16 456
2000	2 856	43	4000	3746 - 4295	15 248	14 078–16 586
2001	2 859	24	3854	3620 - 4127	15 005	13 870–16 282
2002	2 282	34	3707	3488 - 3964	13 890	12 856–15 079
2003	2 518	38	3583	3373 - 3830	14 141	13 107–15 295
2004	2 515	40	3469	3266 - 3705	14 096	13 057–15 278
2005	2 148	34	3356	3159 - 3582	13 369	12 383–14 518
2006	2 089	30	3216	3024 - 3436	13 110	12 150–14 156
2007	2 224	38	3085	2909 - 3282	13 199	12 231–14 215
2008	2 175	44	2804	2646 - 2981	12 733	11 786–13 757

2009	1 501	16	2692	2541 - 2853	12 065	11 160–13 061
2010	1 814	36	2548	2413 - 2694		
2011	1 550[3]	41	2458	2324 - 2602		
2012	1 684	22	2257	2140 - 2382		
2013**	1 940	50	2191	2076 - 2319		
2014**	1 575	19	2053	1945 - 2173		
2015	1 576		1939	1826 - 2068	***	
2016	1 727		1844	1674 - 2030		
2017	1 965					
2018	1 792					

* Calculated as the sum of standard errors associated with estimates for Sandy Bay and Dundas (estimates for other rookeries from direct count rather than mark-recapture).

** Field season began later in these years and pups that died early in the pupping period were unlikely to have been included in pup production estimates.

*** Roberts & Doonan 2016 estimated 11 755 for the entire species.

In addition to these identified threats, there are other anthropogenic effects that may also affect New Zealand sea lion populations, but for which the actual level of impact is presently unclear. These include: potential trophic overlap between New Zealand sea lions and the various fisheries (Robertson & Chilvers 2011, Bowen 2012, Roberts et al. in press); effects of organic and inorganic pollutants, including polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT) and heavy metals such as mercury and cadmium (Baker 1999, Robertson & Chilvers 2011); and impacts of eco-tourism.

Other known threats include epizootics, particularly *Campylobacter* that killed 1600 pups (53% of pup production) and at least 74 adult females on the Auckland Islands in 1997–98 (Wilkinson et al. 2003, Robertson & Chilvers 2011), and *Klebsiella pneumoniae* that killed 33% and 21% of diagnosed pups at the Auckland Islands in 2001–02 and 2002–03, respectively (Wilkinson et al. 2006) and 55% of pups between 2009 and 2014 (Roe et al. 2014). A highly sticky strain of *K. pneumoniae* was isolated from a number of pups that died in field seasons 2005–06 to 2009–10 (Roe 2011). In this period, disease-related mortalities occurred late in the field season relative to the period 1998–99 to 2004–05 and were still occurring up to the end of sampling (Castinel et al. 2007, Roe 2011). The 1998 epizootic event may have affected the fecundity of the surviving pups, reducing their breeding rate relative to other cohorts (Gilbert & Chilvers 2008), though their pupping rate estimate for this cohort is likely to have been negatively biased by particularly high tag shedding rates for individuals tagged in that year (Roberts et al. 2014a). There are also occurrences of predation by white pointer sharks (Cawthorn et al. 1985, Robertson & Chilvers 2011), starvation of pups if they become separated from their mothers (Walker & Ling 1981, Castinel et al. 2007), drowning in wallows and male aggression towards females and pups (Wilkinson et al. 2000, Chilvers et al. 2005a).

Temporally coincident changes in annual abundance, spatial distribution, and/or reproductive success have been observed in different ecosystem components at Subantarctic latitudes -- including New Zealand sea lions, Antipodean albatrosses, and demersal and pelagic fish communities. These observations suggest that climatic variability at decadal scales (sometimes labeled 'regime shifts') may affect ecosystem productivity in these systems, in turn affecting critical demographic rates for Subantarctic Islands sea lions and resulting decadal-scale population dynamics (see below). FNZ is progressing new research to investigate these phenomena (project ZBD2018-05).

Despite the historic reduction in population size as a result of subsistence hunting and commercial harvest, the New Zealand sea lion population does not display low genetic diversity at microsatellite loci and thus does not appear to have suffered effects of genetic drift and inbreeding depression (Robertson & Chilvers 2011).

4.2.6 RELATING DEMOGRAPHIC RATES TO DRIVERS OF POPULATION CHANGE

Demographic assessments have been conducted to identify the proximate demographic causes of population decline at

[3] Due to extreme weather conditions there was some delay in making the 2010–11 pup count, which may affect comparability with previous years. However, DOC's analysis suggests any such effect is unlikely to be large (Chilvers & Wilkinson 2011).

the Auckland Islands. An assessment using mark-resighting data from the Enderby Island sub-population yielded estimates of average annual survival for prime-age females of 0.90 for females that did not breed and 0.95 for females that did breed (MacKenzie 2011). In another assessment, state space demographic models fitted to pup production estimates, age distribution observations and a long time series of mark-resighting observations were developed using NIWA's demographic modelling software SeaBird to estimate year-varying survival, probability of pupping and age-at-first-pupping (Roberts et al. 2014a). This study concluded that low pupping rates (including occasional years with very low rates), a declining trend in cohort survival to age two and to age five since the early 1990s, and relatively low adult survival (age 6–14) from 1999–2000 to 2010–11 are responsible for declining pup production at Sandy Bay from the late 1990s to 2009. In addition, very low pup survival estimates were obtained between 2005 and 2007, which will have compromised breeder numbers and pup production resulting from births at Sandy Bay, though pup survival is estimated to have increased again since 2009 (Figure 4.4) (Roberts & Doonan 2016).

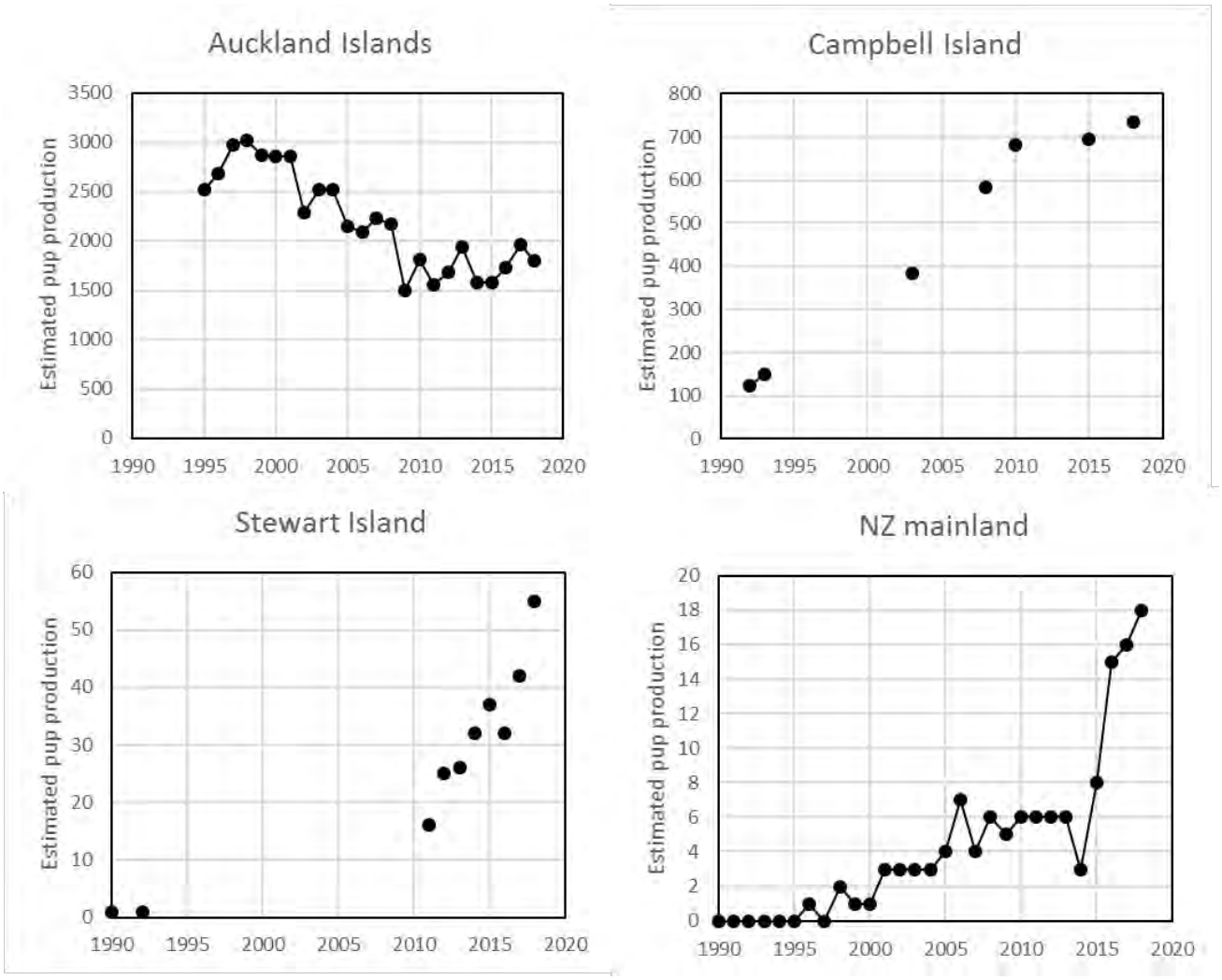


Figure 4.3: Annual sea lion pup count estimates from breeding sites. Note that the scale for each figure is different (DOC & MPI 2017, adapted from Roberts & Doonan 2016). Where count methodology is not always consistent between years (Stewart Island and Campbell Island), point estimates are not joined by lines in the figures. [Updated for AEBAR 2018].

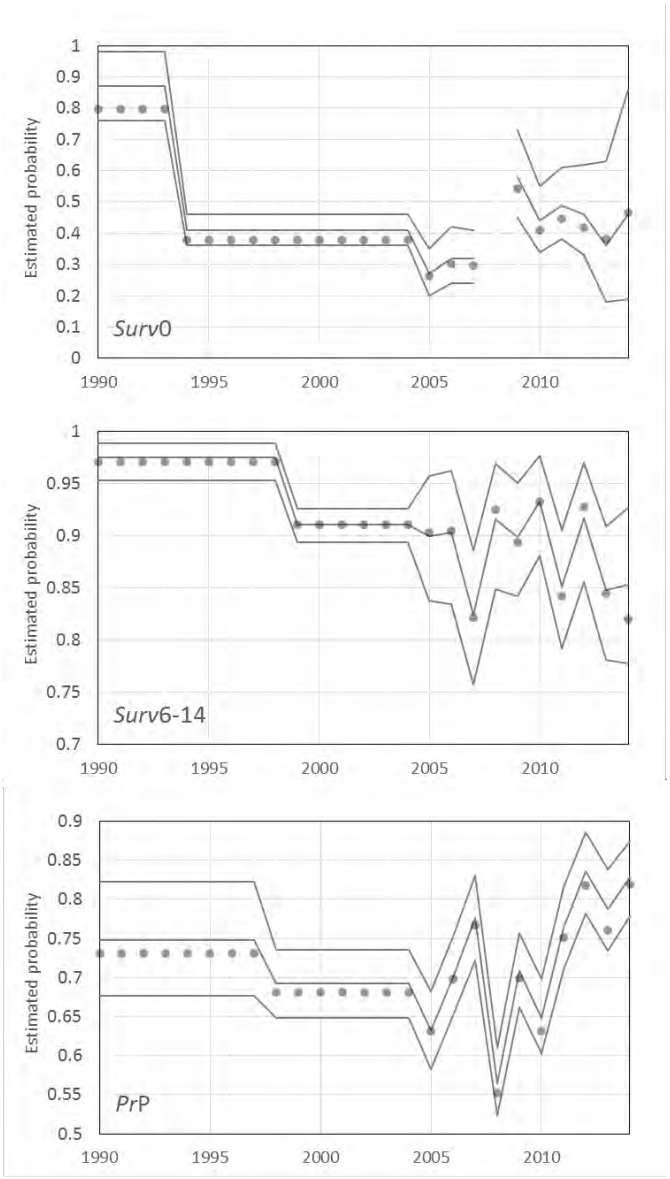


Figure 4.4: Annual estimates of pup survival to age 1 (top), annual survival at age 6–14 (middle) and annual probability of pupping (bottom) of female New Zealand sea lions at the Auckland Islands; points are point estimates; lines are median estimates and 95% c.i. (Roberts & Doonan 2016). [Updated for AEBAR 2018].

A correlative assessment was conducted to identify the causes of varying demographic rates at Sandy Bay, for which hypothetical models developed with expert consultation were used as a framework for testing relationships between demographic rate estimates, biological observations (e.g., diet composition, maternal body condition or pup mass) and candidate drivers of population change (e.g., changes in prey availability, disease-related pup mortality or direct fishery-related mortalities) (Roberts & Doonan 2014).

Climate indices including Inter-decadal Pacific Oscillation (IPO) and sea surface height (SSH) were well correlated with the occurrence of an array of key prey species in scats (Childerhouse et al. 2001, Stewart-Sinclair 2013). A weak, though significant, positive correlation was identified between maternal body condition and pup mass in seasons prior to 2004–05. In this time period, pup mass at three weeks appeared to have been a good predictor of cohort-specific survival to age two, though there was no relationship with cohorts born 2004–05 to 2009–10, for which survival estimates were consistently low despite high pup mass (Figure 4.5). A correlation between cohort survival to age two and the rate of pup mortalities attributed to *K. pneumonia* infection late in the field season (Castinel et al. 2007, Roe 2011) was consistent with disease-

related mortality affecting a decline in pup/yearling survival after 2004–05. Survival at ages 2–5 (juveniles) or ages 6–14 (adults) were not correlated with the estimated level of fishery interactions in the Auckland Islands southern arrow squid (SQU 6T) trawl fishery (Thompson et al. 2011). However, from 1998–99 to 2003–04 survival at ages 6–14 was negatively correlated with the survival of pups born in the previous year, suggesting that the high energetic costs of lactation may compromise maternal survival (Roberts & Doonan 2014).

In most cases observations were available only for short time periods and longer series would be required to identify a causative relationship. However, broad changes in diet composition (e.g., an increased prevalence of small-sized prey species), reduced maternal body condition and depressed pupping rates, are all consistent with a sustained period of nutritional stress negatively affecting the productivity of New Zealand sea lions at the Auckland Islands. In addition, disease-related mortality of pups since 2005–06 (Roe 2011) has caused a decline in pup/yearling survival, which may further compromise breeder numbers at the Auckland Islands in the immediate future. It is likely that these effects are not independent, as nutritional stress can be expected to predispose the population to higher rates of disease mortality.

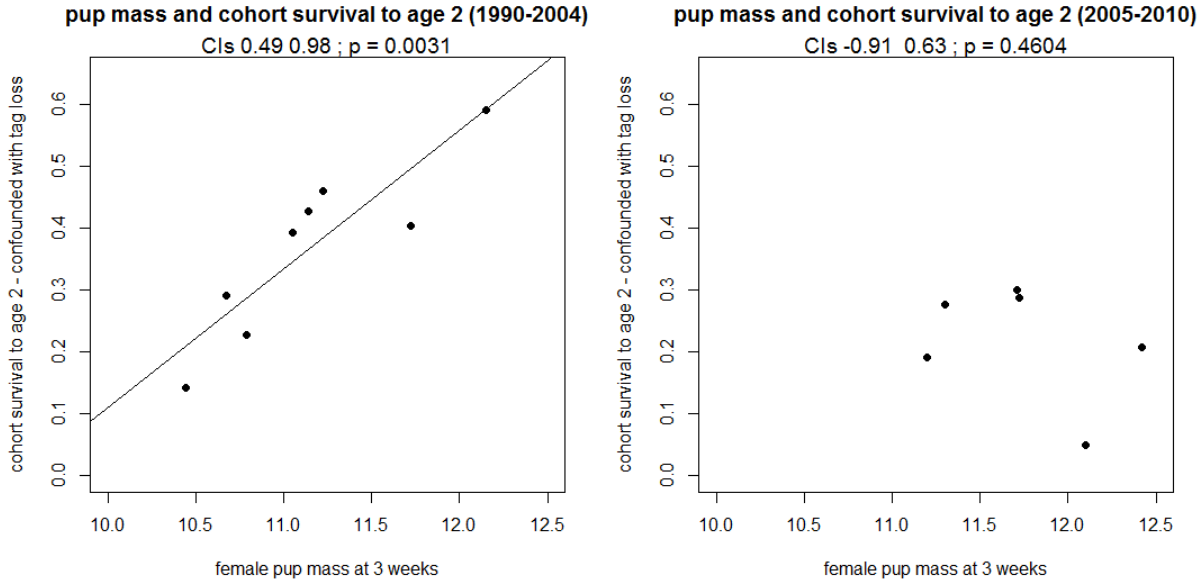


Figure 4.5: Pup mass of females and demographic modelling estimate of cohort survival to age 2; survival estimates confounded with tag loss rate; regression line shown for correlations significant at the 5% level.

4.2.7 CONSERVATION BIOLOGY AND THREAT CLASSIFICATION

Threat classification is an established approach for identifying species at risk of extinction (IUCN 2010). The risk of extinction for New Zealand sea lions has been assessed under two threat classification systems, the International Union for the Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2010) and the New Zealand Threat Classification System (Townsend et al. 2008).

In 2015, the IUCN updated the Red List status of New Zealand sea lions, listing them as Endangered, A4bd on the basis on a projected decline in breeders over three generations (calculated to be 32 years) exceeding a 50% reduction (estimated to be 72%), assuming a linear extrapolation of the observed rate of decline in pup production at the Auckland Islands between 1997-98 and 2013-14 (Chilvers 2015).

In 2013, DOC updated the New Zealand Threat Classification status of all New Zealand marine mammals (Baker et al. 2010). In the revised list, New Zealand sea lions had their threat classification increased from At Risk, Range

Restricted[4] to Nationally Critical under criterion C[5] with a Range Restricted qualifier based on the recent rate of decline at the Auckland Islands (Baker et al. 2016). The New Zealand Threat Classification status for New Zealand sea lions will be updated in early 2019.

4.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Reviews of fisheries interactions among pinnipeds globally can be found in Read et al. (2006), Woodley & Lavigne (1991), Katsanevakis (2008) and Moore et al. (2009). Because New Zealand sea lions are endemic to New Zealand, the global understanding of fisheries interactions for this species is outlined under state of knowledge in New Zealand.

4.4 STATE OF KNOWLEDGE IN NEW ZEALAND

New Zealand sea lions interact with some trawl fisheries, sometimes resulting in incidental capture and death of the sea lion in the net. Observed trawl fishery interactions are confined to Subantarctic waters (Figure 4.6); particularly the two trawl fisheries around the Auckland Islands – the arrow squid fishery (SQU 6T), and the scampi fishery (SCI 6A), Male sea lions are caught in the southern blue whiting (*Micromesistius australis*) fishery near Campbell Island (SBW 6I) and occasional captures occur in the Stewart-Snares Shelf fisheries targeting mainly arrow squid (SQU 1T; Thompson & Abraham 2010, Thompson et al. 2011, 2013).[6] New Zealand sea lions forage to depths of up to 600 m and overlap with trawling at up to 500 m depth for arrow squid, 250–600 m depth for spawning southern blue whiting, and 350–550 m depth for scampi (Tuck 2009, Ministry of Fisheries 2011).

There is seasonal variation in the overlap between New Zealand sea lions and the target species fisheries (Table 4.4) Breeding male sea lions in the Auckland Island area are ashore between November and January with occasional trips to sea, then migrate away from the area (Robertson et al. 2006). Breeding females are in the Auckland Island area year-round, coming ashore for up to 10 days to give birth during December and January and then alternately foraging at sea (~2 days) and suckling their pup ashore (~1.5 days; Chilvers et al. 2005b). The SQU 6T fishery currently operates between December and June, peaking between February and May, whereas the SQU 1T fishery operates between December and May, peaking between January and April, before the squid spawn. The SBW 6I fishery operates in August and September, peaking in the latter month, when the fish aggregate to spawn. The SCI 6A fishery may operate at any time of the year. Research is currently underway (project PRO2017-10) applying the SEFRA method (see AEBAR Chapter 3) to estimate encounters, captures, and deaths arising from overlap between commercial fisheries and sea lion distributions.

4.4.1 QUANTIFYING FISHERIES CAPTURES VS INTERACTIONS

Incidental captures of New Zealand sea lions are recorded by fisheries observers. From 2007-2017, capture rates on the observed portion of the fishing fleet were used in models to estimate total captures across the entire fishing fleet in each fishing year (Smith and Baird 2007b, Thompson and Abraham 2010, Abraham and Thompson 2011, Abraham & Berkenbusch 2017). Estimates for the SQU 6T and Campbell Island fisheries were generated using Bayesian models, whereas those for Auckland Islands scampi fisheries, other Auckland Islands trawl fisheries, and the Stewart-Snares Shelf fisheries were produced using ratio estimates (see Tables 4.6–4.8, and detailed information in Thompson et al. 2013, Abraham & Berkenbusch 2017).

In those fisheries in which the majority of historical captures had been observed, observer coverage levels have increased substantially in recent years. (Tables 4.5-4.8). For example, in the SQU 6T fishery, observer coverage ranged 28–45% from 2002–03 to 2011–12, but increased to 85–92% in the four years since the 2012-13 fishing year (Table 4.5). Similarly, the Campbell Island southern blue whiting fishery has had 100% observer coverage since 2012-13

[4] A taxon is listed as 'Range Restricted' if it is confined to specific substrates, habitats or geographic areas of less than 1000 km^2 (100 000 ha); this is assessed by taking into account the area of occupied habitat of all sub-populations (Townsend et al. 2008).

[5] A taxon is listed as 'Nationally Critical' under criterion C if the population (irrespective of size or number of sub-populations) has a very high (rate of) ongoing or predicted decline; greater than 70% over 10 years or three generations, whichever is longer (Townsend et al. 2008).

[6] See the Report from the Fisheries Assessment Plenary, May 2011 (Ministry of Fisheries 2011) for further information regarding the biology and stock assessments for these species.

(Table 4.8Table 4.5). For fisheries with observer coverage approaching 100%, statistical captures estimation becomes largely unnecessary. However, early in the same period during which observer coverage was increasing (i.e. 2001-02 to 2007-08), the SQU 6T fishery transitioned to widespread adoption of Sea Lion Exclusion Devices (SLEDs), a mitigation device designed to allow sea lions entering the trawl net to exit via the SLED and survive (see below). Observed capture rates on vessels employing SLEDs decreased, but there was uncertainty about the number and fate of sea lions exiting the trawl net via the SLED. To reflect this uncertainty, FNZ sought to estimate the number sea lion / trawl *interactions*, independent of the number of observable *captures*.

Table 4.4: Monthly distribution of New Zealand sea lion activity and the main trawl fisheries with observed reports of incidental captures.

New Zealand sea lions	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Breeding males	Dispersed at sea or at haul outs		At breeding colony			Dispersed at sea or at haul outs						
Breeding females	At sea			At breeding colony		At breeding colony and at-sea foraging and suckling						
New pups				At breeding colony								
Non-breeders	Dispersed at sea, at haulouts, or at breeding colony periphery											
Major fisheries	**Sep**	**Oct**	**Nov**	**Dec**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**
Hoki trawl		Chatham Rise and Stewart-Snares Shelf								Cook Strait, west coast South Island, Puysegur		
Squid				Stewart-Snares Shelf		Auckland Islands and Stewart-Snares Shelf						
Southern blue whiting	Pukaki Rise and Campbell Rise											Bounty Islands
Scampi	Auckland Islands											

Captures include sea lions captured in nets and brought on deck (both dead and alive). Captures necessarily exclude the animals that exit trawls through the SLED, as well as bodies that are recovered in a decomposed state at the time of capture (Smith & Baird 2007b, Thompson & Abraham 2010, Thompson et al. 2013).

Interactions in the SQU 6T fishery are defined as the number of sea lions that enter the net and would have been captured if no SLED had been used. Until 2017 interactions were estimated using a statistical model fitting to observed capture rates both before and after the deployment of SLEDs, with an additional term to estimate SLED efficacy, i.e. the proportion of interactions in which the sea lion escapes via the SLED and survives (Thompson et al. 2013). For trawl fisheries that do not deploy SLEDs, the number of estimated interactions is equivalent to the number of estimated captures.

Early models suggested that the rate at which sea lions interacted with trawl nets was influenced by a number of factors, including year, distance from the rookery, tow duration, time of day, and change of tow direction (Smith & Baird 2005).

Unsurprisingly, following the introduction of SLEDs to the SQU 6T fishery in 2001–02, both the observed and estimated numbers of New Zealand sea lion captures declined (Table 4.6). The same trend was initially present in the mean estimated number of total interactions (reflecting reduced fishing effort over time). However model estimates of interaction rates became increasingly uncertain over time, because since the universal adoption of SLEDs in 2006–07, the interaction rate was confounded in the model with a corresponding and inversely correlated estimate of SLED efficacy, both of which were effectively unobservable. In the most recent models (Abraham & Berkenbusch 2017), estimated interaction rates were

effectively unbounded, and model estimates in particular years became unstable as new years' data were added. For this reason, from 2017 Fisheries New Zealand discontinued these models and no longer relies on new estimates of interaction rates generated using data from the period after SLEDs were in universal use. FNZ cautions against uncritical acceptance of previously published interaction rate estimates in Tables 4.5 and 4.6, for which the confidence intervals extend to increasingly implausible values in the period after 2006-07. New research employing an application of the SEFRA method described in Chapter 3 is underway for Auckland Island sea lions (PRO2017-10), and is expected to provide a more reliable means to estimate sea lion interactions in fisheries using SLEDs.

In fisheries not employing SLEDs, capture model estimates have remained useful. Observed and estimated New Zealand sea lion captures and capture rates in the Campbell Island southern blue whiting fishery were highly variable (Table 4.7). Following the 2012–13 season in which 21 male sea lion captures were observed (17 dead and 4 released alive), the fishing industry developed voluntary initiatives to mitigate sea lion mortalities including 100% SLED use. Fisheries New Zealand supports these non-regulatory mitigation measures which are set out in the Operational Plan for SBW6I. Since 2013, 100% of tows have been observed, and annual captures have ranged from 2 to 6 sea lions (annual average < 3). For the Auckland Islands scampi and the Stewart-Snares Shelf trawl fisheries, the observed and estimated numbers of New Zealand sea lion captures have fluctuated without trend (Table 4.8).

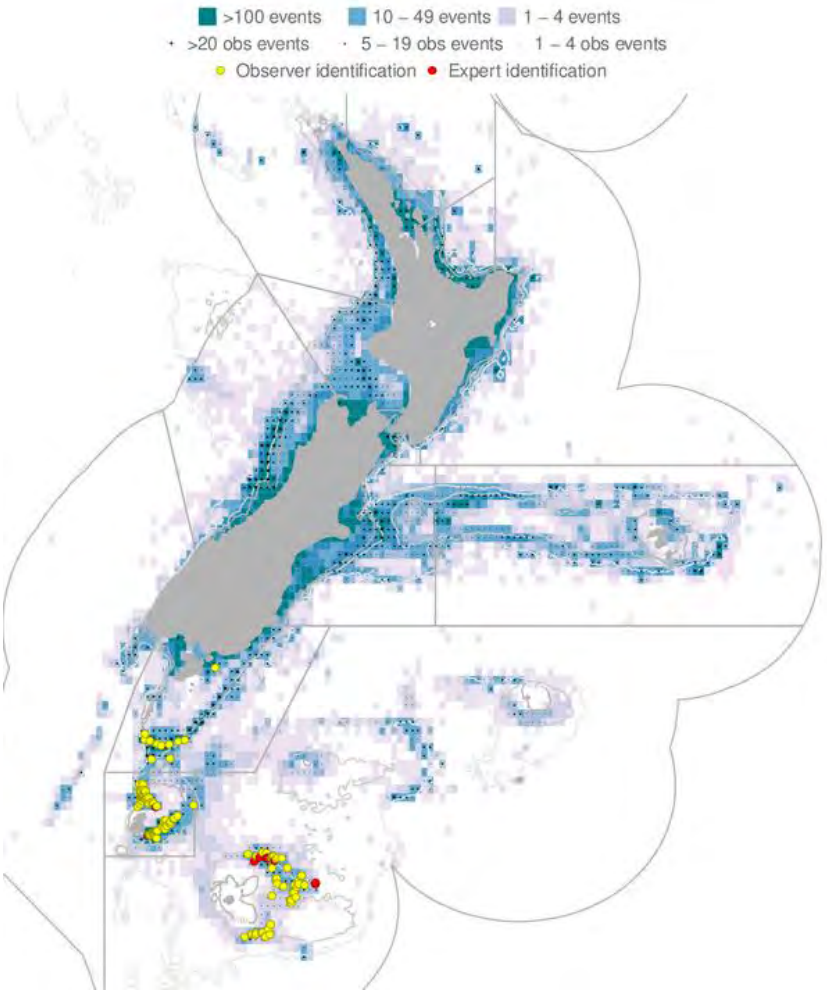


Figure 4.6: Distribution of trawl fishing effort and observed New Zealand sea lion captures, 2002–03 to 2016–17 (http://data.dragonfly.co.nz/psc. Data version v2017001). Fishing effort is mapped into 0.2-degree cells, with the colour of each cell indicating the amount of effort (number of fishing events). Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. Fishing is only shown if the effort could be assigned a latitude and longitude, and if there were three or more vessels fishing within a cell. [Updated for AEBAR 2018].

Table 4.5: Sea lion captures in all commercial trawl fisheries in New Zealand's Exclusive Economic Zone between 2002 and 2016 (http://data.dragonfly.co.nz/psc. Data version v2017001). Annual fishing effort (total number of tows), observer coverage (percentage of tows observed), number of observed sea lion captures (both dead and alive), observed capture rate (captures per 100 tows), the estimation method used (model, ratio estimate, or both combined), the number of estimated sea lion captures, and estimated interactions (with 95% confidence intervals, c.i.). Interactions are defined as the number of sea lions that would have been caught if no Sea Lion Exclusion Devices (SLEDs) had been used (see Thompson et al. 2013 and 2016 for details). [Updated for AEBAR 2018].

Fishing year	Fishing effort		Observed captures		Estimated captures			Estimated interactions+	
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.	Mean	95% c.i.
2002–03	130 195	5.25	12	0.175	Both	31	21–44	59	36–93
2003–04	120 850	5.42	21	0.321	Both	58	41–79	225	122–402
2004–05	120 468	6.40	14	0.182	Both	50	33–72	187	95–344
2005–06	109 933	6.02	15	0.227	Both	49	33–70	175	87–328
2006–07	103 323	7.68	12	0.151	Both	42	27–60	119	57–245
2007–08	89 533	10.11	11	0.122	Both	30	19–43	178	40–823
2008–09	87 552	11.15	3	0.031	Both	19	10–32	146	25–686
2009–10	92 892	9.71	15	0.166	Both	44	30–63	197	52–847
2010–11	86 085	8.65	6	0.081	Both	27	16–40	113	27–519
2011–12	84 418	11.09	1	0.011	Both	12	5–20	70	12–326
2012–13	83 838	14.79	25	0.202	Both	32	27–39	120	51–437
2013–14	85 110	15.49	4	0.030	Both	10	6–17	69	18–255
2014–15	78 767	17.22	8	0.059	Both	12	8–17	81	27–281
2015–16	78 030	16.64	4	0.031					
2016–17	78 172	17.56	3	0.022					

+ from 2017, interaction rate modelling was discontinued do to model instability arising from confounded interaction and SLED efficacy rates

Table 4.6: Sea lion captures in the Auckland Islands squid trawl fishery between 2002 and 2016 (http://data.dragonfly.co.nz/psc. Data version v2017001). Annual fishing effort (total number of tows), observer coverage (percentage of tows observed), number of observed sea lion captures (both dead and alive), observed capture rate (captures per 100 tows), the estimation method used (model, ratio estimate, or both combined), the number of estimated sea lion captures, estimated interactions, and estimated strike rate (with 95% confidence intervals, c.i.). Interactions are defined as the number of sea lion that would have been caught if no Sea Lion Exclusion Devices (SLEDs) had been used (see Thompson et al. 2013 and 2016 for details). [Updated for AEBAR 2018].

Fishing year	Fishing effort		Observed captures		Estimated captures			Estimated interactions+		Estimated interaction rate+	
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.	Mean	95% c.i.	Mean	95% c.i.
2002–03	1 466	28.38	11	2.644	Model	18	12–28	47	25–79	3.1	1.9–4.9
2003–04	2 594	30.57	16	2.018	Model	39	26–59	206	104–383	7.8	4.0–14.0
2004–05	2 693	29.93	9	1.117	Model	30	16–49	167	76–323	6.1	2.8–11.7
2005–06	2 459	22.37	10	1.818	Model	26	15–43	153	65–306	6.1	2.7–12.3
2006–07*	1 317	40.70	7	1.306	Model	15	9–25	93	33–216	6.8	2.4–15.6
2007–08	1 265	46.72	5	0.846	Model	12	6–22	160	24–804	9.4	1.8–39.8
2008–09	1 925	39.64	2	0.262	Model	7	2–15	134	14–672	5.3	0.7–24.5
2009–10	1 188	25.51	3	0.990	Model	12	5–26	165	22–818	10.8	1.9–44.8
2010–11	1 583	34.55	0	0.000	Model	3	0–10	90	5–501	4	0.3–17.9
2011–12	1 281	44.57	0	0.000	Model	2	0–6	60	3–319	3.5	0.3–16.5
2012–13	1 027	86.17	3	0.339	Model	4	3–6	73	8–384	5.3	0.8–24.5
2013–14	737	84.40	2	0.322	Model	2	2–4	47	5–231	5.0	0.7–22.8
2014–15	633	88.31	1	0.179	Model	1	1–3	44	3–236		
2015–16	1 367	92.25	0	0.000							
2016–17	1 280	70.39	3	0.333							

* SLEDs standardised and in widespread use.

+ from 2017, interaction rate modelling was discontinued due to model instability arising from confounded interaction and SLED efficacy rates

Table 4.7: Sea lion captures in Campbell Island southern blue whiting (SBW) and in Stewart-Snares Shelf trawl fisheries between 2002-03 and 2016-17 (http://data.dragonfly.co.nz/psc. Data version v2017001). Annual fishing effort (total number of tows), observer coverage (percentage of tows observed), number of observed sea lion captures (both dead and alive), observed capture rate (captures per 100 tows), the estimation method used (model or ratio estimate), and the number of estimated sea lion captures (with 95% confidence interval, c.i.) (see Thompson et al. 2013 and 2016 for details). [Updated for AEBAR 2018].

Fishing year	Fishing	Observed captures			Estimated captures		
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.
Subantarctic SBW							
2002–03	599	43	0	0	Model	1	0–3
2003–04	690	34	1	0.4	Model	3	1–9
2004–05	726	37	2	0.7	Model	5	2–13
2005–06	521	28	3	2.1	Model	10	3–22
2006–07*	544	32	6	3.5	Model	15	6–30
2007–08	557	41	2	0.9	Model	8	5–14
2008–09	627	20	0	0	Model	1	0–7
2009–10	550	43	11	4.7	Model	24	15–37
2010–11	886	39	6	1.7	Model	15	8–25
2011–12	592	77	0	0	Model	1	0–4
2012–13	693	100	21	3	Model	21	21–21
2013–14	588	100	2	0.3	Model	2	2–2
2014–15	566	100	6	1.1	Model	6	6–6
2015–16							
2016–17							
Stewart-Snares							
2002–03	17 093	6.69	0	0.000	Ratio	3	0-7
2003–04	16 452	7.49	1	0.081	Ratio	4	1-8
2004–05	17 303	10.81	3	0.160	Ratio	7	3-11
2005–06	15 664	7.20	1	0.089	Ratio	4	1-8
2006–07	15 024	9.01	1	0.074	Ratio	3	1-6
2007–08	12 491	12.34	1	0.065	Ratio	3	1-6
2008–09	11 065	15.24	0	0.000	Ratio	1	0-4
2009–10	12 433	15.89	1	0.051	Ratio	2	1-5
2010–11	10 752	12.42	0	0.000	Ratio	1	0-4
2011–12	11 848	16.01	1	0.053	Ratio	2	1-4
2012–13	11 629	26.18	1	0.033	Ratio	2	1-4
2013–14	11 299	21.91	0	0.000	Ratio	1	0-3
2014–15	10 007	24.10	1	0.041	Ratio	1	0-3
2015–16	9 592	21.15	1	0.049			
2016–17	9 571	20.99	0	0.000			

*SLEDs introduced in that year.

Table 4.8: Sea lion captures in trawl fisheries targeting scampi and targeting other species adjacent to the Auckland Islands between 2002-03 and 2016-17 (http://data.dragonfly.co.nz/psc. Data version v2017001). Annual fishing effort (total number of tows), observer coverage (percentage of tows observed), number of observed sea lion captures (both dead and alive), observed capture rate (captures per 100 tows), the estimation method used (model or ratio estimate), and the number of estimated sea lion captures (with 95% confidence interval, c.i.) (see Thompson et al. 2013 and 2016 for details). [Table continued on next page]. [Updated for AEBAR 2018].

Fishing year	Fishing effort		Observed captures		Estimated captures		
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.
Auckland Islands scampi							
2002–03	1 351	11.10	0	0.000	Ratio	7	2–15
2003–04	1 363	12.40	3	1.775	Ratio	10	5–18
2004–05	1 275	0.00	0	0.000	Ratio	8	2–16
2005–06	1 331	8.87	1	0.847	Ratio	8	3–16
2006–07	1 328	7.61	1	0.990	Ratio	8	3–16
2007–08	1 327	7.01	0	0.000	Ratio	8	2–15
2008–09	1 457	4.19	1	1.639	Ratio	10	3–18
2009–10	941	9.78	0	0.000	Ratio	5	1–11
2010–11	1 400	14.79	0	0.000	Ratio	7	2–15

2011–12	1 247	9.54	0	0.000	Ratio	7	2–14
2012–13	1 093	12.44	0	0.000	Ratio	6	1–12
2013–14	850	6.12	0	0.000	Ratio	5	1–11
2014–15	548	0.00	0	0.000	Ratio	3	0–8
2015–16	1 414	4.67	0	0.000			
2016–17	1 677	21.11	0	0.000			
Auckland Islands other							
2002–03	543	12.89	0	0.000	Ratio	2	0.0–1.1
2003–04	289	16.96	0	0.000	Ratio	1	0.0–1.0
2004–05	170	7.06	0	0.000	Ratio	1	0.0–1.8
2005–06	37	16.22	0	0.000	Ratio	0	0.0–2.6
2006–07	38	5.26	0	0.000	Ratio	0	0.0–2.6
2007–08	147	44.90	0	0.000	Ratio	0	0.0–1.4
2008–09	119	48.74	0	0.000	Ratio	0	0.0–0.8
2009–10	77	67.53	0	0.000	Ratio	0	0.0–1.3
2010–11	129	37.21	0	0.000	Ratio	0	0.0–1.5
2011–12	54	31.48	0	0.000	Ratio	0	0.0–1.8
2012–13	60	43.33	0	0.000	Ratio	0	0.0–1.7
2013–14	203	23.15	0	0.000	Ratio	1	0.0–1.0
2014–15	224	30.80	0	0.000	Ratio	1	0.0–1.3
2015-16	140	25.71	0	0.000			
2016–17	169	50.30	0	0.00			

4.4.2 MANAGING FISHERIES INTERACTIONS

For New Zealand sea lions, efforts to mitigate incidental captures in fisheries have historically focused on the SQU 6T fishery.

Current management reflects previously designated spatial fisheries closures. In 1982 the Minister of Fisheries established a 12-nautical mile exclusion zone around the Auckland Islands from which all fishing activities were excluded (Wilkinson et al. 2003); in 1995, the exclusion zone was replaced with a Marine Mammal Sanctuary with the same controls on fishing (Chilvers 2008). The area was subsequently designated as a Marine Reserve in 2003.

From 1992, the Ministry adopted a Fisheries-Related Mortality Limit (FRML; previously referred to as a maximum allowable level of fisheries-related mortality or MALFiRM) to set an upper limit on the number of New Zealand sea lions that can be incidentally killed each year in the SQU 6T trawl fishery (Chilvers 2008). If this limit is reached, the fishery is closed for the remainder of the season. The original 'MALFiRM' was calculated using the 'potential biological removals' approach (PBR; Wade 1998) and was used from 1992–93 to 2003–04 (Smith & Baird 2007a). Since 2003–04 the FRML has been translated into a fishing effort limit (maximum permitted number of tows) calculated based on assumptions about the interaction rate and SLED efficacy rate, regardless of the number of observed New Zealand sea lion captures. This approach was taken because since the introduction of Sea Lion Exclusion Devices (SLEDs), observed sea lion captures were no longer a reliable index of the number of sea lions interacting with the net, and there was uncertainty about the survival rate of sea lions exiting the net via the SLED.

SLEDs were first utilised on some vessels in the SQU 6T fishing fleet in 2001–02. SLED use increased in subsequent years. The use of SLEDs is not mandatory, but use of a certified SLED is required by the industry body (the Deepwater Group Ltd) and is necessary to receive the 'Discount Rate' relative to the tow limit applied by Fisheries New Zealand to ensure that estimated mortalities remain below the designated FRML (see section 4.4.2.2, below). For these reasons, from 2006–07 a standardised model Mark 13/3 SLED has been universally employed by all vessels in the SQU 6T fleet (Figure 4.7). SLED deployment is monitored and audited by Fisheries New Zealand Observers.

From 2017, advice to manage sea lion interactions in this fishery has been developed in consultation with the Squid 6T Operational Plan Technical Advisory Group (SqOPTAG), including representatives from government and stakeholder groups as well as technical experts and advisors. Under the present Operational Plan, adopted in December 2017, Fisheries New Zealand sets a fishing-related mortality limit (FRML) for sea lions in the Auckland Islands squid trawl fishery (SQU 6T) based on estimation of a Population Sustainability Threshold (PST) using a Bayesian population dynamic model (Roberts & Doonan 2016). The

PST represents the maximum number of anthropogenic mortalities that the population can sustain while still achieving a defined population objective. For the Auckland Islands sea lion population, the choice of population objective underlying the current PST is as follows: 'Fisheries mortalities will be limited to ensure that the impacted population is no more than 5% lower than it would otherwise be in the absence of fishing mortality, with 90% confidence, over five years. The choice of the population objective is a policy decision.

4.4.2.1 SEA LION EXCLUSION DEVICE (SLED) DEVELOPMENT AND USE

In 2004, the Minister of Fisheries requested that the squid fishery industry organisation (Squid Fishery Management Company), government agencies and other stakeholders with an interest in sea lion conservation work collaboratively to develop a plan of action to determine SLED efficacy. In response, an independently chaired working group (the SLED Working Group) was established to develop an action plan to determine the efficacy of SLEDs, with a particular focus on the survivability of New Zealand sea lions that exit the nets via the exit hole in the SLED. The group undertook a number of initiatives, most notably the standardisation of SLED specifications (including grid spacing) across the fleet (DOC CSP project MIT 2004/05; Clement and Associates Ltd. 2007) and the establishment of an underwater video monitoring programme to help understand the fate of New Zealand sea lions that exit the net via the SLED. White light and infra-red illuminators were tested. Sea lions were observed outside the net on a number of occasions, but only one fur seal and one New Zealand sea lion were observed exiting the net via the SLED (on tows when white light illumination was used). The footage contributed to understanding of SLED performance, but established that video monitoring was only suitable for tows using mid-water gear, as the camera view was often obscured on tows where bottom gear was used (Middleton & Banks 2008). The SLED Working Group was disbanded in early 2010.

Table 4.9: Maximum allowable level of fisheries-related mortality (MALFiRM) or fisheries-related mortality limit (FRML) from 1991 to 2015. Note that direct comparisons among years of the limits in Table 4.9 are not possible because the assumptions underlying the MALFiRM or FRML changed over time. [Updated for AEBAR 2018].

Year	MALFiRM or FRML	Discount rate	Management actions
1991–92	16 (female only)		
1992–93	63		
1993–94	63		
1994–95	69		
1995–96	73		Fishery closed by MFish (4 May)
1996–97	79		Fishery closed by MFish (28 March)
1997–98	63		Fishery closed by MFish (27 March)
1998–99	64		
1999–00	65		Fishery closed by MFish (8 March)
2000–01	75		Voluntary withdrawal by industry
2001–02	79		Fishery closed by MFish (13 April)
2002–03	70		Fishery closed by MFish (29 March), overturned by High Court
2003–04	62 (124)	20%	Fishery closed by MFish (22 March), overturned by High Court FRML increased
2004–05	115	20%	Voluntary withdrawal by industry on reaching the FRML
2005–06	97 (150)	20%	FRML increased in mid-March due to abundance of squid
2006–07	93	20%	
2007–08	81	35%	
2008–09	113 (95)	35%	Lower interim limit agreed due to the decrease in pup numbers
2009–10	76	35%	
2010–11	68	35%	
2011–12	68	35%	
2012–13	68	82%	
2013–14	68	82%	
2014–15	68	82%	
2015–16	68	82%	
2016–17	68	82%	
2017–18	38	75%	
2018–19	38	75%	

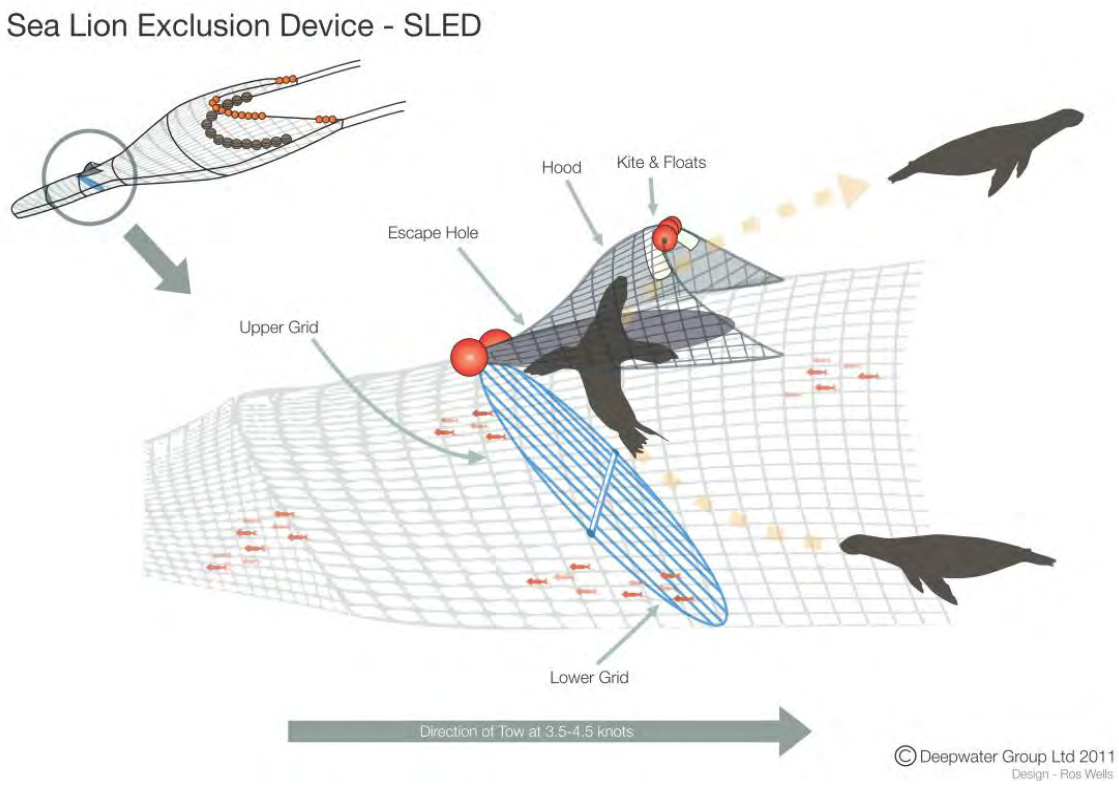


Figure 4.7: Diagram of a New Zealand sea lion exclusion device (SLED) inside a trawl net. Image courtesy of the Deepwater Group.

4.4.2.2 MANAGEMENT SETTINGS IN THE SQUID 6T FISHERY

Before the widespread use of SLEDs, New Zealand sea lions incidentally caught during fishing were usually retained in trawl nets and hauled onboard, allowing observers to gain an accurate assessment of the number of New Zealand sea lions interactions on observed tows in a given fishery. This enabled a robust estimation of the total number of New Zealand sea lions killed. However, following the introduction of SLEDs, the number of New Zealand sea lions interacting with trawls but exiting via the SLED was unobservable, so interaction rate is instead estimated statistically. Subsequently, a management setting meant to approximate the interaction rate, i.e., the 'Strike Rate' is set by Fisheries New Zealand (along with a second setting, the 'Discount Rate' representing SLED efficacy, see below) to inform a proxy estimate of potential sea lion fatalities per 100 tows. This proxy estimate is then used to set an effort limit on the operation of the fishery, to ensure that sea lion fisheries mortalities remain below the FRML.

The 'Discount Rate' is a management setting that approximates SLED efficacy, i.e., the proportion of sea lion interactions in which the sea lion exits the SLED and survives. The current management regime for the SQU 6T fishery provides that the discount rate will be applied to all tows in which an approved Mark 13/3 SLED is used and relevant requirements of the Operational Plan met (e.g., notification of intention to fish in SQU 6T and reporting requirements). Discount rates applied between 2003–04 and 2018–19 are shown in Table 4.9.

The SLED Discount Rate is a fisheries management setting and should not be confused with the actual estimated survival rate of New Zealand sea lions exiting the SLED; for example the Discount Rate may be set deliberately lower than the actual estimated SLED efficacy rate, reflecting cautious management in the presence of uncertainty. New research is planned to provide more robust estimates of SLED efficacy (see Section 4.4.3.1, cryptic mortality, below).

4.4.3 MODELLING POPULATION-LEVEL IMPACTS OF FISHERIES INTERACTIONS

Consistent with terminology used in the SEFRA methodology (Chapter 3), Fisheries New Zealand has now adopted the term 'Population Sustainability Threshold' or

PST to denote the number of anthropogenic deaths that a population can sustain while still meeting a defined population recovery or stabilisation outcome, evaluated via simulations using a demographic population model. The choice of reference outcome is a policy decision.

For sea lions, the estimation of the PST derives from a demographic population model informed by mark-recapture observations, annual pup census results, estimated fisheries-related deaths, and the estimated age distribution of lactating females, as described in Roberts & Doonan (2016). The model also supported a quantitative risk assessment to estimate the effects of non-fishery threats (Section 4.4.3.4, below). In 2017, additional model runs were carried out under project SEA2026-30, incorporating the newest pup count data from Figure 4.3 but not the 2014-15 and 2015-16 mark-recapture data (Roberts 2017b). These outputs were then used to update management options for the Operational Plan for the SQU 6T fishery in 2017. Applying the chosen population reference outcome -- i.e. 'the impacted population is no more than 5% lower than it would otherwise be in the absence of fishing mortality, with 90% confidence, over five years' -- the model yielded a corresponding PST of 46 annual fishery-related deaths for the Auckland Island sub-population. Deducting an estimated 7 annual mortalities for the SCI 6A fishery, and one for estimated annual mortality in other trawl fisheries adjacent to the Auckland Islands, results in a FRML of 38 for the SQU 6T fishery (Table 4.9).

Separately, a simpler PBR approach has been used to the number of fisheries deaths that would cause adverse effects on the Campbell Island sea lion sub-population; the PBR is currently estimated to be 25 annual deaths (Roberts et al. 2014b). See Section 4.4.3.2, below. The Campbell Island PBR estimate will be updated in 2019.

4.4.3.1 CRYPTIC MORTALITY

SLEDs are effective in allowing most New Zealand sea lions to exit a trawl but occasionally a sea lion does not exit and is drowned and retained in the net. These are recorded as observed captures. However there remains some uncertainty about the fate of sea lions that are not retained in the net, some of which may nonetheless die as a consequence of the interaction. Interactions that result in unobservable deaths are termed 'cryptic mortality'. Sources of cryptic mortality are best understood by categorising five potential fates of a sea lion entering a trawl:

i. exits the net via SLED and survives (survivor);
ii. dies in net and is retained (observable capture);
iii. dies in the net but the body is subsequently lost without being recovered on the vessel ('body non-retention');
iv. exits the net but dies from head injuries sustained during interaction with the SLED ('mild traumatic brain injury', or MTBI);
v. exits the SLED but is at the limit of its breath hold ability and drowns before reaching the surface ('post-escape drowning').

Collectively, points iii–v constitute cryptic mortality. The following section describes past research undertaken to estimate various components of cryptic mortality.

New research is underway to utilize and integrate the results of these separate projects, to estimate cryptic mortality in trawls employing SLEDs in a comprehensive way, fitted to observed capture rates both before and after the adoption of SLEDs. Project PRO2018-09 will deliver these estimates in 2019.

4.4.3.1.1 POST-ESCAPE DROWNING

Between 1999–2000 and 2002–03, an experimental approach was taken to estimate interaction rates and SLED efficacy rates, by intentionally capturing animals as they exited the escape hole of a SLED. Cover nets were added over the escape holes of some SLEDs and sea lions were restrained in these nets after they exited the SLED. An underwater video camera was deployed in 2001 to assess the behaviour and the likelihood of post-exit survival of those animals that were retained in the cover nets (Wilkinson et al. 2003, Mattlin 2004). Due to low sample sizes and ambiguous interpretation of necropsy results, this work was judged to be inconclusive (Roe 2010). However video data collected in the course of this study may prove useful to inform the estimation of post-escape drowning rates for sea lions exiting the net via the SLED.

4.4.3.1.2 MILD TRAUMATIC BRAIN INJURY

Necropsies were conducted on animals recovered from the cover net trials and on those incidentally caught and recovered from vessels operating in the SQU 6T, SQU 1T and SBW 6I fisheries. All of the New Zealand sea lions returned for necropsy died as a result of drowning rather than physical trauma from interactions with the trawl gear

including the SLED grid (Roe & Meynier 2010, Roe 2010). Necropsies were designed to assess the nature and severity of trauma sustained during capture and to infer the survival prognosis had those animals been able to exit the net (Mattlin 2004). However, problems associated with this approach limited the usefulness of the results. For example, sea lions had to be frozen on vessels and stored for periods of up to several months before being thawed for 3–5 days to allow necropsy. Roe & Meynier (2010) concluded that this freeze-thaw process created artefactual lesions that mimic trauma but, particularly in the case of brain trauma, could also obscure real lesions. Further, two reviews in 2011 concluded that the lesions in retained animals may not be representative of the injuries sustained by animals that exit a trawl via a SLED (Roe & Meynier 2010, Roe 2010).

Notwithstanding the limitations of the necropsy data in assessing trauma for previously frozen animals, it was possible to determine that none of the necropsied animals sustained sufficient injuries to the body (excluding the head) to compromise survival (Roe & Meynier 2010, Roe 2010). Any head trauma, (most likely due to impacts with the SLED grid), could not be ruled out as a potential contributing factor to the animals death (Roe & Meynier 2010, Roe 2010). In order to quantify the likelihood of a sea lion experiencing physical trauma sufficient to render the animal insensible (and therefore likely to drown) after a collision with a SLED grid, a number of factors need to be assessed. These include the likelihood of a head-first impact, the speed of impact, the angle of impact relative to individual SLED grid bars and relative to the grid plane, the location of impact on the grid, sea lion head mass, and the risk of brain injury for a given impact speed and head mass. The effect of multiple impacts also needs to be considered. Estimates for each of these factors were obtained from a number of sources, including necropsies (for head mass), video footage of Australian fur seals interacting with Seal Exclusion Devices (SEDs) (for impact speed, location and body orientation) and biomechanical modelling of impacts on the SLED grid (for the risk of brain injury).

In the absence of sufficient video footage of New Zealand sea lions interacting with SLEDs, footage of fur seals (thought to be Australian fur seals) interacting with SEDs in the Tasmanian small pelagic mid-water trawl fishery has been used (Lyle 2011). The SEDs are similar, but not identical, to the New Zealand SLEDs in that both have sloping steel grids to separate the catch from pinnipeds and guide the latter toward an escape hole in the trawl. Lyle & Willcox (2008) conducted a camera trial between January 2006 and February 2007 to assess the efficacy of the SED and documented 457 interactions for about 170 individual fur seals. Lyle (2011) reanalysed the footage to estimate impact speed, impact location across the SED grid and body orientation at the time of impact. The situation faced by New Zealand sea lions in a squid trawl is different to that faced by the fur seals studied by Lyle and co-workers, but these are closely related otariids of similar size and, in the absence of specific data, Australian fur seals are considered a reasonable proxy to estimate impact speed, impact location and body orientation for New Zealand sea lions.

The risk of brain injury was assessed by biomechanical testing and modelling. Tests using an artificial 'head form' (as used in vehicular 'crash test' studies) were used to assess the likelihood of brain injury to New Zealand sea lions colliding with a SLED grid (Ponte et al. 2010, 2011). In an initial trial, the head form (weighing 4.8 kg) was launched at three locations on the SLED grid at a speed of 10 $m.s^{-1}$ (about 20 knots). This was considered a 'worst feasible case' collision, representing the combined velocities of a sea lion swimming with a burst speed of 8 $m.s^{-1}$ (after Ray 1963, Fish 2008) and a net being towed at 2 $m.s^{-1}$ (about 4 knots). A 'head injury criterion' (HIC, a predictor of the risk of brain injury), was calculated based on criteria validated against human-vehicle impact studies and translated into the probability of 'mild traumatic brain injury' (MTBI) for a given collision, taking into account differences between human and sea lion head and brain masses. MTBI is assumed to have the potential to lead to insensibility or disorientation and subsequent death through drowning for a New Zealand sea lion experiencing such an injury at depth. Ponte et al. (2010) calculated that a collision at the stiffest part of the SLED grid at this highest feasible speed had a very high risk of MTBI, especially for smaller sea lions (female and small, immature males). This provides an upper bound for the assessment of risk but Ponte et al. (2010) also imputed risk at speeds below the maximum tested (10 $m.s^{-1}$).

In a follow-up study, after a research advisory group meeting with other experts, Ponte et al. (2011) tested a wider variety of impact locations on the grid and various angles of impact relative to the bars and to the plane of the grid and combined these to produce a HIC 'map' for a SLED grid. This HIC map can be used to estimate the risk of MTBI for a collision by a sea lion at any given speed, location, and orientation used to model the risk of MTBI.

The data collected from the footage of Australian fur seal SED interactions (Lyle 2011) and the biomechanical modelling (Ponte et al. 2010, 2011), were combined in a simulation-based probabilistic model to estimate the risk of a sea lion suffering a mild traumatic brain injury when striking a SLED grid (Abraham 2011). The simulation involved selecting an impact location on the SLED grid (from the fur seal data), selecting a head mass (from New Zealand sea lion necropsy data) and an impact speed (from the fur seal data), calculating the head impact criterion (from the HIC map), scaling the HIC to the head mass and impact speed and calculating the expected probability of mild traumatic brain injury, MTBI. Both 45° and 90° impacts were considered, with the former reflecting the angle of a grid when deployed, adopted as the base case. The head masses used may be at the lower end of the range of head masses for New Zealand sea lions, due to the possible bias in those that were caught and necropsied. Impact speeds were drawn from the distribution of speeds observed for fur seals colliding with SEDs (2–6 $m.s^{-1}$) and these are broadly consistent with the combined tow speed and observed swimming speeds of New Zealand sea lions in the wild (Crocker et al. 2001). Different scaling of HIC values was assessed to gauge sensitivity.

For the base case, the simulation results indicated there was a 3.3% chance of a single head-first collision resulting in MTBI with a 95 percentile of 15.7% risk of MTBI (Abraham 2011). Sensitivities modulating single parameters resulted in up to 6.2% probability of a single collision resulting in MTBI. One sensitivity trial involving changes in multiple parameters resulted in a 10.9% probability of MTBI. This scenario considered impact speeds 20% above those measured for fur seals, multiple collisions with the grid, and the least favourable values of scaling exponents used in scaling the test HIC values and calculating MTBI from the HIC (Abraham 2011). These results are probabilities of MTBI resulting from a single head-first collision but, because each individual can have multiple interactions with the grid while in a trawl, and some of these will not be head-first, the risk of a head collision that results in MTBI for an individual animal over the course of a tow will then be higher than the risk associated with a single interaction sequence. Using Australian observations, Abraham (2011) estimated the number of head-first collisions per interaction as 0.74, leading to an estimated probability of MTBI for a New Zealand sea lion interacting with a trawl of 2.7%. Single parameter sensitivity runs increased this to up to 4.6% and the multiple parameter sensitivity using the scenario described above increased it to 8.2% (Abraham 2011). Assuming synergistic interaction between successive head-first strikes (each collision carrying five times more risk than previous ones) did not appreciably increase the overall risk because few fur seals had multiple head-first collisions. These results indicate that rates of death by MTBI for New Zealand sea lions interacting with the SLED grid is probably low, although some remaining areas of uncertainty were identified (see below).

4.4.3.1.3 BODY NON-RETENTION

From first principles and considering SLED design (Figure 4.7) it seems unlikely that body non-retention rates are high, because:

i) the escape opening of SLEDs employed in New Zealand fisheries is at the top of the net, while drowned pinnipeds are observed to be negatively buoyant;

ii) forward-facing hoods are designed to allow exit for actively swimming animals;

iii) hood floats are designed to close the escape opening in the event that the trawl net becomes inverted (turns upside down), Preliminary results of SLED monitoring trials in overseas jurisdictions appear to support the conclusion that drowned pinnipeds are unlikely to be lost, and thereby not counted among observed captures, in trawls employing SLEDs. Collaboration with overseas researchers is ongoing to optimise the use of all available data. Developing or accessing improved means of estimating body non-retention is a high priority.

4.4.3.2 PBR ASSESSMENT FOR CAMPBELL ISLAND POPULATION

Following an unprecedented number of incidental captures of New Zealand sea lions in the Campbell Rise Southern blue whiting fishery (SBW 6I) in 2013, a review was conducted of PBR guidelines and relevant scientific literature to inform the selection of appropriate PBR parameter values for the Campbell Island sub-population (Roberts et al. 2014b). The PBR is a traditional approach to defining a safe level of human-related mortalities of marine mammals, which was originally developed for the US

Marine Mammals Protection Act (Wade 1998). It is calculated as:

$$PBR = N_{min} \times {}^{R_{max}}/_{2} \times F_R$$

where R_{max} is the population growth rate at very low population size with only natural mortality operating, N_{min} is a 'minimum' estimate of the total population size and F_R is a recovery factor applied to account for uncertainty or biases that may otherwise lead to overestimation of the PBR and so hinder recovery to an optimum sustainable population (OSP) level. The value of F_R may also be adjusted to meet different population management objectives.

The pup census at Campbell Island of 681 pups in 2010 (Maloney et al. 2012) was taken as a robust lower estimate of total pup production. A matrix modelling analysis was conducted to estimate plausible pup to whole-of-population multipliers of 4.5 and 5.5, which were applied to the pup census estimate to calculate N_{min} values of 3065 and 3746. The rate of increase in pup counts from a time series of pup censuses was used as an approximation to whole-of-population growth rate for estimating a credible lower limit of R_{max}. Values of 0.06, 0.08 and 0.10 were used in PBR calculations, with the upper and lower limits considered as plausible bounds for this parameter used in a sensitivity analysis. The Auckland Islands and Campbell Island sub-populations are likely to constitute demographically independent populations and so, according to the latest guidelines on PBR assessment, may be assessed as separate stocks (Moore & Merrick 2011). Therefore the recovery factor (F_R) of 0.5 was used for stocks of a threatened species with unknown (or not declining) population trajectory. The latest PBR guidance literature recommends a more conservative F_R of 0.1 for stocks of an endangered species and is the lower limit that might be considered for declining populations of a threatened species (Roberts et al. 2014b).

Previous to 2005–06 the annual number of captures was very low, though capture rate appears to have increased since, with the greatest number of captures in 2012–13 (Table 4.7). Running means of capture levels (3- and 5-year) were also calculated for comparison with PBR estimates. For an F_R of 0.5, and the selected estimates of N_{min} (3065) and R_{max} (0.08) the calculated PBR was 61. Estimated captures did not exceed the PBR in any year when the default F_R of 0.5 was used, regardless of which other parameter values used. When the lower F_R of 0.1 was used, the calculated PBR of 12 was exceeded in two years when using a 3-year running mean of captures and in one year with a 5-year running mean of captures. When a F_R of 0.2 was used, the calculated PBR of 25 was not exceeded in any year. There has been a very strong bias towards males in observed captures (Thompson et al. 2013). An array of female-only PBRs was estimated by halving the PBR for all animals and was not exceeded by female captures in any year regardless of which combination of parameter values was used (Roberts et al. 2014b).

4.4.3.3 INTEGRATED BAYESIAN MANAGEMENT PROCEDURE EVALUATION MODEL

Note that from 2016, on the advice of the AEWG, the 'BFG model' and its precursors described in this section are no longer used to inform MPI management, having been replaced by the integrated Bayesian multi-threat risk model described in Roberts & Doonan (2016). The description in this section is only retained for historical completeness.

From 2000 to 2011, an integrated Bayesian management procedure evaluation model having both population and fishery components was used to assess the likely performance of a variety of management control rules, each of which could be used to determine the FRML for a given SQU 6T season (Breen et al. 2003, 2016, Breen & Kim 2006a, 2006b). The model underwent several iterations. An early version, developed in 2000–01, was a relatively simple, deterministic, partially age-structured population model with density-dependence applied to pup production (Breen et al. 2003). An updated version called the Breen-Kim model was built in 2003 to render it fully age-structured and to incorporate various datasets supplied by DOC (Breen & Kim 2006a, 2006b). This model was further revised in 2007–08 to incorporate the latest New Zealand sea lion population data and to address various model uncertainties and called the BFG model (after its authors, Breen et al. 2016). In 2009, the model was again updated to incorporate the low New Zealand sea lion pup counts observed in 2008–09 (and thus better reflect the observed variability in pup survival and pupping rates), as well as incidental captures in fisheries other than SQU 6T. The BFG model was re-run in 2011 using the same underlying data and structure as in 2009 to evaluate the effect of different model assumptions about the survival of New Zealand sea lions that exit trawl nets via SLEDs (see below). Additional details can be found in Breen et al. (2016).

The BFG model incorporated various population dynamics observations (tag-resighting observations, pup births and mortality, age at maturity) as well as incidental captures and catch-at-age data from the SQU 6T trawl fishery. The model was projected into the future by applying the observed dynamics and a virtual fishery model that is managed in roughly the same way as the real SQU 6T fishery. A large number of projections were run and used to assess the likely performance of a wide range of different management control rules against defined performance criteria.

4.4.3.4 COMPREHENSIVE QUANTITATIVE RISK ASSESSMENT

In 2016 a quantitative risk assessment estimating the potential impacts of both fisheries and non-fishery threats to the New Zealand sea lion (*Phocarctos hookeri*) was undertaken to inform the development of a Threat Management Plan for the species (Roberts & Doonan 2016).

A panel of national and international independent experts, supported by relevant subject matter advisors, was convened to provide guidance on the level of threats to New Zealand sea lions and review the demographic assessment. The first of two workshops was held from 28 April to 1 May 2015. It built on previous discussions at a pup mortality workshop held in June 2014, but considered all threats to all sea lion age groups. The initial stage of the risk assessment model – the demographic assessment – was completed in advance of the first workshop, in order for the panel to review and provide recommendations for model improvements (Debski & Walker 2016).

Separate demographic assessment models were developed for females at the Auckland Islands and Otago Peninsula populations, integrating information from mark-recapture observations, pup census and the estimated age distribution of lactating females (Auckland Islands only). With respect to the Auckland Islands assessment, good fits were obtained to all three types of observation and the model structure and parameter estimates appeared to be a good representation of demographic processes that have affected population decline there (primarily low pup survival and low adult survival). The Otago Peninsula assessment made use of a much smaller number of observations, however still produced good estimates of all key demographic rates, with much higher pup survival relative to the Auckland Islands population (Roberts & Doonan 2016).

A two-stage assessment of the effects of threats was undertaken where the consequences of removing the effects of a threat was estimated in terms of the population growth rate of mature individuals in 2037. This used threat-specific mortality estimates at age (provided by MPI/DOC subsequent to two dedicated TMP workshops, see Debski & Walker 2016), in which:

1. 'Triage' projections were undertaken for all assessed threats using the upper bound estimates of threat-related mortality to screen out threats that had little effect on projected growth rate;

2. 'Best-estimate' projections were undertaken using the best estimate of threat-specific mortality for all threats that passed through the triage stage (Roberts & Doonan 2016).

The triage of the risks posed to New Zealand sea lions was conducted in order to limit the number of risks to be included in the more detailed Markov chain Monte Carlo, (MCMC) modelling. To do this, a simple model was used to assess the upper bound, or worst case scenario, of the threat by predicting the response of the population to that threat being removed. The results of this triage are not considered to be the best estimate of the risks posed to the New Zealand sea lions, but a mechanism to reduce the list of the threats to those that have the largest influence.

Triage model run projection outputs for the Auckland Islands using the final model are shown in Figure 4.8, and for the Otago Peninsula population in Figure 4.9. The black line in Figures 4.8 and 4.9 indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately.

The effects of removing the threats that act on pups (i.e., *Klebsiella pneumoniae* (a bacterial disease), hookworm,

wallows[7]) have a delayed effect on the size of the mature population of sea lions. This is because the pups that will survive still need time to mature before they are included in the modelled mature female population (Roberts & Doonan 2016, Debski & Walker 2016).

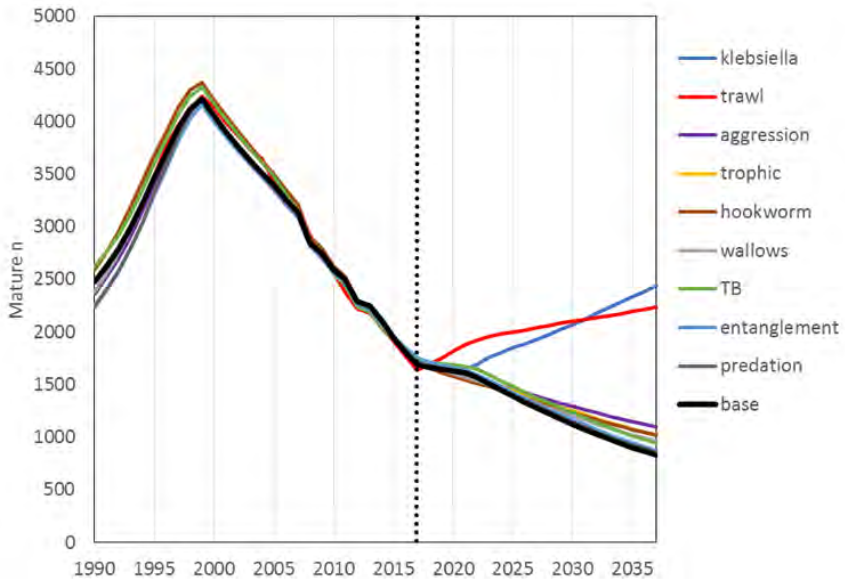


Figure 4.8: Triage projections of model estimated mature n at the Auckland Islands in the period 1990–2037, using upper values of threat mortality. The black dotted line indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately as coloured lines.

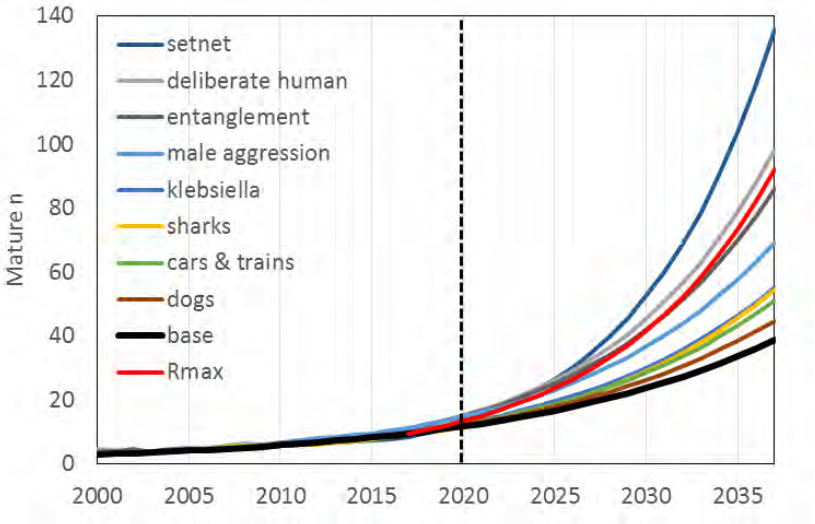


Figure 4.9: Triage projections of model estimated mature n at the Otago Peninsula in the period 1990–2037, using upper values of threat mortality. The black dashed line indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately as coloured lines except for the red line, which shows population growth at R_{max} (assumed to be 0.12).

[7]While this report refers to this threat as 'wallows', this includes all types of hole, drop or barrier that either causes a sea lion pup to be separated from its mother or drown.

Removal of the upper bound of *Klebsiella* risk creates the largest change in population size over the 20-year time period (2017–37), however the population reacts more quickly to the removal of the upper bound of estimated trawl interactions as this acts directly on the mature females. The ratio of mature female population in 2037 compared with 2017 is 1.30 when *Klebsiella* is removed, and 1.24 when trawl interactions are removed (Roberts & Doonan 2016). The independent panel considered that some of the upper bounds used in the triage process were unlikely to be realistic and should be treated with caution (Debski & Walker 2016).

For the Otago Peninsula model, the removal of upper bounds of some threats produced a very rapidly growing population, higher than the assumed maximum optimal growth rate (R_{max}) (Figure 4.9). This indicates that the upper bounds used for set net and deliberate human threats were probably unrealistically high (Roberts & Doonan 2016, Debski & Walker 2016).

For the Auckland Islands population, best-estimate projections were undertaken for commercial trawl-related mortality, *Klebsiella*-related mortality of pups, trophic effects (food limitation), pups drowning in wallows, male aggression and hookworm mortality. These threats were compared with the base run – a continuation of demographic rates since 2005 (λ_{2037} = 0.961, 95% c.i.: 0.890–1.020). A positive growth rate was obtained only with the alleviation of *Klebsiella* (λ_{2037} = 1.005, 95% c.i.: 0.926–1.069). When assuming the most pessimistic view of cryptic mortality (all interactions resulted in mortality and associated death of pups), alleviating the effects of commercial trawl-related mortality resulted in an increased population growth rate relative to the base run, but did not reverse the declining trend (λ_{2037} = 0.977, 95% c.i.: 0.902–1.036) (Figure 4.10). The alleviation of trophic effects (food limitation), had the next greatest effect (λ_{2037} = 0.974, 95% c.i.: 0.905–1.038) and all other threats had a minor effect relative to the base run projection (increase in λ_{2037} of less than 0.01) (Figure 4.11, Roberts & Doonan 2016).

For the Otago Peninsula population, similar effects were estimated with the alleviation of any of the threats that passed through triage: commercial set-net fishery-related mortality; direct human mortality; pollution-related entanglement; and male aggression, relative to the base run projection (λ_{2037} = 1.070, 95% c.i.: 1.053–1.087). Deliberate human mortality was estimated to have the greatest effect on projected population size (λ_{2037} = 1.093, 95% c.i.: 1.075–1.112) (Figure 4.12, Roberts & Doonan 2016).

For the Auckland Islands population (the largest breeding site for the species), model outputs suggest that if demographic rates used to simulate forward population trajectories (i.e., sampled from the past 20-year period) are accurate, then the TMP goals would be difficult to achieve with the complete alleviation of a single threat. In this context, the most effective approach to meeting the goals of the TMP may be to spread the management effort across the suite of key perceived threats identified from this assessment.

The population projections are sensitive to assumptions about what demographic rates are being realised in the population, in the context of considerable environmental variability on a decadal scale, with likely effects on critical demographic rates driving population change. A high priority is the development of tools for monitoring the effects of environmental and management drivers on threat-specific mortality and influential demographic rates (Roberts & Doonan 2016). For example new work to examine factors affecting pup survival (PRO2017-08C) commenced in 2018, and integrative ecosystems research to investigate decadal scale climate variability potentially affecting sea lion demographic parameters is being progressed under separate contract (ZBD2018-05).

The assessment for some of the key threats to New Zealand sea lions was hampered by incomplete information for estimating threat-specific mortality, e.g., relating to the causes of pup mortality during the entire first year of life and of cryptic commercial trawl-related mortality. In addition, a lack of demographic observations for the Campbell Island and Stewart Island populations (the second- and third-largest breeding populations, respectively), precluded the development of comprehensive quantitative risk assessments for these populations (Roberts & Doonan 2016).

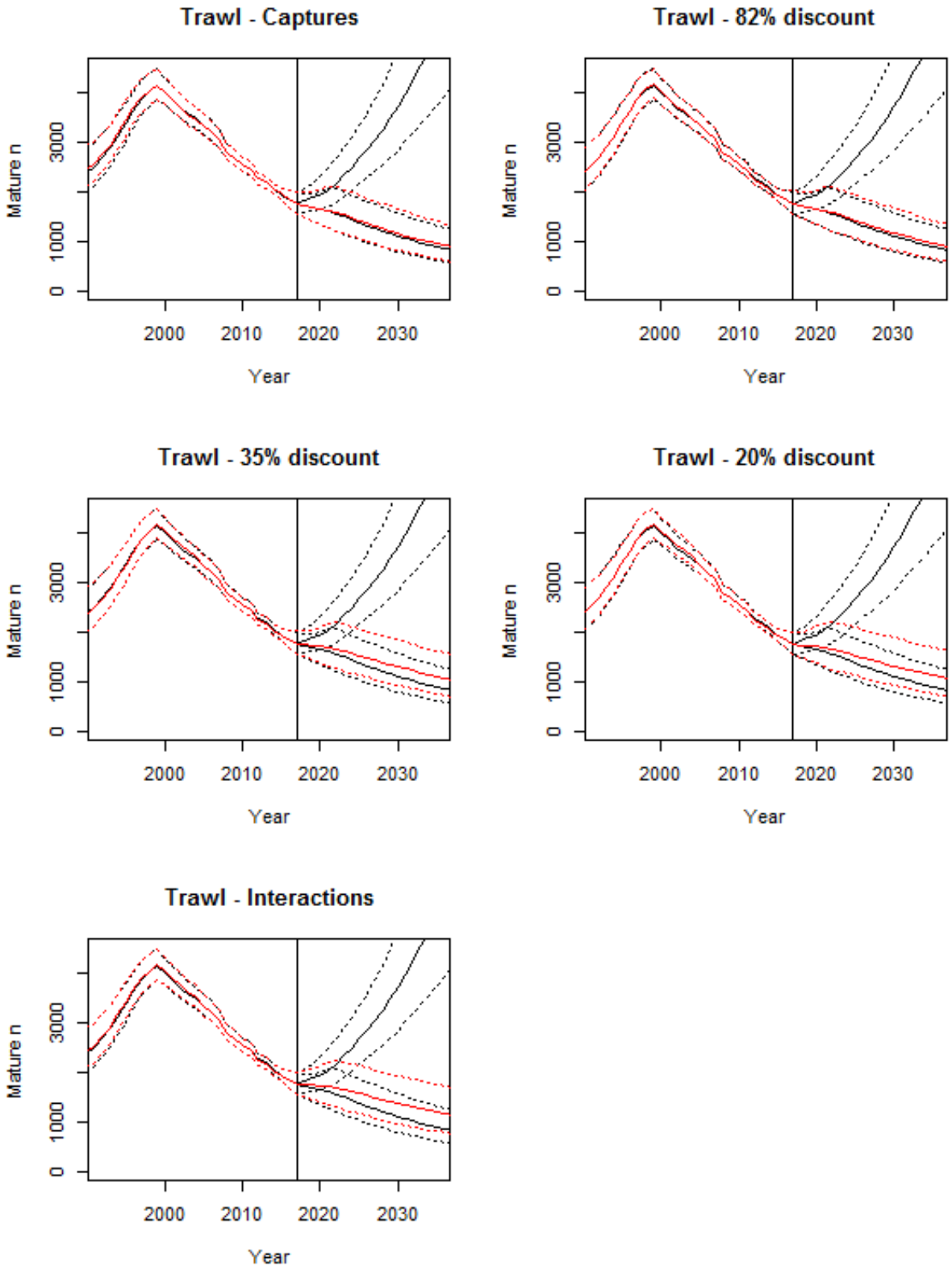


Figure 4.10: Best-estimate projections of mature n at the Auckland Islands in the period 1990–2037 for trawl fishery mortality scenarios. Lower black lines are with all threats (base run); upper black lines are with the 'max growth' scenario (1990–93 estimate of Surv0, 1990–98 estimates of Surv6–14 and 1990–99 estimate of PrP; red lines are with a threat alleviated (from Roberts & Doonan 2016).

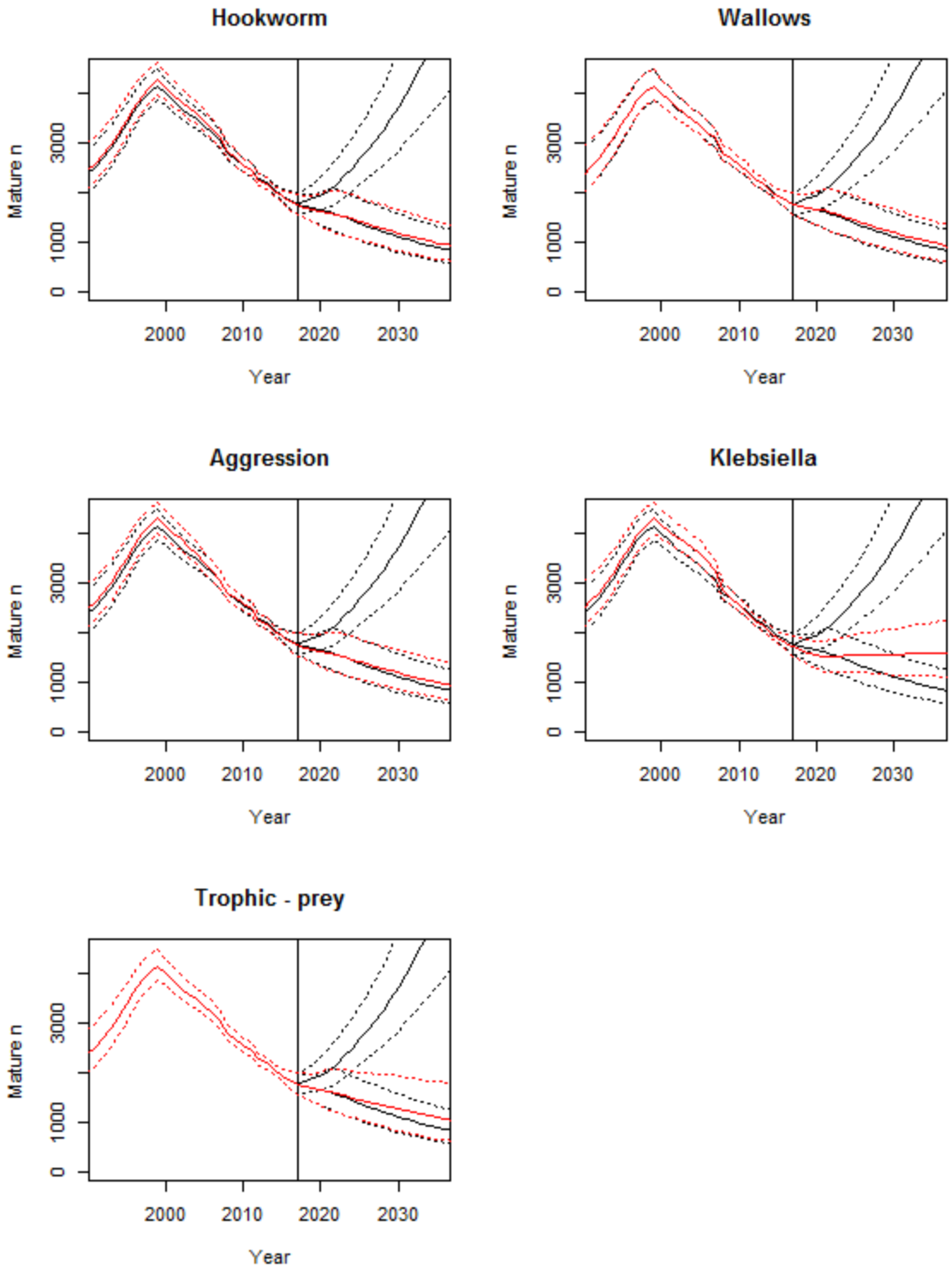


Figure 4.11: Best-estimate projections of mature n at the Auckland Islands in the period 1990–2037 for all other threat scenarios. Lower black lines are with all threats (base run); upper black lines are with the 'max growth' scenario (1990–93 estimate of Surv0, 1990–98 estimates of Surv6–14 and 1990–99 estimate of PrP; red lines are with a threat alleviated (from Roberts & Doonan 2016).

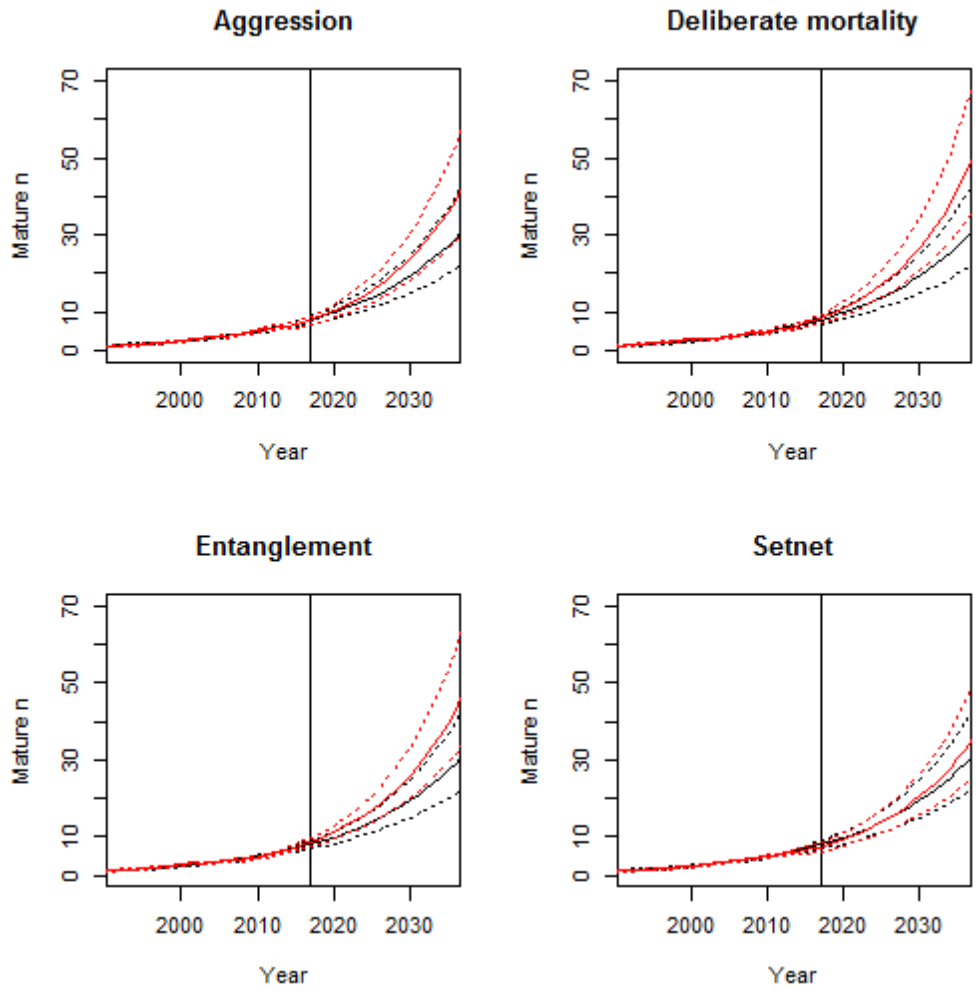


Figure 4.12: Best-estimate projections of mature n at the Otago Peninsula in the period 1990–2037 for all threat scenarios (from Roberts & Doonan 2016). Black lines are with all threats (base run); red lines are with the threat alleviated.

4.4.3.5 MULTI-SPECIES MARINE MAMMAL RISK ASSESSMENT

In 2017 the first iteration of a New Zealand Marine Mammal Risk Assessment (NZMMRA) was completed (Abraham et al. 2017), applying an adaptation of the Spatially Explicit Fisheries Risk Assessment (SEFRA) method described in Chapter 3.

The risk assessment outputs suggest that sea lions are the twelfth-highest at-risk species of marine mammal from New Zealand commercial fisheries. Fisheries risk to sea lions is attributable to a range of trawl fisheries. Estimated annual potential fishery-related deaths for sea lions by fishery group are shown in Figure 4.13.

The estimated cumulative fisheries risk score for sea lions is well estimated and ranges from approximately 0.1 to 0.2 (Figure 4.14), consistent with species-specific population modelling results suggesting that direct fisheries-related mortality is unlikely to be a primary cause of sea lion deaths (Roberts and Doonan 2016).

Because sea lions have been the subject of considerable directed data collection and research over many years, and sea lion captures occur primarily in well-observed deepwater or pelagic trawl fisheries, the results of the multi-species NZMMRA in its current form are useful to supplement but not replace the conclusions of more focused population research such as presented by Roberts and Doonan (2016) (see Section 4.4.3.4, Comprehensive quantitative risk assessment, above). An independent external review of the SEFRA method (Lonergan et al. 2017) noted that the reliability and specific applicability of the current NZMMRA is limited by its reliance on species spatial distributions derived from expert knowledge in which animal densities are assigned to discrete spatial blocks using a Delphi approach. The reviewers recommended that

the NZMMRA should be updated using more reliable species spatial distributions as these become available.

Input data layers reflecting finer-scale spatial and seasonal patterns are likely to be especially important for coastal and/or colony-associated species such as sea lions. New research utilising satellite telemetry and spatial distribution modelling to estimate spatial foraging densities of sea lions, and then applying these layers in species-specific implementations of the SEFRA method, are currently underway for Auckland Islands sea lions (PRO2017-10) and for Stewart and South Island sea lions (PMM2018-04B), with outputs expected in 2019 and 2020, respectively. Because sea lions show sex-specific movement patterns and foraging behaviours, and because male and female deaths are likely to have very different implications for the population response of harem-breeding mammals, these risk assessments may need to be sex-specific, or limited to females only (as in the population modelling of Roberts & Doonan 2016).

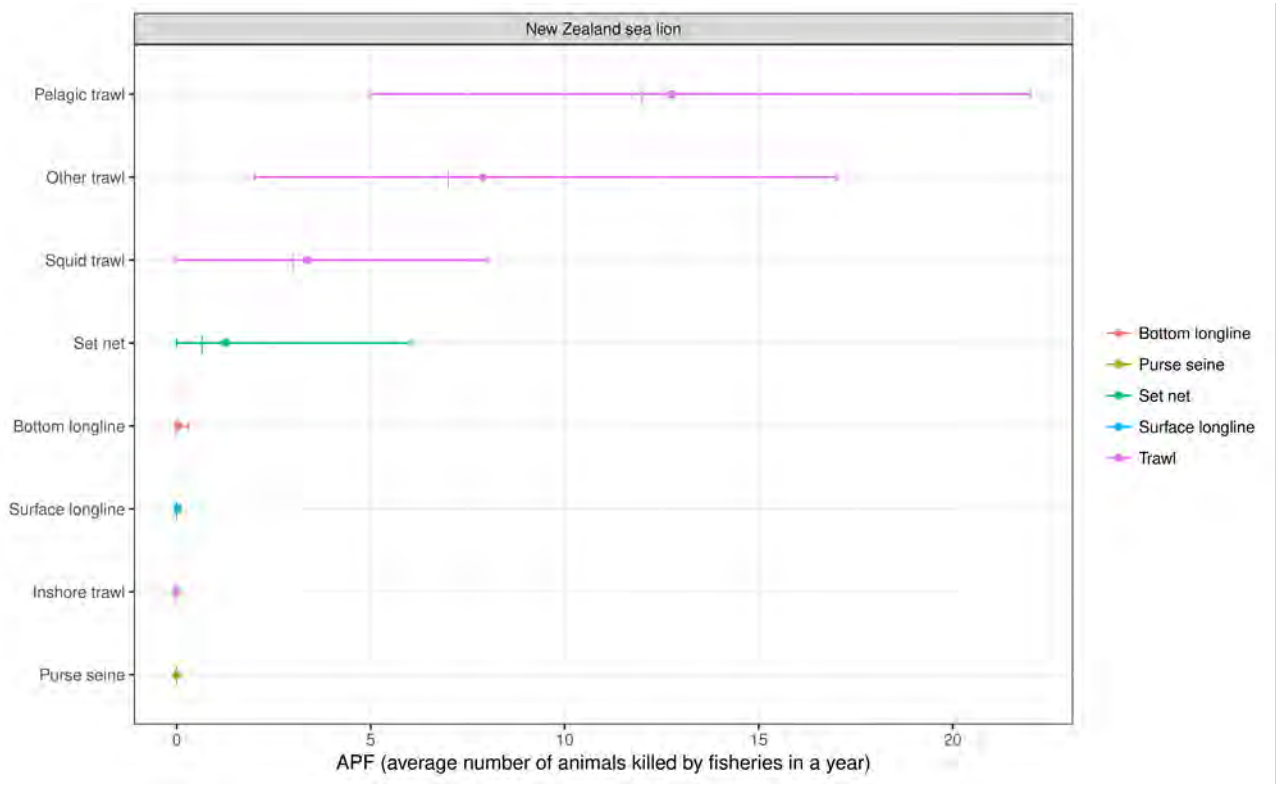


Figure 4.13: Estimated annual potential fishery-related deaths of sea lions by fishery group, as estimated by the 2017 New Zealand Marine Mammal Risk Assessment (Abraham et al. 2017). In this assessment 'pelagic trawl' includes the southern blue whiting trawl fishery, and 'other trawl' includes the scampi trawl fishery.

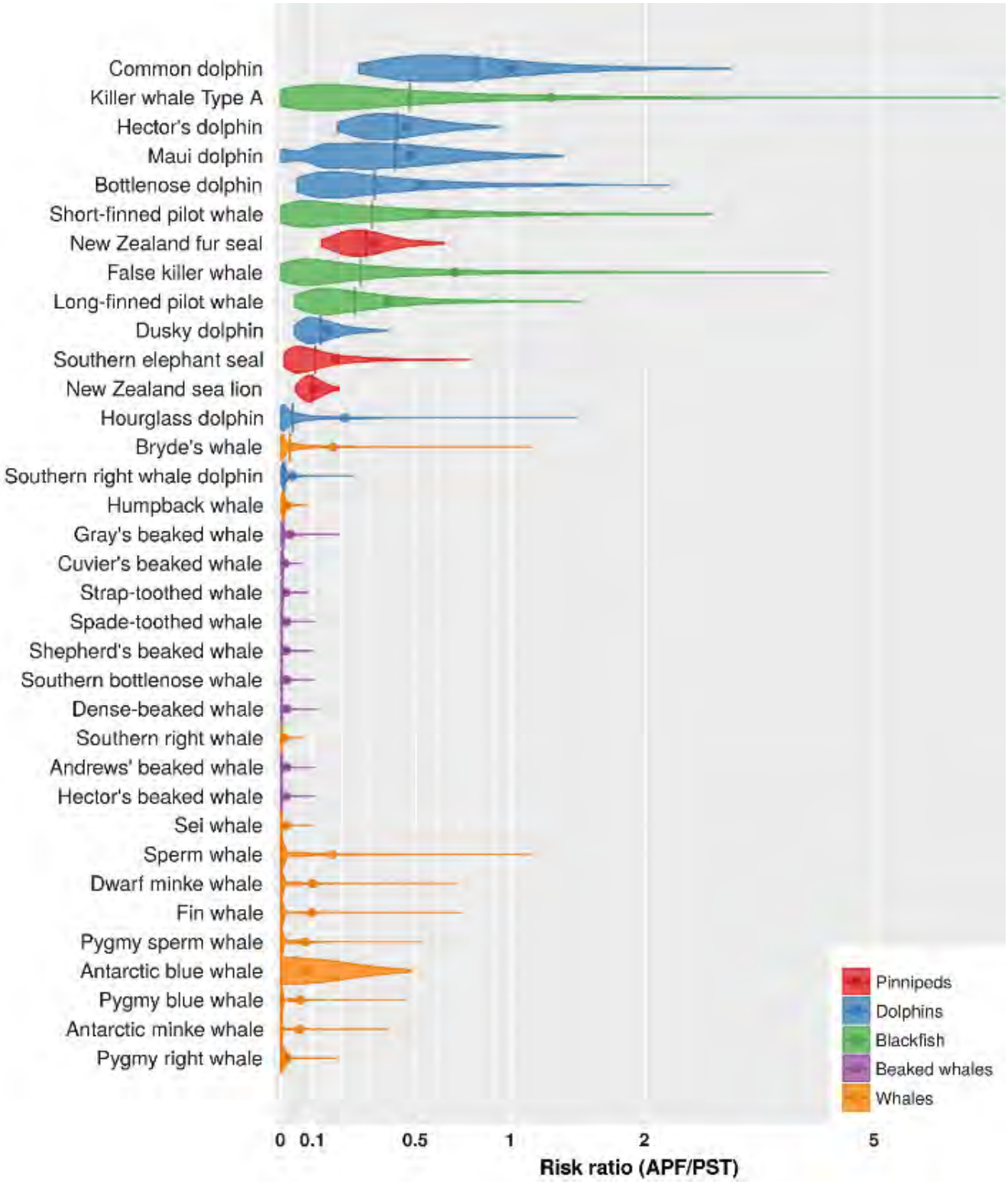


Figure 4.14: Cumulative fishery risk across all fishery groups as estimated by the 2017 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017). Species groups are colour coded.

4.4.3.6 SOURCES OF UNCERTAINTY

There are several sources of uncertainty in modelling the effects of fisheries on New Zealand sea lions at the Auckland Islands, In particular, in addition to statistical uncertainty arising from uncertain model input parameters, the population projections in Roberts & Doonan (2016) relied on the untestable structural assumption that future demographic rates would approximate historically observed rates from the past 20 years. This period included the period of steepest population decline and not the preceding period of population growth. It is likely that changes in demographic rates reflect changing environmental conditions. Because it is not possible to anticipate what environmental conditions are likely to prevail in the future, with unknown potential consequences for sea lion demographic rates, uncertainty of this nature is best addressed with model sensitivities.

The Roberts & Doonan (2016) model makes no assumptions about the current status of the Auckland Islands sea lion population relative to ecological carrying capacity. Previously a review of life-history traits such as pup mass, pup survival and female fecundity found no evidence for density dependent responses in the Auckland Islands population (Chilvers 2012b). However a number of indicators of nutritional stress have been identified during the period of population decline, including a temporal shift in diet composition to small-sized prey (Childerhouse et al. 2001, Stewart-Sinclair 2013), low pupping rate/delayed age at first pupping (Childerhouse et al. 2010a, Roberts et al. 2014a), low pup/yearling survival rate (Roberts et al. 2014a) and reduced maternal condition (Riet-Sapriza et al. 2012, Roberts & Doonan 2014) – all of which are common density dependent responses. These responses have become more apparent as the population has decreased, suggesting that changes in carrying capacity may have occurred. The underlying causes of the apparent change is unknown; and it is unknown whether similar changes can be expected in future, and on what time scales. Simulations may be useful to investigate the potential effect of environmental regime shifts affecting demographic rates on a cyclical or stochastic basis on a range of time scales.

In addition to sources of uncertainty reflected in the population model, there are other sources of uncertainty reflected in current management measures. In particular, estimated interaction rates in the SQU 6T fishery reflect past model outputs that are highly uncertain (Table 4.5; see also Thompson et al. 2013, Abraham & Berkenbusch 2017). New research will rely instead on estimating interaction rates based on the SEFRA method (PMM2017-10), and estimation of cryptic mortality multipliers so that fisheries deaths can be estimated directly as a function of observed capture rates (PMM2018-09).

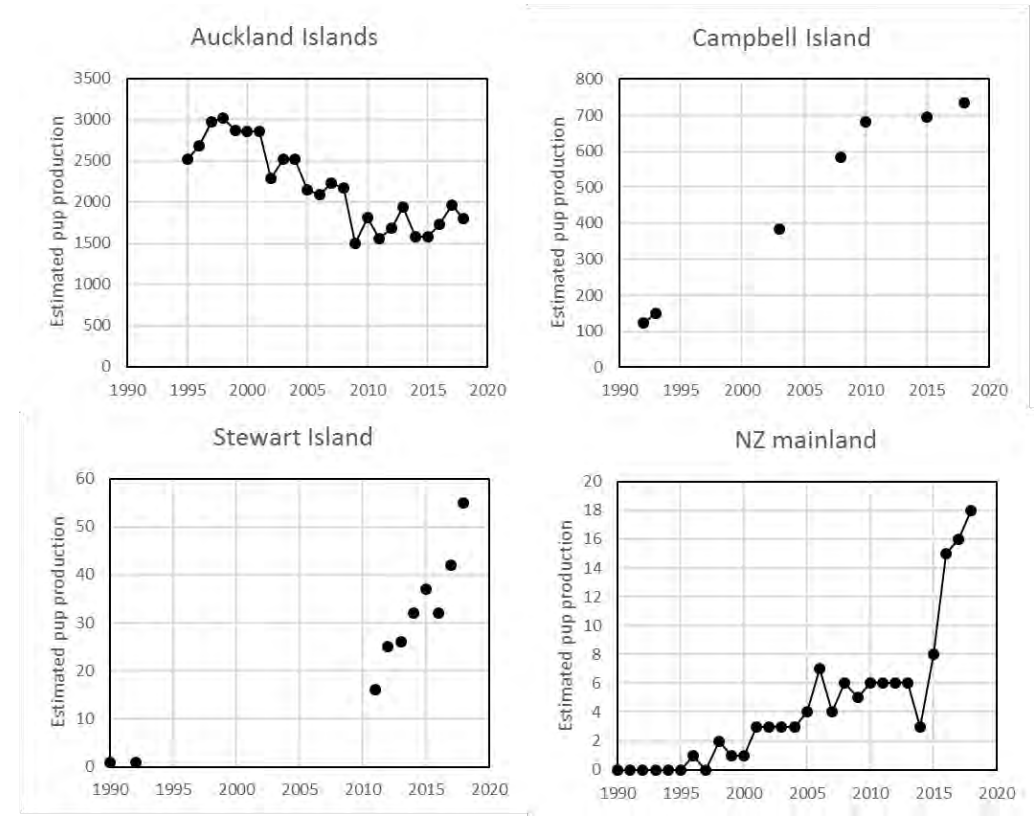
4.4.3.7 POTENTIAL INDIRECT THREATS

It is possible that indirect fisheries effects may have population-level consequences for New Zealand sea lions. Such indirect effects may include competition for food resources between various fisheries and New Zealand sea lions (Robertson & Chilvers 2011; Roberts et al. 2018). In order to determine whether resource competition is present and is having a population-level effect on New Zealand sea lions, research has sought to identify if there are resources in common for New Zealand sea lions and the various fisheries within their preferred foraging range, and to what extent those resources are limiting. Diet studies have revealed some overlap in the species consumed by New Zealand sea lions and those caught in fisheries within the range of New Zealand sea lions, particularly hoki and arrow squid (Cawthorn et al. 1985, Childerhouse et al. 2001, Meynier et al. 2009). A recent study focused on energy and amino acid content of prey, determined that the selected prey species contained all essential amino acids and were of low to medium energy levels (Meynier 2010). This study concluded that given low energy densities of prey, sea lions may be able to sustain energy requirements, but not necessarily store energy reserves and, thus, sea lions may be sensitive to factors that negatively affect trophic resources. Meynier (2010) also developed a bio-energetic model and used it to estimate the amount of prey consumed by New Zealand sea lions at 17 871 t (95% c.i.: 17 738–18 000 t) per year. This is equivalent to ~30% of the tonnage of arrow squid, and ~15% of the hoki harvested annually by the fisheries in the Subantarctic between 2000 and 2006 (Meynier 2010). Comparison of the temporal and spatial distributions of sea lion prey, sea lion foraging and of historical fishing extractions may help to identify the mechanisms whereby resource competition might occur (Bowen 2012). The effects of fishing on sea lion prey species are likely to be complicated by food web interactions. Multi-species models may help to assess the extent to which resource

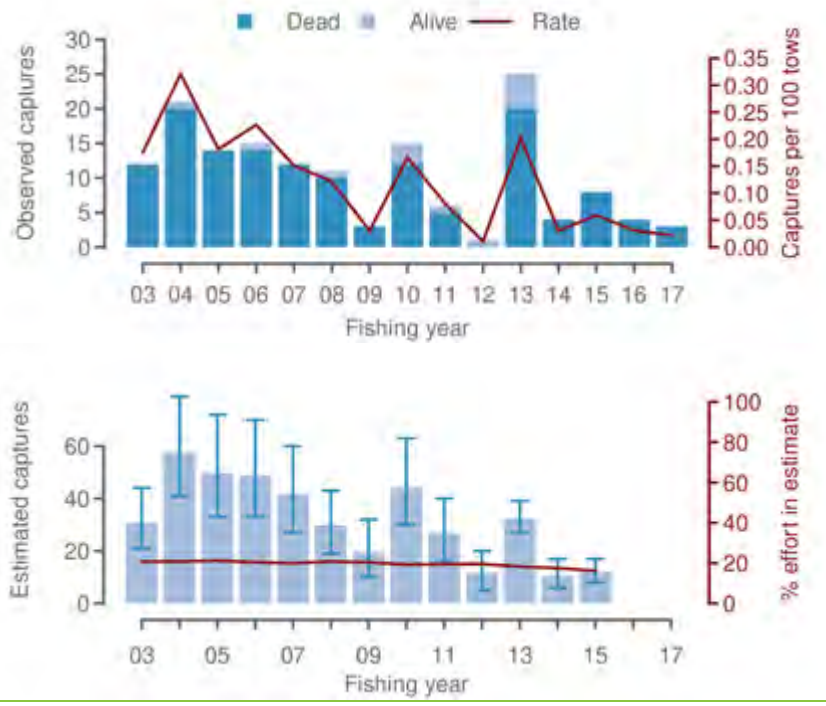
competition can impact on sea lion populations. In addition, multi-species models may provide a means for simultaneously assessing multiple drivers of sea lion population change (Robertson & Chilvers 2011), which may be a more effective approach than focusing on single factor explanations for the recent observed decline in New Zealand sea lions (Bowen 2012).

New research investigating the abundance and distribution of sea lion prey species, including via a dedicated trawl survey, suggests that the Auckland Islands sea lion sub-population has endured a protracted period of nutritional stress, and during unfavourable periods this population may have been limited by the availability of key prey (Roberts et al. in 2018). However conclusions regarding the extent to which this may reflect indirect fisheries effects are inconclusive pending a more thorough understanding of sea lion diet and foraging behaviour under different environmental conditions.

4.5 INDICATORS AND TRENDS

Population size	11,755 New Zealand sea lions including pups (immediately after pupping) (Roberts & Doonan 2016). It is estimated that there were: 1,792 pups born at the Auckland Islands in 2017-18; 734 pups born at Campbell Island in 2018 (Boren 2018); 55 pups born at Stewart Island in 2018; and 18 pups born on the Otago coast in 2018.[8]
Population trend	Estimated annual pup production at the Auckland Islands, Campbell Island, Stewart Island, and New Zealand South Island is shown below. 

[8] Jim Fyfe, Otago pup numbers 2018, https://www.doc.govt.nz/globalassets/documents/conservation/marine-and-coastal/marine-conservation-services/reports/otago-pup-numbers-2018-jim-fyfe.pdf.

Threat status	New Zealand: Nationally Critical, Criterion C[9], Range Restricted[10], in 2013[11] IUCN: Endangered, A4bd[12], in 2015[13]; to be updated in early 2019
Number of captures	4 observed captures in trawl fisheries in 2015–16[15] 3 observed captures in trawl fisheries in 2016–17[15]
Trends in observed and estimated captures	

[9] A taxon is listed as 'Nationally Critical' under criterion C if the population (irrespective of size or number of sub-populations) has a very high (rate of) ongoing or predicted decline; greater than 70% over 10 years or three generations, whichever is longer (Townsend et al. 2008).

[10] A taxon is listed as 'Range Restricted' if it is confined to specific substrates, habitats or geographic areas of less than 1000 km^2 (100 000 ha); this is assessed by taking into account the area of occupied habitat of all sub-populations (Townsend et al. 2008).

[11] Baker et al. (2016).

[12] A taxon is listed as 'Vulnerable' if it is considered to be facing a high risk of extinction in the wild. A3b refers to a reduction in population size (A), based on a reduction of 30% or more over the last 10 years or three generations (whichever is longer up to a maximum of 100 years (3); and when considering an index of abundance that is appropriate to the taxon (b; IUCN 2010).

[13] Chilvers (2015). For more information, see: http://data.dragonfly.co.nz/psc.

[16] Abraham et al. (2017).

4.6 REFERENCES

Abraham, E R (2011) Probability of Mild Traumatic Brain Injury for sea lions interacting with SLEDs. Final Research Report for Ministry of Fisheries project SRP2011-03. 21 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Abraham, E R; Berkenbusch, K (2017) Estimated captures of New Zealand fur seal, New Zealand sea lion, common dolphin, and turtles in New Zealand commercial fisheries, 1995–96 to 2014–15. *New Zealand Aquatic Environment and Biodiversity Report No. 188*. 66 p.

Abraham, E R; Berkenbusch, K; Neubauer, P; Richard, Y (2017) Assessment of the risk to New Zealand marine mammals from commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 189*. 123 p.

Abraham, E R; Thompson, F N (2011) Summary of the capture of seabirds, marine mammals, and turtles in New Zealand commercial fisheries, 1998–99 to 2008–09. Final Research Report prepared for Ministry of Fisheries project PRO2007/01. 170 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Anonymous (2009) DWG and MFish SLED Specification for SQU 6T 2010 Operation Plan (MK 3/13). Report presented to SLED WG September 2009, Wellington.

Augé, A A (2006) Terrestrial spatial ecology of female New Zealand sea lions. Unpublished Master's Thesis, University of Otago, Dunedin.

Augé, A A; Chilvers, B L; Moore, A B; Davis, L S (2011a) Foraging behaviour indicates marginal habitat for New Zealand sea lions: remnant vs recolonising. *Marine Ecology Progress Series 432*: 247–256.

Augé, A A; Chilvers, B L; Davis, L S; Moore, A B (2011b) In the shallow end: diving behaviour of recolonising female New Zealand sea lions (*Phocarctos hookeri*) around the Otago Peninsula. *Canadian Journal of Zoology 89*: 1195–1205.

Augé, A A; Chilvers, B L; Moore, A B; Davis, L S (2014) Importance of studying foraging site fidelity for spatial conservation measures in a mobile predator. *Animal Conservation 17 (1)*: 61–71. doi:10.1111/acv.12056.

Augé, A A; Lalas, C; Davis, L S; Chilvers, B L (2012) Autumn diet of recolonizing female New Zealand sea lions based at Otago Peninsula, South Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research 46*: 97–110.

Baker, A J (1999) Unusual mortality of the New Zealand sea lion, *Phocarctos hookeri*, Auckland Islands, January–February 1998: A report of a workshop held 8–9 June 1998, Wellington, and a contingency plan for future events. pp 29–33. Department of Conservation, Wellington, New Zealand.

Baker, B; Jensz, K; Chilvers, L (2013) DRAFT Final Report for the Department of Conservation, Conservation Services Programme. Aerial survey of New Zealand sea lions – Auckland Islands 2012/2013. Latitude 42 Environmental Consultants Pty Ltd. 11 p.

Baker, C S; Chilvers, B L; Constantine, R; DuFresne, S; Mattlin, R H; van Helden, A; Hitchmough, R (2010) Conservation status of New Zealand marine mammals (suborders Cetacea and Pinnipedia), 2009. *New Zealand Journal of Marine and Freshwater Research 44*: 101–115.

Baker, C S; Chilvers, B L; Childerhouse, S; Constantine, R; Currey, R; Mattlin, R; van Helden, A; Hitchmough, R; Rolfe, J (2016) Conservation status of New Zealand marine mammals, 2013. *New Zealand Threat Classification Series 14*. Department of Conservation, Wellington. 18 p.

Boren, L (2018) Campbell Island NZ sea lion programme 2017/2018. CSP NZSL TWG May 2018.

Bowen, W D (2012) A review of evidence for indirect effects of commercial fishing on New Zealand sea lions (*Phocarctos hookeri*) breeding on the Auckland Islands. Final Report for Contract POP 2010/01, Objectives 4 and 5. Department of Conservation, Wellington. 41 p.

Bradshaw, C J A; Lalas, C; McConkey, S (1998) New Zealand sea lion predation on New Zealand fur seals. *New Zealand Journal of Marine and Freshwater Research 32*: 101–104.

Breen, P A (2008) Sea lion data for use in the population model for Project IPA200609. Final Research Report for Ministry of Fisheries. 18 p. plus Confidential Appendix of 6 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Breen, P A; Hilborn, R; Maunder, M N; Kim, S W (2003) Effects of alternative control rules on the conflict between a fishery and a threatened sea lion (*Phocarctos hookeri*). *Canadian Journal of Fisheries and Aquatic Sciences 60*: 527–541.

Breen, P A; Fu, D; Gilbert, D J (2016) Sea lion population modelling and management procedure evaluations. *New Zealand Aquatic Environment and Biodiversity Report No. 175*. 89 p.

Breen, P A; Kim, S W (2006a) Exploring alternative management procedures for controlling bycatch of Hooker's sea lions in the SQU 6T squid fishery. Final Research Report for Ministry of Fisheries project M0F2002/03L, Objective 3. Revision 5. 88 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Breen, P A; Kim, S W (2006b) An integrated Bayesian evaluation of Hooker's sea lion bycatch limits. *In*: Trites, A; Atkinson, S; DeMaster, D; Fritz, L; Gelatt, T; Rea, L; Wynne, K (Eds.), Sea lions of the world. pp 471–493. Alaska Sea Grant College Program, University of Alaska Fairbanks.

Castinel, A; Duignan, P J; Pomroy, W E; López-Villalobos, N; Gibbs, N J; Chilvers, B L; Wilkinson, I S (2007) Neonatal mortality in New Zealand sea lions (*Phocarctos hookeri*) at Sandy Bay, Enderby Island, Auckland Islands from 1998 to 2005. *Journal of Wildlife Diseases 43*: 461–474.

Cawthorn, M W; Crawley, M C; Mattlin, R H; Wilson, G J (1985) Research on pinnipeds in New Zealand. Wildlife Research Liaison Group Report No. 7. Wellington, New Zealand.

Childerhouse, S J; Amey, J; Hamer, D; McCrone, A (2013) Final Report for CSP project 4426: New Zealand sea lion ground component 2012/2013. Unpublished Report to the Conservation Services Programme, Department of Conservation, Wellington, New Zealand. Version 1.0. 20 p.

Childerhouse, S J; Dawson, S M; Fletcher, D J; Slooten, E; Chilvers, B L (2010a) Growth and reproduction of female New Zealand sea lions. *Journal of Mammalogy 91*: 165–176.

Childerhouse, S J; Dawson, S M; Slooten, E; Fletcher, D J; Wilkinson, I S (2010b) Age distribution of lactating New Zealand sea lions: interannual and intersite variation. *Marine Mammal Science 26*: 123–139.

Childerhouse, S J; Dix, B; Gales, N J (2001) Diet of New Zealand sea lions (*Phocarctos hookeri*) at the Auckland Islands. *Wildlife Research 28*: 291–298.

Childerhouse, S J; Fyfe, J; Muller, C; Gonzalez-Argandona, K; French, R; Roe, W (2015a) Final report: New Zealand sea lion research at Campbell Island-Motu Ihupuku, 2014/15. Blue Planet Marine.

Childerhouse, S; Gales, N (1998) The historic distribution and abundance of the New Zealand sea lion *Phocarctos hookeri*. *New Zealand Journal of Zoology 25*: 1–16.

Childerhouse, S J; Gales, N J (2001) Fostering behavior in New Zealand sea lions *Phocarctos hookeri*. *New Zealand Journal of Zoology 28*: 189–195.

Childerhouse, S J; Gibbs, N; McAlister, G; McConkey, S; McConnell, H; McNally, N; Sutherland, D (2005) Distribution, abundance and growth of New Zealand sea lion *Phocarctos hookeri* pups on Campbell Island. *New Zealand Journal of Marine and Freshwater Research 39*: 889–898.

Childerhouse, S J; Hamer, D; Maloney, A; Michael, S; Donnelly, D; Schmitt, N (2014) Final report for CSP project 4522: New Zealand sea lion ground component 2013/14. Unpublished report to the Conservation Services Programme, Department of Conservation, Wellington, New Zealand. Version 1.1. 31 p.

Childerhouse, S J; Micheal, S; Adams, L; Bunrs, T; Cockburn, S; Hamer, D; Maloney, A; Pugsley, C (2015b) Final report: New Zealand sea lion research at the Auckland Islands 2014/15. Report produced by Blue Planet Marine for the New Zealand Department of Conservation.

Childerhouse, S J; Muller, C; Burns, T; French, R; Kay, E (2016) Final report for CSP project: New Zealand sea lion ground component 2015/16. Report held by the Department of Conservation. 52 p.

Chilvers, B L (2008) New Zealand sea lions *Phocarctos hookeri* and squid trawl fisheries: bycatch problems and management options. *Endangered Species Research 5*: 193–204.

Chilvers, B L (2009) Foraging locations of a decreasing colony of New Zealand sea lions (*Phocarctos hookeri*). *New Zealand Journal of Ecology 33*: 106–113.

Chilvers, B L (2012a) Research to assess the demographic parameters of New Zealand sea lions, Auckland Islands. Final Research Report for Contract POP 2011/01. Department of Conservation, Wellington. 11 p.

Chilvers, B L (2012b) Using life-history traits of New Zealand sea lions, Auckland Islands to clarify potential causes of decline. *Journal of Zoology 287*: 240–249.

Chilvers, B L (2014) Report of the survey to assess distribution and pup production of New Zealand sea lions on Stewart Island 2014. Report prepared for the Department of Conservation.

Chilvers, B L (2015) *Phocarctos hookeri*. The IUCN Red List of Threatened Species 2015: e.T17026A1306343. doi:10.2305/IUCN.UK.2015-2.RLTS.T17026A1306343.en.

Chilvers, B L (2017a) Stable isotope signatures of whisker and blood serum confirm foraging strategies for females New Zealand sea lions derived from telemetry. *Canadian Journal of Zoology, 95*: 955-963.

Chilvers, B L (2017b) Preliminary assessment of the foraging behaviour and population dynamics of a cryptic population of the endangered New Zealand sea lion. *New Zealand Journal of Ecology, 42*: 48-57.

Chilvers, B L; Amey, J M; Huckstadt, L A; Costa, D P (2011) Investigating foraging utilization distribution of female New Zealand sea lions, Auckland Islands. *Polar Biology 34*: 565–574.

Chilvers, B L; Childerhouse, S J; Gales, N J (2013) Winter foraging behaviour of lactating New Zealand sea lions (*Phocarctos hookeri*). *New Zealand Journal of Marine and Freshwater Research 47*: 125–138.

Chilvers, B L; Robertson, B C; Wilkinson, I S; Duignan, P J; Gemmell, N J (2005a) Male harassment of female New Zealand sea lions: mortality, injury and harassment avoidance. *Canadian Journal of Zoology 83*: 642–648.

Chilvers, B L; Wilkinson, I S (2008) Philopatry and site fidelity of New Zealand sea lions (*Phocarctos hookeri*). *Wildlife Research 35*: 463–470.

Chilvers, B L; Wilkinson, I S (2009) Diverse foraging strategies in lactating New Zealand sea lions. *Marine Ecology Progress Series 378*: 299–308.

Chilvers, B L; Wilkinson, I S (2011) Research to assess the demographic parameters of New Zealand sea lions, Auckland Islands. Draft Final Report for Contract POP 2010/01. Department of Conservation, Wellington. 20 p.

Chilvers, B L; Wilkinson, I S; Childerhouse, S J (2007) New Zealand sea lion, *Phocarctos hookeri*, pup production–1995 to 2006. *New Zealand Journal of Marine and Freshwater Research 41*: 205–213.

Chilvers, B L; Wilkinson, I S; Duignan, P J; Gemmell, N J (2005b) Summer foraging areas for lactating New Zealand sea lions *Phocarctos hookeri*. *Marine Ecology Progress Series 304*: 235–247.

Chilvers, B L; Wilkinson, I S; Duignan, P J; Gemmell, N J (2006) Diving to extremes: are New Zealand sea lions pushing their limits in a marginal habitat? *Journal of Zoology 269*: 233–240.

Chilvers, B L; Wilkinson, I S; Mackenzie, D I (2010) Predicting life-history traits for female New Zealand sea lions, *Phocarctos hookeri*: integrating short-term mark–recapture data and population modelling. *Journal of Agricultural, Biological, and Environmental Statistics 15*: 259–278.

Clement & Associates Ltd. (2007) Squid trawl fleet sea lion escape device audit. A report commissioned by the Department of Conservation for project CSP MIT2004/05. Department of Conservation, Wellington. 19 p.

Collins, C J; Rawlence, N J; Prost, S; Anderson, C N K; Knapp, M; Scofield, P; Robertson, B; Smith, I; Matisoo-Smith, E A; Chilvers, B L; Waters, J M (2014a) Extinction and recolonization of coastal megafauna following human arrival in New Zealand. *Proceedings of the Royal Society B 281 (1786)*: 20140097.

Collins, C J; Rawlence, N J; Worthy, T H; Scofield, R P; Tennyson, A J D; Smith, I; Knapp, M; Waters, J M (2014b) Pre-human New

Zealand sea lion (*Phocarctos hookeri*) rookeries on mainland New Zealand. *Journal of the Royal Society of New Zealand 44*: 1–16.

Costa, D P; Gales, N J (2000) Foraging energetic and diving behaviour of lactating New Zealand sea lions, *Phocarctos hookeri*. *Journal of Experimental Biology 203*: 3655–3665.

Crocker, D E; Gales, N J; Costa, D P (2001) Swimming speed and foraging strategies of New Zealand sea lions, *Phocarctos hookeri*. *Journal of Zoology 254*: 267–277.

Debski, I; Walker, N (2016) A summary of the risk assessment of threats to New Zealand sea lions. *MPI Information Paper No 2016/03*. 36 p.

Department of Conservation (2007) Draft Population Management Plan for New Zealand Sea Lion. Draft Document for Public Consultation, August 2007. Department of Conservation, Wellington. Retrieved from http://www.doc.govt.nz/getting-involved/consultations/closed/nz-sea-lion-management.

Department of Conservation (2009) New Zealand sea lion species management plan: 2009–2014. Department of Conservation, Wellington. 31 p.

Department of Conservation and Ministry for Primary Industries (2017) New Zealand sea lion/rāpoka Threat Management Plan: Joint paper of the Department of Conservation and the Ministry for Primary Industries. 17 p.

Dickie, G (1999) Population dynamics of New Zealand fur seals (*Arctocephalus forsteri*) and New Zealand sea lions (*Phocarctos hookeri*). Master's Thesis, University of Otago, Dunedin, New Zealand. 117 p.

Fish, F (2008) Streamlining. *In*: Perrin, W F; Wursig, B; Thewissen, J G M (Eds.) Encyclopedia of Marine Mammals. pp 1123–1126. Academic Press.

Gales, N J (1995) New Zealand (Hooker's) sea lion recovery plan. *Threatened Species Recovery Plan Series 17*. Department of Conservation, Wellington.

Gales, N J; Fletcher, D J (1999) Abundance, distribution and status of the New Zealand sea lion, *Phocarctos hookeri*. *Wildlife Research 26*: 35–52.

Gales, N J; Mattlin, R H (1997) Summer diving behaviour of lactating female New Zealand sea lions. *Canadian Journal of Zoology 75*: 1695–1706.

Gilbert, D J; Chilvers, B L (2008) Final report on New Zealand sea lion pupping rate. POP2006-01, Objective 3: Analysis from sea lion database to estimate pupping rate and associated parameters. NIWA Report WLG2008-77. 26 p.

Gill, B J (1998) Prehistoric breeding sites of New Zealand sea lions (*Phocarctos hookeri*, *Carnivora: Otariidae*) at North Cape. *Records of the Auckland Museum 35*: 55–64.

Graham, B S (in prep) Stable isotope analysis of sea lion teeth. Draft final research report for Ministry for Primary Industries project SEA2014-12.

Graham, B S; Koch, P L; Newsome, S D; McMahon, K W; Aurioles, D (2010) Chapter 14: Using isoscapes to trace the movement and foraging behaviour of top predators in oceanic predators. *In*: West, J B, et al. (Eds.), Isoscapes: Understanding Movement, Pattern, and Process on Earth through Isotope Mapping. doi:10.1007/978-90-481-3354-3_14.

Hankel, W; Witte, M; Miersch, L; Brede, M; Oeffner, J; Michael, M; Hankel, F; Leder, A; Dehnhardt, G (2010) Harbor seal vibrissa morphology suppresses vortex-induced vibrations. *Journal of Experimental Biology 213*: 2665–2672.

IUCN (2010) IUCN Red List of Threatened Species. Version 2010.4. Retrieved from http://www.iucnredlist.org.

Katsanevakis, S (2008) Marine debris, a growing problem: Sources, distribution, composition, and impacts. *In*: Hofer, T N (Ed.), Marine Pollution: New Research. pp 53–100. Nova Science Publishers, New York.

Lalas, C (1997) Prey of Hooker's sea lions *Phocarctos hookeri* based at Otago Peninsula, New Zealand. *In*: Hindell, M; Kemper, C (Eds.), Marine Mammal Research in the Southern Hemisphere, Status, Ecology and Medicine. Volume 1. pp 130–136. Surrey Beatty and Sons, Chipping Norton.

Lalas, C; Bradshaw, C J A (2003) Expectations for population growth at new breeding locations for the vulnerable New Zealand sea lion using a simulation model. *Biological Conservation 114*: 67–78.

Lalas, C; McConnell, H M; Meynier, L (2014) Estimating size of opalfish from otoliths: implications for analyses of New Zealand sea lion diet. *New Zealand Journal of Marine and Freshwater Research 48*: 1–14.

Lalas, C; Ratz, H; McEwan, K; McConkey, S D (2007) Predation by New Zealand sea lions (*Phocarctos hookeri*) as a threat to the viability of yellow-eyed penguins (*Megadyptes antipodes*) at Otago Peninsula, New Zealand. *Biological Conservation 135*: 235–246.

Lalas, C; Webster, T (2014) Contrast in the importance of arrow squid as prey of male New Zealand sea lions and New Zealand fur seals at The Snares, subantarctic New Zealand. *Marine Biology 161*: 631–643.

Laws, E A; Popp, B N; Bidigare, R R; Kennicutt, M C; Macko, S A (1995) Dependence of phytoplankton carbon isotopic composition on growth rate and $[CO2]_{aq}$: Theoretical considerations and experimental results. *Geochimica et Cosmochimica Acta 59 (6)*: 1131–1138.

Leung, E S; Augé, A A; Chilvers, B L; Moore, A B; Robertson, B C (2013a) Foraging behaviour of juvenile female New Zealand sea lions (*Phocarctos hookeri*) in contrasting environments. *PLoS ONE 8 (5)*: e62728.

Leung, E S; Chilvers, B L; Moore, A B; Nakagawa, S; Robertson, B C (2012) Sexual segregation in juvenile New Zealand sea lion foraging ranges: implications for intraspecific competition, population dynamics and conservation. *PLoS ONE 7 (9)*: e45389.

Leung, E S; Chilvers, B L; Moore, A B; Robertson, B C (2013b) Mass and bathymetry influences on the foraging behaviour of dependent yearling New Zealand sea lions (*Phocarctos hookeri*). *New Zealand Journal of Marine and Freshwater Research 47*: 38–50.

Leung, E S; Chilvers, B L; Moore, A B; Robertson B C (2014a) Do yearling New Zealand sea lions (*Phocarctos hookeri*) learn foraging behaviour from their mothers? *Marine Mammal Science 30*: 1220–1228.

Leung, E S; Chilvers, B L; Nakagawa, S; Robertson B C (2014b) Size and experience matter: diving behaviour of juvenile New Zealand sea lions (*Phocarctos hookeri*). *Polar Biology 37*: 15–26.

Lonergan, M E; Phillips, R A; Thomson, R B; Zhou, S (2017) Independent review of New Zealand's Spatially Explicit Fisheries Risk Assessment approach – 2017. *New Zealand Fisheries Science Review 2017/2*. 36 p.

Lyle, J M (2011) SRP2010-03: Fur seal interactions with SED excluder device. Final Research Report for Ministry of Fisheries. 20 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Lyle, J M; Willcox, S T (2008) Dolphin and seal interactions with mid-water trawling in the Commonwealth small pelagic fishery, including an assessment of bycatch mitigation. Final Report for project R05/0996. Australian Fisheries Management Authority, National Library of Australia. 39 p.

McConkey, S; McConnell, H; Lalas, C; Heinrich, S; Ludmerer, A; McNally, N; Parker, E; Borofsky, C; Schimanski, K; McIntosh, M (2002) A northward spread in the breeding distribution of the New Zealand sea lion. *Australian Mammalogy 24*: 97–106.

MacKenzie, D I (2011) Estimation of demographic parameters for New Zealand sea lions breeding on the Auckland Islands. Final Report for Contract POP 2011/01, Objective 3. Department of Conservation, Wellington. 88 p.

McNally, N; Heinrich, S; Childerhouse, S J (2001) Distribution and breeding of New Zealand sea lions *Phocarctos hookeri* on Campbell Island. *New Zealand Journal of Zoology 28*: 79–87.

Maloney, A; Chilvers, B L; Haley, M; Muller, C G; Roe, W D; Debski, I (2009) Distribution, pup production and mortality of New Zealand sea lion (*Phocarctos hookeri*) on Campbell Island / Motu Ihupuku, 2008. *New Zealand Journal of Ecology 33*: 97–105.

Maloney, A; Chilvers, B L; Muller, C G; Haley, M (2012) Increasing pup production of New Zealand sea lions at Campbell Island/Motu Ihupuku: can it continue? *New Zealand Journal of Zoology 39*: 19–29.

Marshall, C D (2008) Feeding Morphology. *In*: Perrin, W F; Wursig, B; Thewissen, J G M (Eds.), Encyclopedia of Marine Mammals. pp 406–413. Academic Press.

Marlow, B J (1975) The comparative behaviour of the Australasian sea lions *Neophoca cinerea* and *Phocarctos hookeri*. *Mammalia 39*: 159–230.

Mattlin, R H (2004) QMA SQU 6T New Zealand sea lion incidental catch and necropsy data for the fishing years 2000–01, 2001–02 and 2002–03. Report prepared for the Ministry of Fisheries. 21 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Meynier, L (2010) New Zealand sea lion bio-energetic modelling: Final Report for project IPA2009-09. Presented to AEWG 17 May 2011, Wellington, New Zealand. 34 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Meynier, L; Mackenzie, D D S; Duignan, P J; Chilvers, B L; Morel, P C H (2009) Variability in the diet of New Zealand sea lion (*Phocarctos hookeri*) at the Auckland Islands, New Zealand. *Marine Mammal Science 25*: 302–326.

Meynier, L; Morel, P C H; Chilvers, B L; Mackenzie, D D S (2014) Foraging diversity in lactating New Zealand sea lions: insights from qualitative and quantitative fatty acid analysis. *Canadian Journal of Fisheries and Aquatic Sciences 71*: 984–991.

Meynier, L; Morel, P C H; Chilvers, B L; Mackenzie, D D S; MacGibbon, A; Duignan, P J (2008) Temporal and sex differences in the blubber fatty acid profiles of the New Zealand sea lion *Phocarctos hookeri*. *Marine Ecology Progress Series 366*: 271–278.

Middleton, D A J; Banks, D A (2008) Analysis of underwater video footage of SELDs from the 2008 squid fishery. Deepwater Group Ltd.

Ministry of Fisheries (2010) National Fisheries Plan for Deepwater and Middle-depth Fisheries. Ministry of Fisheries, Wellington, New Zealand. 59 p. Retrieved from http://www.fish.govt.nz/en-nz/Consultations/Archive/2010/National+Fisheries+Plan+for+Deepwater+and+Middle-Depth+Fisheries/default.htm.

Ministry of Fisheries (2011) Report from the Fisheries Assessment Plenary, May 2011: stock assessments and yield estimates. Ministry of Fisheries, Wellington, New Zealand. 1178 p.

Moore, J E; Merrick, R (Eds.) (2011) Guidelines for Assessing Marine Mammal Stocks: Report of the GAMMS III Workshop, 15–18 February 2011, La Jolla, California. Dept. of Commerce, NOAA Technical Memorandum NMFS-OPR-47.

Moore, J M; Wallace, B; Lewison, R L; Zydelis, R; Cox, T; Crowder, L (2009) A review of marine mammal, sea turtle and seabird bycatch in USA fisheries and the role of policy in shaping management. *Marine Policy 33*: 435–451.

Moore, P J; Charteris, M; Larsen, E J (2008) Notes on New Zealand mammals 8. Predation on nesting southern royal albatrosses *Diomedea epomophora* by a New Zealand sea lion *Phocarctos hookeri*. *New Zealand Journal of Zoology 35*: 201–204.

Moore, P J; Moffat, R D (1992) Predation of yellow-eyed penguin by Hooker's sealion. *Notornis 39*: 68–69.

Morrison, K W; Armstrong, D P; Battley, P F; Jamieson, S E; Thompson, D R (2017) Predation by New Zealand sea lions and Brown Skuas is causing the continued decline of an Eastern Rockhopper Penguin colony on Campbell Island. *Polar Biology 40*: 735–751.

Nagaoka, L (2001) Using Diversity Indices to Measure Changes in Prey Choice at the Shag River Mouth Site, Southern New Zealand. *International Journal of Osteoarchaeology 11*: 101–111.

Nagaoka, L (2006) Prehistoric seal carcass exploitation at the Shag Mouth site, New Zealand. *Journal of Archaeological Science 33*: 1474–1481.

Ponte, G; van den Berg, A; Anderson, R W G (2010) Impact characteristics of the New Zealand Fisheries sea lion exclusion device stainless steel grid. Final Research Report for Ministry of Fisheries project IPA2009-06. 24 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Ponte, G; van den Berg, A; Anderson, R W G (2011) Further analysis of the impact characteristics of the New Zealand Fisheries sea lion exclusion device stainless steel grid. Final Research Report for Ministry of Fisheries project SRP2010-05. 36 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Rawlence, N J, et al. (2016) Human-mediated extirpation of the unique Chatham Islands sea lion and implications for the conservation management of remaining New Zealand sea lion populations. *Molecular Ecology 25 (16)*: 3950–3961. doi:10.1111/mec.13726.

Ray, G C (1963) Locomotion in pinnipeds. *Natural History 72*: 10–21.

Read, A J; Drinker, P; Northridge, S (2006) Bycatch of marine mammals in U.S. and global fisheries. *Conservation Biology 20*: 163–169.

Richard, Y; Abraham, E R; Filippi, D (2011) Assessment of the risk to seabird populations from New Zealand commercial fisheries. Final Research Report for projects IPA2009-19 and IPA2009-20. 66 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Riet-Sapriza, F G (2007) Milk composition of the New Zealand sea lion and factors that influence it. PhD Thesis, Massey University, Palmerston North, New Zealand.

Riet-Sapriza, F G; Duignan, P J; Chilvers, B L; Wilkinson, I S; Lopez-Villalobos, N; Mackenzie, D D S; MacGibbon, A; Costa, D P; Gales, N (2012) Interannual and individual variation in milk composition of New Zealand sea lions (*Phocarctos hookeri*). *Journal of Mammalogy 93*: 1006–1016.

Roberts, J (2015) Review of threats to the recovery of New Zealand sea lions and other otariid species. Report prepared for the Department of Conservation and Ministry for Primary Industries. 29 p.

Roberts, J (2017a) Review of potential NZ sea lion interactions with aquaculture at Port Pegasus/Pikihatiti. Report produced for and held by the Department of Conservation, Wellington. NIWA Client Report Number 2017264WN. 24 p.

Roberts, J (2017b) Supplemental sea lion population modelling to support an updated Squid Trawl Fishery Operational Plan. Draft final research report for Ministry for Primary Industries project SEA201624. 16 p.

Roberts, J; Doonan, I (2014) New Zealand sea lions – demographic assessment of the cause of decline at the Auckland Islands (POP2012-02): demographic model options – correlative assessment. Report for the Department of Conservation.

Roberts, J; Doonan, I (2016) Quantitative Risk Assessment of Threats to New Zealand Sea Lions. *New Zealand Aquatic Environment and Biodiversity Report No. 166*. 111 p.

Roberts, J; Fu, D; Doonan, I; Francis, R I C C (2014a) New Zealand sea lions – demographic assessment of the cause of decline at the Auckland Islands (POP2012-02): demographic model options – demographic assessment. Report for the Department of Conservation.

Roberts, J; O'Driscoll, R L; Hart, A; Graham, B (in prepSurvey of New Zealand sea lion prey at the Auckland Islands and Stewart Snares Shelf. New Zealand Aquatic Environment and Biodiversity Report No. 204. 64 p.

Roberts, J; Roux, M-J; Ladroit, Y (2014b) PBR assessment for the Campbell Island sub-population of New Zealand sea lions. NIWA Client Report WLG2014-8, prepared for the Deepwater Group.

Robertson, B C; Chilvers, B L (2011) The population decline of the New Zealand sea lion *Phocarctos hookeri*: a review of possible causes. *Mammal Review 41*: 253–275.

Robertson, B C; Chilvers, B L; Duignan, P J; Wilkinson, I S; Gemmel, N J (2006) Dispersal of breeding adult male *Phocarctos hookeri*: implications for disease transmission, population management and species recovery. *Biological Conservation 127*: 227–236.

Roe, W D (2010) External review of New Zealand sea lion bycatch necropsy data and methods. Report prepared for the Ministry of Fisheries, Wellington. 8 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Roe, W D (2011) A study of brain injury in New Zealand sea lion pups. PhD Thesis, Massey University, Palmerston North, New Zealand.

Roe, W D; Meynier, L (2010) Review of Necropsy Records for Bycaught New Zealand sea lions (*Phocarctos hookeri*), 2000–2008. Report for Ministry of Fisheries project PRO2008-03. 46 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Roe, W D; Roberts, J; Childerhouse, S (2014) Discussion paper on New Zealand pup mortality: causes and mitigation. Unpublished paper to Department of Conservation. 10 p. Retrieved from http://www.doc.govt.nz/Documents/conservation/marine-and-coastal/marine-conservation-services/meetings/sea-lion-pup-mortality-discussion-paper.pdf.

Schell D M; Barnett B A; Vinette K A (1998) Carbon and nitrogen isotope ratios in zooplankton of the Bering, Chukchi and Beaufort seas. *Marine Ecology Progress Series 162*: 11–23.

Smith, I W G (1989) Maori Impact on the Marine Magafauna: Pre-European Distributions of New Zealand Sea Mammals. *In*: Sutton, D G (Ed.), Saying so doesn't make it so. Papers in Honour of B. Foss Leach. *New Zealand Archaeological Association Monograph 17*: 76–108.

Smith, I W G (2011) Estimating the magnitude of pre-European Maori marine harvest in two New Zealand study areas. *New Zealand Aquatic Environment and Biodiversity Report No. 82*. 71 p.

Smith, M H; Baird, S J (2005) Factors that may influence the level of incidental mortality of New Zealand sea lions (*Phocarctos hookeri*) in the squid (*Nototodarus* spp.) trawl fishery in SQU 6T. *New Zealand Fisheries Assessment Report 2005/20*. 35 p.

Smith, M H; Baird, S J (2007a) Estimation of incidental captures of New Zealand sea lions (*Phocarctos hookeri*) in New Zealand fisheries in 2003–04, with particular reference to the SQU 6T squid trawl fishery. *New Zealand Fisheries Assessment Report 2007/7*. 32 p.

Smith, M H; Baird, S J (2007b) Estimation of incidental captures of New Zealand sea lions (*Phocarctos hookeri*) in New Zealand fisheries in 2004–05, with particular reference to the SQU 6T squid trawl fishery. *New Zealand Aquatic Environment and Biodiversity Report No. 12*. 31 p.

Stewart-Sinclair, P (2013) The role of long-term diet changes in the decline of the New Zealand sea lion population. Master's Thesis, Massey University, Palmerston North, New Zealand.

Thompson, F N; Abraham, E R (2010) Estimation of the capture of New Zealand sea lions (*Phocarctos hookeri*) in trawl fisheries, from 1995–96 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 66*. 25 p.

Thompson, F N; Abraham, E R; Berkenbusch, K (2011) Marine mammal bycatch in New Zealand trawl fisheries, 1995–96 to 2009–10.

Final Research Report for project PRO2010-01. 80 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Thompson, F N; Abraham, E R; Berkenbusch, K (2012) Marine mammal bycatch in New Zealand trawl fisheries, 1995–96 to 2010–11. Final Research Report for project PRO2010-01. 90 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Thompson, F N; Berkenbusch, K; Abraham, E R (2013) Marine mammal bycatch in New Zealand trawl fisheries, 1995–96 to 2011–12. *New Zealand Aquatic Environment and Biodiversity Report No. 105*. 76 p.

Thompson, F N; Berkenbusch, K; Abraham, E R (2016) Incidental capture of marine mammals in New Zealand trawl fisheries, 1995–96 to 2011–12. *New Zealand Aquatic Environment and Biodiversity Report No. 167*. 78 p.

Townsend, A J; de Lange, P J; Duffy, C A J; Miskelly, C M; Molloy, J; Norton, D (2008) New Zealand Threat Classification System Manual. Department of Conservation, Wellington, New Zealand.

Tuck, I D (2009) Characterisation of scampi fisheries and the examination of catch at length and spatial distribution of scampi in SCI 1, 2, 3, 4A and 6A. *New Zealand Fisheries Assessment Report 2009/27*. 102 p.

Wade, P R (1998) Calculating limits to the allowable human caused mortality of cetaceans and pinnipeds. *Marine Mammal Science 14*: 1–37.

Walker, G E; Ling, J K (1981) New Zealand sea lion. *In*: Ridway S H; Harrisson, R J (Eds.), Handbook of Marine Mammals. Volume 1. Academic Press, London.

Wilkinson, I S; Burgess, J; Cawthorn, M W (2003) New Zealand sea lions and squid – managing fisheries impacts on a threatened marine mammal. *In*: Gales, N; Hindell, M; Kirkwood, R (Eds.), Marine mammals: Fisheries, tourism and management issues. pp 192–207. CSIRO Publishing, Melbourne.

Wilkinson, I S; Childerhouse, S J; Duignan, P J; Gulland, F M D (2000) Infanticide and cannibalism in the New Zealand sea lion. *Marine Mammal Science 16*: 495–500.

Wilkinson, I S; Duignan, P J; Grinberg, A; Chilvers, B L; Robertson, B C (2006) Klebsiella pneumonia epidemics: Possible impact on New Zealand sea lion recruitment. *In*: Trites, A; Atkinson, S; DeMaster, D; Fritz, L; Gelatt, T; Rea, L; Wynne, K (Eds.), Sea lions of the world. pp 455–471. Alaska Sea Grant College Program, University of Alaska Fairbanks.

Woodley, T H; Lavigne, D M (1991) Incidental capture of pinnipeds in commercial fishing gear. *International Marine Mammal Association Technical Report 91-01*. 35 p.

5 NEW ZEALAND FUR SEAL (*ARCTOCEPHALUS FORSTERI*)

Status of chapter	This chapter has been partially updated for AEBAR 2018. Updated figures and tables have been identified in their captions.
Scope of chapter	This chapter describes: the biology New Zealand fur seals (*Arctocephalus forsteri)* the nature and extent of potential interactions with fisheries; management of fisheries interactions; means of estimating fisheries impacts and population level risk; and remaining sources of uncertainty, to guide future work.
Area	All of the New Zealand EEZ and Territorial Sea, but primarily in coastal environments extending to the continental slope.
Key locations	Areas with the potential for significant fisheries interactions include waters over or close to the continental shelf surrounding the South Island and southern offshore islands, notably Cook Strait, West Coast South Island, Banks Peninsula, Stewart-Snares Shelf, Campbell Rise, and the Bounty Islands, plus offshore of Bay of Plenty-East Cape. Interactions also occur off the west coast of the North Island.
Key issues	Improved means of estimating fisheries captures and and risk in poorly observed inshore fisheries; improved understanding of population size, structure and trend on a regional basis; improved understanding of spatio-temporal distributions affecting encounter rates between fur seals and fishing effort.
Emerging issues	Improved ability to assess risk and apply risk management solutions on a regional sub-population basis, or at finer spatial and temporal scales.
MPI research (current)	*PMM2018-04A: Estimate spatial distributions for at-risk marine mammals to assess fisheries overlap and risk: fur seal; PMM2018-07: Updated spatially explicit fisheries risk assessment for New Zealand marine mammal populations*
NZ government research (current)	DOC Marine Conservation Services Programme (CSP): INT2015-02 *To determine which marine mammal, turtle and protected fish species are captured in fisheries and their mode of capture*; MIT2014-01 *Protected species bycatch newsletter*.
Related chapters/issues	Chapter 3: Spatially Explicit Fisheries Risk Assessment (SEFRA); Chapter 4 New Zealand Sea Lions

5.1 CONTEXT

Management of fisheries impacts on New Zealand fur seals is legislated under the Marine Mammals Protection Act (MMPA) 1978 and the Fisheries Act (FA) 1996. Under s.3E of the MMPA or s.14F of the Wildlife Act 1953, the Minister of Conservation, with the concurrence of the Minister for Primary Industries (formerly the Minister of Fisheries), may approve a population management plan (PMP). There is no PMP in place for New Zealand fur seals.

In the absence of a PMP, the Ministry for Primary Industries (MPI) manages fishing-related mortality of New Zealand fur seals under s.15(2) of the FA '*to avoid, remedy, or mitigate the effect of fishing-related mortality on any protected species, and such measures may include setting a limit on fishing-related mortality.*'

All marine mammal species are designated as protected species under s.2(1) of the FA. In 2005, the Minister of Conservation approved the Conservation General Policy, which specifies in Policy 4.4 (f) that '*Protected marine species should be managed for their long-term viability and recovery throughout their natural range.*' DOC's Regional Conservation Management Strategies outline specific policies and objectives for protected marine species at a regional level. Baker et al. (2016) list New Zealand fur seals as Not Threatened in 2009, and the IUCN classification is Least Concern (Chilvers & Goldsworthy 2015).

In 2004, DOC approved the *Department of Conservation Marine Mammal Action Plan for 2005–2010*, which still reflects their for marine mammal conservation (Suisted & Neale 2004). The plan specifies a number of species-specific key objectives for New Zealand fur seals, of which the following is most relevant for fisheries interactions: '*To control/mitigate fishing-related mortality of New Zealand*

fur seals in trawl fisheries (including the WCSI hoki and Bounty Island southern blue whiting fisheries). Management of New Zealand fur seal incidental captures aligns with Fisheries 2030 Objective 6: Manage impacts of fishing and aquaculture. Further, the management actions follow Strategic Action 6.2: *Set and monitor environmental standards, including for threatened and protected species and seabed impacts*.

All National Fisheries Plans except those for inshore shellfish and freshwater fisheries are relevant to the management of fishing-related mortality of New Zealand fur seals.

The relevant Fisheries Plan for the management of incidental captures of New Zealand fur seals is the 'National Fisheries Plan for Deepwater and Middle-depth Fisheries Part 1A' (the National Deepwater Plan). Under the National Deepwater Plan, the objective most relevant for management of New Zealand fur seals is Environmental Outcome 8: *Manage deepwater and middle-depth fisheries to avoid, remedy or mitigate the adverse effects of these fisheries on the long-term viability of endangered, threatened and protected species*

Specific objectives for the management of incidental captures of New Zealand fur seals are outlined in the fishery-specific chapters of the National Deepwater Plan for the fisheries with which New Zealand fur seals are most likely to interact. These fisheries include trawl fisheries for hoki, hake and ling, jack mackerel, and southern blue whiting. The chapters are being reviewed and updated in 2019.

Fisheries New Zealand works closely with the fishing industry to increase awareness amongst the fishing fleet of how to minimise interactions with fur seals, and emphasises the importance of adherence to the industry Marine Mammal Operational Procedures (MMOP). These procedures aim to reduce the risk of interactions with marine mammals by requiring that vessels:

- Minimise the length of time the fishing gear is on the surface;
- Remove all pieces of dead fish from the net before shooting the gear;
- Steam away from any congregations of marine mammals before shooting the gear; and
- Appoint a crew member to watch for marine mammal interactions every time the gear is shot or hauled.

Performance in relation to these procedures is audited by Fisheries New Zealand Observers and reported in the Deepwater Annual Review Report (ARR).

5.2 BIOLOGY

5.2.1 TAXONOMY

The New Zealand fur seal (*Arctocephalus forsteri*; Lesson 1828) is an otariid seal (Family Otariidae – eared seals, including fur seals and sea lions), one of two native to New Zealand, the other being the New Zealand sea lion (*Phocarctos hookeri*; Gray 1844).

5.2.2 DISTRIBUTION

Pre-European archaeological evidence suggests that New Zealand fur seals were present along much of the east coasts of the North Island (except the less rocky coastline of Bay of Plenty and Hawke's Bay) and the South Island, and, to a lesser extent, on the west coasts, where fewer areas of suitable habitat were available (Smith 1989, 2005, 2011). A combination of subsistence hunting and commercial harvest resulted in contraction of the species' range and in population decline almost to the point of extinction (Smith 1989, 2005, 2011, Ling 2002, Lalas 2008). New Zealand fur seals became fully protected in the 1890s and, with the exception of one year of licensed harvest in the 1950s, have remained protected since that time.

Currently, New Zealand fur seals occur throughout New Zealand waters, predominantly in waters south of 40°S and as far south as Macquarie Island. On land, New Zealand fur seals are distributed around the New Zealand coastline, on offshore islands, and on subantarctic islands (Crawley & Wilson 1976, Wilson 1981, Mattlin 1987). The recolonisation of the coastline by New Zealand fur seals has resulted in the northward expansion of the distribution of breeding colonies and haulouts (Lalas & Bradshaw 2001), and breeding colonies are now present on many exposed rocky areas (Baird 2011). The extent of breeding colony distribution in New Zealand waters is bounded to the north by a very small (space-limited) colony at Gannet Island off the North Island west coast (latitude 38°S), to the east by colonies of unknown sizes at the Chatham Islands group, to the west by colonies of unknown size on Fiordland offshore islands, and to the south by unknown numbers on Campbell

Island. Outside New Zealand waters, breeding populations exist in South and Western Australia (Shaughnessy et al. 1994, Shaughnessy 1999, Goldsworthy et al. 2003), with smaller colonies in Tasmania (Gales et al. 2010).

The seasonal distribution of the New Zealand fur seals is determined by the sex and maturity of each animal. Males are generally at the breeding colonies from late October to late January then move to haulout areas around the New Zealand coastline (see Bradshaw et al. 1999), with peak density of males and sub-adult males at haulouts during July–August and lowest densities in September–October (Crawley & Wilson 1976). Females arrive at the breeding colony from November and lactating females remain at the colony (apart from short foraging trips) for about 10 months until the pups are weaned, usually during August–September (Crawley & Wilson 1976).

5.2.3 FORAGING ECOLOGY

Most New Zealand fur seal foraging research in New Zealand has focused on lactating females at Open Bay Islands off the South Island west coast (Mattlin et al. 1998), Otago Peninsula (Harcourt et al. 2002), and Ohau Point, Kaikoura (Boren 2005), using time-depth recorders, satellite-tracking, or very-high-frequency transmitters. Individual females show distinct dive pattern behaviour and may be relatively shallow or deep divers, but most forage at night and in depths shallower than 200 m. At Open Bay Islands, dives were generally deeper and longer in duration during autumn and winter. Females dove to at least 274 m (for a 5.67 min dive in autumn) and remained near the bottom at over 237 m for up to 11.17 min in winter (Mattlin et al. 1998). Females in some locations undertook longer dive trips, with some to deeper waters, in autumn (in over 1000 m beyond the continental shelf; Harcourt et al. 2002).

The relatively shallow dives and nocturnal feeding observed during summer suggests that seals feed on pelagic and vertical migrating prey species (for example, arrow squid, *Nototodarus sloanii*). Conversely, the deeper dives and increased number of dives in daylight during autumn and winter suggest that prey species at this time may include benthic, demersal, and pelagic species (Mattlin et al. 1998, Harcourt et al. 2002). The deeper dives enabled seals to forage along or off the continental shelf (within 10 km) of the studied colony (at Open Bay Islands). These deeper dives may be demersal or to depths in the water column where spawning hoki are concentrated.

Methods to analyse New Zealand fur seal diets have included investigation of freshly killed animals (Sorensen 1969), scats, and regurgitates (e.g., Allum & Maddigan 2012). Fish prey items can be recognised by the presence of otoliths, bones, scales, and lenses, while cephalopods are indicated by beaks and pens. Foraging modes appear to vary between specific individuals, and distinct diets may be apparent in the scats and regurgitations of males vs females vs juveniles from the same colony. These analyses can be biased, however, particularly if only one collection method is used, and this limits fully quantitative assessment of prey species composition.

Dietary studies of New Zealand fur seals have been conducted at colonies in Nelson-Marlborough, west coast South Island, Otago Peninsula, Kaikoura, Banks Peninsula, Snares Islands, and off Stewart Island, and summaries are provided by Carey (1992), Harcourt (2001), Boren (2010), and Baird (2011).

New Zealand fur seals are opportunistic foragers and, depending on the time of year, method of analysis, and location, their diet includes at least 61 taxa (Holborow 1999) of mainly fish (particularly lanternfish (myctophids) in all studied colonies except Tonga Island (in Golden Bay; Willis et al. 2008), as well as anchovy (*Engraulis australis*), aruhu (*Auchenoceros punctatus*), barracouta (*Thrysites atun*), hoki (*Macruronus novaezelandiae*), jack mackerel (*Trachurus* spp.), pilchard (*Sardinops sagax*), red cod (*Pseudophycis bachus*), red gurnard (*Chelidonichthys kumu*), silverside (*Argentina elongate*), sprat (*Sprattus* spp.) and cephalopods (octopus (*Macroctopus maorum*), squid (*Nototodarus sloanii*, *Sepioteuthis bilineata*)). For example, myctophids were present in Otago scats throughout the year (representing offshore foraging), but aruhu, sprat, and juvenile red cod were present only during winter-spring (Fea et al. 1999). Medium-large arrow squid predominated in summer and autumn. Jack mackerel species, barracouta, and octopus were dominant in winter and spring. Prey such as lanternfish and arrow squid rise in the water column at night, the time when New Zealand fur seals exhibit shallow foraging (Harcourt et al. 1995, Mattlin et al. 1998, Fea et al. 1999).

Recent foraging and dietary studies include one on male fur seal diets by Lalas & Webster (2014) and one on lactating females by Meynier et al. (2013). Arrow squid was the most important dietary item in fur seal scats and regurgitations sampled from male fur seals at The Snares during February 2012 (Lalas & Webster 2014). Meynier et al. (2013) assess

the trophic and spatial overlap between fur seals from two different South Island locations with local fisheries using analyses of dietary fatty acids, stable isotope signals, and telemetry. Lactating females from the east coast rookery at Ohau Point fed on oceanic prey in summer and females from the west coast rookery at Cape Foulwind fed on benthic or coastal prey over the continental shelf in summer and winter. The west coast females spent 50% of their at-sea time in winter in and near the Hokitika Canyon, where the winter spawning hoki fishery operates.

5.2.4 REPRODUCTIVE BIOLOGY

New Zealand fur seals are sexually dimorphic and polygynous (Crawley & Wilson 1976); males may weigh up to 160 kg, whereas females weigh up to about 50 kg (Miller 1975, Mattlin 1978a, 1987, Troy et al. 1999). Adult males are much larger around the neck and shoulders than females and breeding males are on average 3.5 times the weight of breeding females (Crawley & Wilson 1976). Females are philopatric and are sexually mature at 4–6 years, whereas males mature at 5–9 years (Mattlin 1987, Dickie & Dawson 2003). The maximum age recorded for New Zealand fur seals in New Zealand waters is 22 years for females (Dickie & Dawson 2003) and 15 years for males (Mattlin 1978a).

New Zealand fur seals are annual breeders and generally produce one pup after a gestation period of about 10 months (Crawley & Wilson 1976). Twinning can occur and females may foster a pup (Dowell et al. 2008), although both are rare. Breeding animals come ashore to mate after a period of sustained feeding at sea. Breeding males arrive at the colonies to establish territories during October–November. Breeding females arrive at the colony from late November and give birth shortly after. Peak pupping occurs in mid-December (Crawley & Wilson 1976).

Females remain at the colony with their newborn pups for about 10 days, by which time they have usually mated. Females then leave the colony on short foraging trips of 3–5 days before returning to suckle pups for 2–4 days (Crawley & Wilson 1976). As the pups grow, these foraging trips are progressively longer in duration. Pups remain at the breeding colony from birth until weaning (at 8–12 months of age).

Breeding males generally disperse after mating to feed and occupy haulout areas, often in more northern areas (Crawley & Wilson 1976). This movement of breeding adults away from the colony area during January allows for an influx of sub-adults from nearby areas. Little is described about the ratio of males to females on breeding colonies (Crawley & Wilson 1976), or the reproductive success. Boren (2005) reported a fecundity rate of 62% for a Kaikoura colony, based on two annual samples of between about 5 and 8% of the breeding female population. This rate is similar to the 67% estimated by Goldsworthy & Shaughnessy (1994) for a South Australian colony.

Newborn pups are about 55 cm long and weigh about 3.5 kg (Crawley & Wilson 1976). Male pups are generally heavier than female pups at birth and throughout their growth (Crawley & Wilson 1976, Mattlin 1981, Chilvers et al. 1995, Bradshaw et al. 2003b, Boren 2005). Pup growth rates may vary by colony (see Harcourt 2001). The proximity of a colony to easily accessible rich food sources will vary, and pup condition at a colony can vary markedly between years (Mattlin 1981, Bradshaw et al. 2000, Boren 2005). Food availability may be affected by climate variation, and pup growth rates probably represent variation in the ability of mothers to provision their pups from year to year. The sex ratio of pups at a colony may vary by season (Bradshaw et al. 2003a, 2003b, Boren 2005), and in years of high food resource availability, more mothers may produce males or more males may survive (Bradshaw et al. 2003a, 2003b).

5.2.5 POPULATION BIOLOGY

Historically, the population of New Zealand fur seals in New Zealand was thought to number above 1.25 million animals (possibly as high as 1.5 to 2 million) before the extensive sealing of the early 19th century (Richards 1994). Present day population estimates for New Zealand fur seals in New Zealand are dated, few and highly localised. In the most comprehensive attempt to quantify the total New Zealand fur seal population, Wilson (1981) summarised population surveys of mainland New Zealand and offshore islands undertaken in the 1970s and estimated the population size within the New Zealand region at between 30 000 and 50 000 animals. Since then, several authors have suggested a population size of ~100 000 animals (Taylor 1990, see Harcourt 2001), but this estimate is very much an approximation and its accuracy is difficult to assess in the absence of comprehensive surveys.

Fur seal colonies provide the best data for consistent estimates of population numbers, generally based on pup production in a season (see Shaughnessy et al. 1994). Data

used to provide colony population estimates of New Zealand fur seals have been, and generally continue to be, collected in an ad hoc fashion. Regular pup counts are made at some discrete populations. A 20-year time series of Otago Peninsula colony data is updated, maintained, and published primarily by Chris Lalas (assisted by Sanford (South Island) Limited), and the most recent published estimate is 20 000–30 000 animals (Lalas 2008). Lalas & MacDiarmid (submitted) applied a logistic growth model, using established parameters, to 13 years of pup production estimates from colonies at Oamaru south to Slope Point, and indicated the 2009 population was at 95% of the asymptote of 19 600 animals (plausible range of 13 000–28 800). In this region, 90% of the population growth occurred over 24–27 years; and the growth rate was faster in seasons up to 1998, than in later years.

Similar population growth rates occurred at Kaikoura, where the population expanded by 32% per annum over the years 1990–2005 (Boren et al. 2006). An estimate of 600 pups was reported for 2005 (Boren 2005), 1508 (s.e. = 28) pups were estimated for 2009, and 2390 (s.e. = 226) pups for 2011 (L. Boren, DOC, pers. comm.).

Since 1991, the Department of Conservation has monitored New Zealand fur seal pup production at three breeding colonies on the West Coast, at Cape Foulwind, Wekakura Point, and Taumaka (Open Bay Islands) (see Best 2011). A DOC-commissioned project is underway to compile the tag, measurement, and mark-recapture data from these colonies and create a New Zealand fur seal database (Roberts & Best 2016). The data have been made available by the scientists who complete the fieldwork, most recently by Hugh Best, who coordinates the population monitoring programme, DOC Regional and District staff, Tai Poutini Papatipu Runanga, and the trustee owners of Taumaka me Popotai. Once the database has been through a quality assurance process, it will be made publically available. The pup production estimates for these colonies are derived using direct counts of dead pups and mark recapture methodology undertaken in the last week of January each year. At Taumaku Island, the largest of the Open Bay Islands and the most southern of these three colonies, approximately 800 pups are marked each year, and the first 100 pups of each sex are weighed and measured. At Cape Foulwind, approximately 200 pups are marked each year, and the first 50 of each sex are weighed and measured. At the most northern of the three colonies, Wekakura Point, approximately 500 pups are marked and 75 of each sex are weighed and measured.

Other studies of breeding colonies generally provide estimates for one or two seasons, but many of these are more than 10 years old. Published estimates suggest that populations have stabilised at the Snares Islands after a period of growth in the 1950s and 1960s (Carey 1998) and increased at the Bounty Islands (Taylor 1996), Nelson-Marlborough region (Taylor et al. 1995), Kaikoura (Boren 2005), Otago (Lalas & Harcourt 1995, Lalas & Murphy 1998, Lalas 2008, Lalas & MacDiarmid, submitted), and near Wellington (Dix 1993).

For many areas where colonies or haulouts exist, count data have been collected opportunistically (generally by Department of Conservation staff during their field activities) and thus data are not often comparable because counts may represent different life stages, different assessment methods, and different seasons (see Baird 2011). Known breeding locations (as at October 2012) are summarised in the NABIS supporting lineage document for the '*Breeding colonies distribution of New Zealand fur seal*' layer[1].

Baker et al. (2010) conducted an aerial survey of the South Island west coast from Farewell Spit to Puysegur Point and Solander Island in 2009, but their counts were quite different, i.e., lower than ground counts collected at a similar time at the main colonies (Mellina & Cawthorn 2009). This discrepancy was thought to be a result mainly of the survey design and the nature of the terrain. However, the aerial survey confirmed the localities shown by Wilson (1981) of potentially large numbers of pups at sites such as Cascade Point, Yates Point, Chalky Island, and Solander Island.

[1] http://www2.nabis.govt.nz/LayerDetails.aspx?layer=Breeding%20colonies%20distribution%20of%20New%20Zealand%20fur%20seal.

Population numbers for some areas, especially more isolated ones, are not well known. The most recent counts for the Chatham Islands were collected in the 1970s (Wilson 1981), and the most recent reported for the Bounty Islands were made in 1993–94. Taylor (1996) reported an increase in pup production at the Bounty Islands since 1980, and estimated that the total population was at least 21 500, occupying over 50% of the available area. Information is sparse for populations at Campbell Island, the Auckland Islands group and the Antipodes Islands

Little is reported about the natural mortality of New Zealand fur seals, other than reports of sources and estimates of pup mortality for some breeding colonies. Estimates of pup mortality or pup survival vary in the manner in which they were determined and in the number of seasons they represent, and are not directly comparable. Each colony will be affected by different sources of mortality related to habitat, location, food availability, environment, and year, as well as the ability of observers to count all the dead pups (may be limited by terrain, weather, or time of day).

Reported pup mortality rates vary: 8% for Otago Peninsula pups up to 30 days old and 23% for pups up to 66 days old (Lalas & Harcourt 1995); 20% from birth to 50 days and about 40% from birth to 300 days for Taumaka Island, Open Bay Islands pups (Mattlin 1978b); and in one year, 3% of Kaikoura pups before the age of 50 days (Boren 2005). Starvation was the major cause of death, although stillbirth, suffocation, trampling, drowning, predation, and human disturbance also occur. Pup survival of at least 85% was estimated for a mean 47-day interval for three Otago colonies, incorporating data such as pup body mass (Bradshaw et al. 2003b), though pup mortality before the first capture effort was unknown. Other sources of natural mortality for New Zealand fur seals include predators such as sharks and New Zealand sea lions (Mattlin 1978b, Bradshaw et al. 1998).

Human-induced sources of mortality include: fishing, for example, entanglement or capture in fishing gear; vehicle-related deaths (Lalas & Bradshaw 2001, Boren 2005, Boren et al. 2006, 2008); and mortality through shooting, bludgeoning, and dog attacks. New Zealand fur seals are vulnerable to certain bacterial diseases and parasites and environmental contaminants, though it is not clear how life-threatening these are. The more obvious problems include tuberculosis infections, *Salmonella*, hookworm enteritis, phocine distemper, and septicaemia (associated with abortion) (Duignan 2003, Duignan & Jones 2007). Low food availability and persistent organohalogen compounds (which can affect the immune and the reproductive systems) may also affect New Zealand fur seal health.

Various authors have investigated fur seal genetic differentiation among colonies and regions in New Zealand (Lento et al. 1994, Robertson & Gemmell 2005). Lento et al. (1994) described the geographic distribution of mitochondrial cytochrome *b* DNA haplotypes. Robertson & Gemmell (2005) described low levels of genetic differentiation (consistent with homogenising gene flow between colonies and an expanding population) based on genetic material from New Zealand fur seal pups from seven colonies. One aim of the latter work is to determine the provenance of animals captured during fishing activities, through the identification and isolation of any colony genetic differences.

In 2015–16, Gooday et al. (unpub., 2016) conducted trials of unmanned aerial vehicle (UAV) technology combined with thermal imaging in the Ohau Point fur seal colony, as part of an investigation into non-invasive population sampling. They found aerial surveys using a T320 19 mm infrared camera were successful in detecting fur seals in open areas and distinguishing them from rocks, but were unsuccessful in areas of high canopy cover (>80%). Ground surveys were also conducted using a higher resolution Optris PL450^{TM} infrared camera and detected more fur seals than paired photographs during cooler times of the day (morning and evening). In the Ohau Stream where seal pups visit the waterfall, the Optris PL450^{TM} detected pups hiding in the forested areas better than the naked eye, but was less effective when they were swimming or if they had recently left the water. The Optris PL450^{TM} is currently under development to be mounted to the UAV, which is expected to increase aerial counts dramatically. Gooday et al. (unpub., 2016) concluded that thermal imagery has the potential to become an effective and widely used tool for ecological population surveys.

5.2.6 CONSERVATION BIOLOGY AND THREAT CLASSIFICATION

Threat classification is an established approach for identifying species at risk of extinction (IUCN 2014). The risk of extinction for New Zealand fur seals has been assessed under two threat classification systems: the New Zealand Threat Classification System (Townsend et al. 2008) and the

International Union for the Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2014).

In 2008, the IUCN updated the Red List status of New Zealand fur seals, listing them as Least Concern on the basis of their large and apparently increasing population size (Chilvers & Goldsworthy 2015). In 2010, DOC updated the New Zealand Threat Classification status of all New Zealand marine mammals (Baker et al. 2016). In the revised list, New Zealand fur seals were classified as Not Threatened with the qualifiers increasing (Inc) and secure overseas (SO) (Baker et al. 2016).

5.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

New Zealand fur seals are found in both Australian and New Zealand waters. Overall abundance has been suggested to be as high as 200 000, with about half of the population in Australian waters (Goldsworthy & Gales 2008). However, this figure is very much an approximation, and its accuracy is difficult to assess in the absence of comprehensive surveys.

Pinnipeds are caught incidentally in a variety of fisheries worldwide (Read et al. 2006). Outside New Zealand waters, species captured include: New Zealand fur seals, Australian fur seals, and Australian sea lions in Australian trawl and inshore fisheries (e.g., Shaughnessy 1999, Norman 2000); Cape fur seals in South African fisheries (Shaughnessy & Payne 1979); South American sea lions in trawl fisheries off Patagonia (Dans et al. 2003); and seals and sea lions in United States waters (Moore et al. 2009).

5.4 STATE OF KNOWLEDGE IN NEW ZEALAND

New Zealand fur seals are attracted to feeding opportunities offered by various fishing gears. Anecdotal evidence suggests that the sound of winches as trawlers haul their gear acts as a cue. The attraction of fish in a trawl net, on longline hooks, or caught in a set net provide opportunities for New Zealand fur seals to interact with fishing gear, which can result in capture and, potentially, death via drowning

Most captures occur in trawl fisheries and New Zealand fur seals are most at risk from capture during shooting and hauling (Shaughnessy & Payne 1979), when the net mouth is within diving depths. Once in the net some animals may have difficulty in finding their way out within their maximum breath-hold time (Shaughnessy & Davenport 1996). The operational aspects that are associated with New Zealand fur seal captures on trawlers include factors that attract the New Zealand fur seals, such as the presence of offal and discards, the sound of the winches, vessel lights, and the presence of 'stickers' in the net (Baird 2005). It is considered that New Zealand fur seals are at particular risk of capture when a vessel partially hauls the net during a tow and executes a turn with the gear close to the surface. At the haul, New Zealand fur seals often attempt to feed from the codend as it is hauled and dive after fish that come loose and escape from the net (Baird 2005).

Factors identified as important influences on the potential capture of New Zealand fur seals in trawl gear include the year or season, the fishery area, gear type and fishing strategies (often specific to certain nationalities within the fleet), time of day, and distance to shore (Baird & Bradford 2000, Mormede et al. 2008, Smith & Baird 2009). These analyses did not include any information on New Zealand fur seal numbers or activity in the water at the stern of the vessel because of a lack of data. Other influences on New Zealand fur seal capture rate (of Australian and New Zealand fur seals) may include inclement weather and sea state, vessel tow and haul speed, increased numbers of vessels and trawl frequency, and potentially the weight of the fish catch and the presence of certain bycatch fish species (Hamer & Goldsworthy 2006). This Australian study found similar mortality rates for tows with and without Seal Exclusion Devices (see also Hooper et al. 2005). The use of fur seal exclusion devices is not required in New Zealand fisheries.

The spatial and temporal overlap of commercial fishing grounds and New Zealand fur seal foraging areas has resulted in New Zealand fur seal captures in fishing gear (Mattlin 1987, Rowe 2009). Most fisheries with observed captures occur in waters over or close to the continental shelf. Because the topography around much of the South Island and offshore islands slopes steeply to deeper waters, most captures occur close to colonies and haulouts. Locations of captures by trawl vessels and surface longline vessels are shown in Figures 5.1 and 5.2. Winter hoki fisheries attract New Zealand fur seals off the west coast South Island and in Cook Strait between late June and September (Table 5.1). In August–October, New Zealand fur seals are caught in southern blue whiting effort near the Bounty Islands and Campbell Island. In September–October captures may occur in hoki and ling fisheries off Puysegur Point on the south-western coast of the South Island.

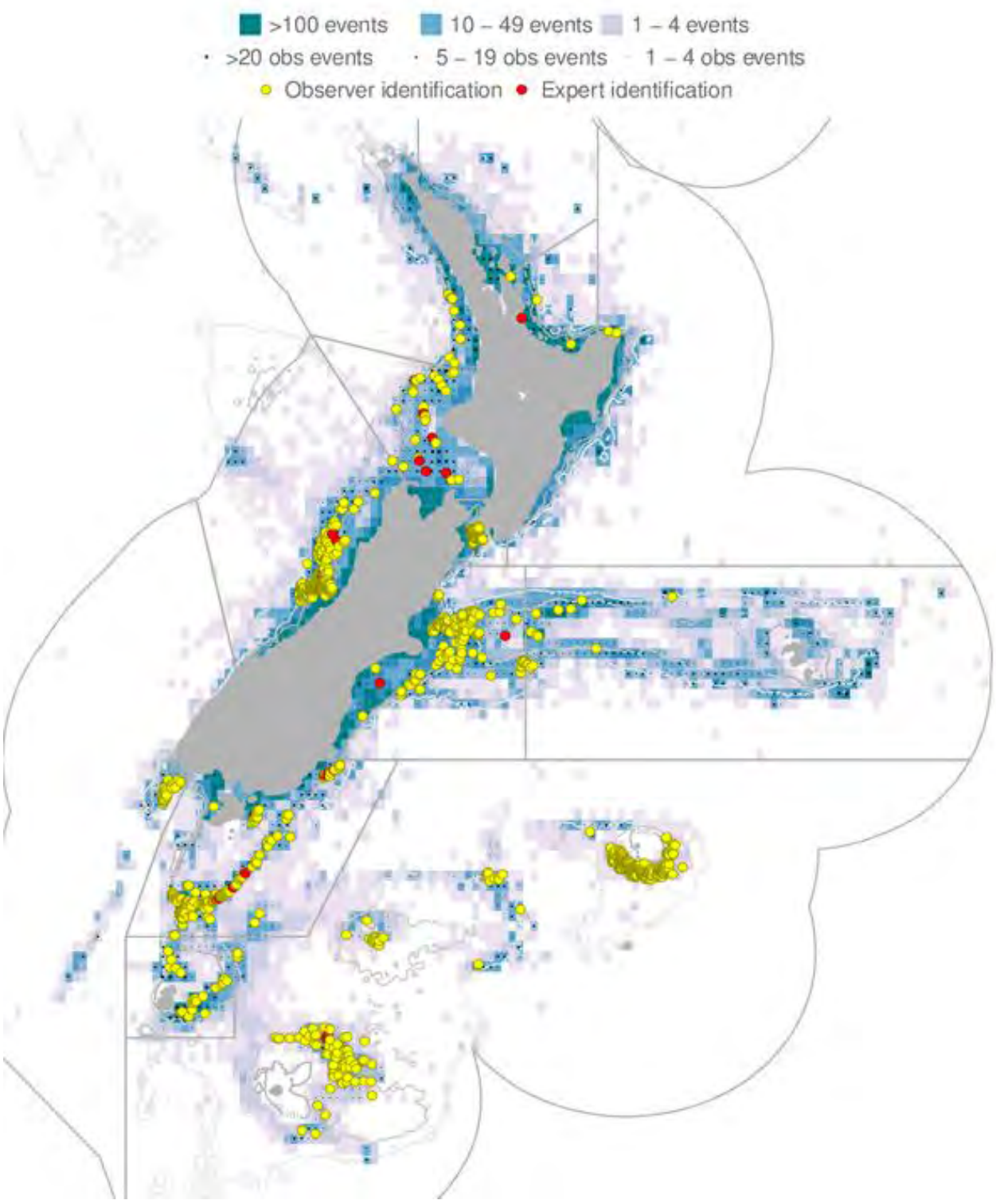


Figure 5.1: Distribution of trawl fishing effort and observed New Zealand fur seal captures, 2002–03 to 2016–17 (for more information see MPI data analysis at http://data.dragonfly.co.nz/psc, data version v2018001). Fishing effort is mapped into 0.2-degree cells, coloured to represent the amount of effort. Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. Fishing effort is shown for all tows with latitude and longitude data, where three or more vessels fished within a cell. [Updated for AEBAR 2018].

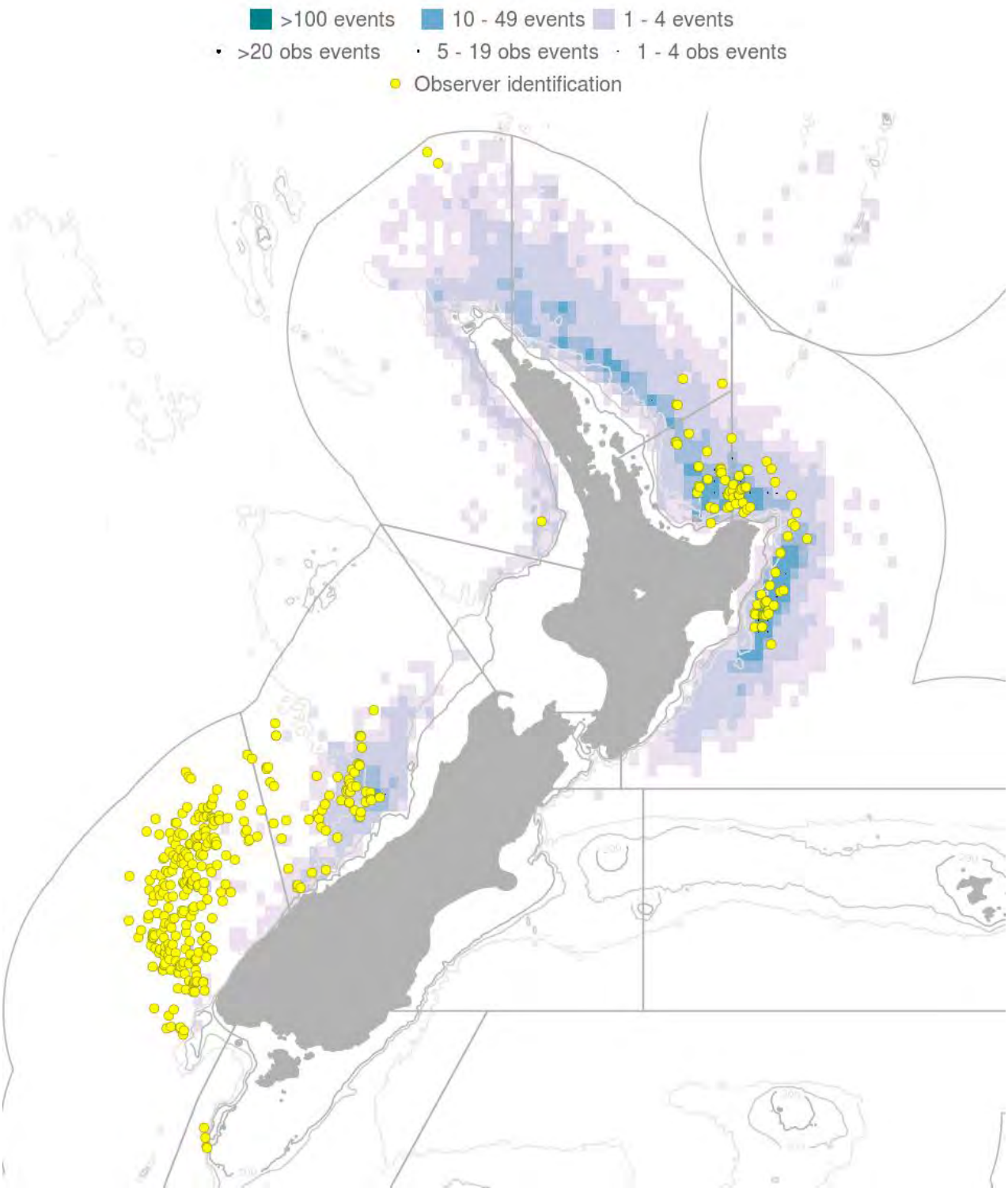


Figure 5.2: Distribution of surface-longline fishing effort and observed New Zealand fur seal captures, 2002–03 to 2016–17 (for more information see MPI data analysis at http://data.dragonfly.co.nz/psc, data version v2018001). Fishing effort is mapped into 0.2-degree cells, coloured to represent the amount of effort. Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. Fishing effort is shown for sets with latitude and longitude data, where three or more vessels and three or more companies or persons fished within a cell. For these years, 89.6% of the effort is shown. [Updated for AEBAR 2018].

Captures are also reported from the Stewart-Snares shelf fisheries that operate during summer months, mainly for hoki and other middle depths species and squid, and from fisheries throughout the year on the Chatham Rise though captures have not been observed east of longitude 180° on the Chatham Rise.

Captures were reported from trawl fisheries for species such as hoki, hake (*Merluccius australis*), ling (*Genypterus blacodes*), squid, southern blue whiting, jack mackerel, and barracouta (Baird & Smith 2007, Abraham et al. 2010b). Between 1 and 3% of observed tows targeting middle-depths fish species catch New Zealand fur seals compared with about 1% for squid tows, and under 1% of observed tows targeting deepwater species such as orange roughy (*Hoplostethus atlanticus*) and oreo species (for example, *Allocyttus niger*, *Pseudocyttus maculatus*) (Baird & Smith 2007). The main fishery areas that contribute to the estimated annual catch of New Zealand fur seals (modelled from observed captures) in middle depths and deepwater trawl fisheries are Cook Strait hoki, west coast South Island middle-depths fisheries (mainly hoki), western Chatham Rise hoki, and the Bounty Islands southern blue whiting fishery (Baird & Smith 2007, Thompson & Abraham 2010). Captures on longlines occur when the New Zealand fur seals attempt to feed on the fish catch during hauling. Most New Zealand fur seals are released alive from surface and bottom longlines, typically with a hook and short snood or trace still attached.

5.4.1 QUANTIFYING FISHERIES INTERACTIONS

Observer data and commercial effort data have been used to characterise fur seal incidental captures and estimate the total catches (Baird & Smith 2007, Smith & Baird 2009, Thompson & Abraham 2010, Abraham & Thompson 2011, Abraham et al. 2017). This approach is currently applied using information collected under DOC project INT2013-01 and analysed under MPI project PRO2013-01 (Thompson et al. 2011, Thompson et al. 2012, Abraham et al. 2017). The analytical methods used to estimate capture numbers across commercial fisheries vary depending on the quantity and quality of the data, i.e., total numbers of observed captures and the representativeness of the observer coverage. Initially, stratified ratio estimates were provided for the main trawl fisheries, starting in the late 1980s, after scientific observers reported 198 New Zealand fur seal deaths during the July to September west coast South Island spawning hoki fishery (Mattlin 1994a, 1994b). In subsequent years, ratio estimation was used to estimate New Zealand fur seal captures in the Taranaki Bight jack mackerel fisheries and Bounty Platform, Pukaki Rise, and Campbell Rise southern blue whiting fisheries, based on observed catches and stratified by area, season, and gear type (Baird 1994).

In the last 10 years, model-based estimates of captures have been developed for all trawl fisheries in waters south of 40°S (Baird & Smith 2007, Smith & Baird 2009, Thompson & Abraham 2010, Abraham & Thompson 2011, Thompson et al. 2011, Thompson et al. 2012, Abraham et al. 2017). These models use fisheries observer data and fishing effort data in a hierarchical Bayesian model that includes season and vessel-season random effects and other covariates (for example, day of fishing year, time of day, tow duration, distance from shore, gear type, target) to model variation in capture rates among tows. This method compensates in part for the lack of representativeness of the observer coverage and includes the contribution from correlation in the capture rate among tows by the same vessel. The method is limited by the very large differences in the observed and non-observed proportions of data for the different vessel sizes; most observer coverage is on larger vessels that generally operate in waters deeper than 200 m. The operation of inshore vessels in terms of the location of effort, gear, and the vessel behaviour is only poorly understood compared with the deepwater fisheries. Nonetheless, following changes to reporting requirements, data collection is improving such that inshore trawl effort (not including flatfish trawl effort) is now included in the captures estimation modelling (Thompson et al. 2012, see also description of the Trawl Catch Effort Return, TCER, in use since 2007–08, in Chapter 11 on benthic effects).

Since 2005, there has been a downward, then relatively flat trend in estimated capture rates and total annual estimated captures of New Zealand fur seals in trawl fisheries (Smith & Baird 2009, Thompson & Abraham 2010, Abraham & Thompson 2011, Thompson et al. 2011, Thompson et al. 2012, Abraham et al. 2017; Figure 5.3). This may reflect bycatch reduction efforts undertaken by vessels (see Section 5.4.2) combined with a reduction in fishing effort since the late 1990s. Simultaneous with this decrease in effort is an increase in fisheries observer coverage, especially since 2007. In 2014–15, about 17% of the 78 696 tows were observed, with a capture rate of 0.93 fur seals per 100 tows, to give an annual mean total of 486 captures (95% c.i.: 299–876) (Table 5.2, Figure 5.3).

Observed and estimated capture rates include animals that are released alive; 13% of 1420 observed trawl captures in the 2002–03 to 2014–15 fishing years were recorded as released alive by the observer.

Ratio estimation was used to calculate total captures in longline fisheries by target fishery fleet and area (Baird 2008) and across all fishing methods (Abraham et al. 2010b). New Zealand fur seal captures in surface-longline fisheries have been generally observed in waters south and west of Fiordland, but also in the Bay of Plenty and off East Cape. Estimated surface-longline captures range from 299 (95% c.i.: 199–428) in 2002–03 to 32 (14–55) in 2006–07 (Table 5.2). These capture rates include animals that are released alive; 5.6% of observed surface-longline captures from 2002–03 to 2014–15 were live releases (Abraham et al. 2017).

Captures of New Zealand fur seals have also been recorded in other fisheries; 39 in set nets, 2 in bottom-longline fisheries and 1 from purse seine fisheries from 2002–03 to 2014–15 (Abraham et al. 2017). Because observer data are too sparse and/or unrepresentative to support the estimation method, capture estimation models are not produced for these fisheries. Captures associated with recreational fishing activities are poorly known (Abraham et al. 2010a)

Table 5.1: Monthly distribution of New Zealand fur seal activity and the main trawl and longline fisheries with observed reports of New Zealand fur seal incidental captures.

New Zealand fur seals	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Breeding males	Dispersed at sea or at haulouts	At breeding colony				Dispersed at sea or at haulouts						
Breeding females	At sea		At breeding colony		At breeding colony and at-sea foraging and suckling							
New pups	At sea			At breeding colony								
Non-breeders	Dispersed at sea, at haulouts, or breeding colony periphery											
Major fisheries	**Sep**	**Oct**	**Nov**	**Dec**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**
Hoki trawl		Chatham Rise and Stewart-Snares Shelf								Cook Strait, west coast South Island, Puysegur		
Squid				Stewart-Snares Shelf		Auckland Islands and Stewart-Snares Shelf						
Southern blue whiting	Pukaki Rise and Campbell Rise											Bounty Islands
Scampi	Mernoo Bank (Chatham Rise) and Auckland Islands											
Southern bluefin tuna longline							South-west South Island					

Table 5.2: Fishing effort and observed and estimated New Zealand fur seal captures in trawl and surface-longline fisheries by fishing year in the New Zealand EEZ (Abraham & Berkenbusch 2017, and see MPI data analysis at http://data.dragonfly.co.nz/psc, data version 2018001). For each fishing year, the table gives the total number of tows or hooks; the observer coverage (the percentage of tows or hooks that were observed); the number of observed captures (both dead and alive); the capture rate (captures per hundred tows or per thousand hooks); the estimation method used (model or ratio); and the mean number of estimated total captures (with 95% confidence interval). For more information on the methods used to prepare the data, see Abraham & Berkenbusch 2017. [Continued on next page]. [Updated for AEBAR 2018].

Fishing year	Fishing effort	Observed captures		Estimated captures		
	All effort	% observed	Number	Rate	Mean	95% c.i.
Trawl fisheries						
2002–03	130 195	5.3	68	0.994	927	646-1 307
2003–04	120 850	5.4	90	1.374	914	646-1 286
2004–05	120 468	6.4	199	2.580	1 579	1 170-2 123
2005–06	109 933	6.0	143	2.160	1 019	734-1 432
2006–07	103 323	7.7	74	0.933	660	469-916
2007–08	89 533	10.1	142	1.569	737	552-993
2008–09	87 552	11.2	72	0.737	493	353-682
2009–10	92 892	9.7	72	0.798	487	353-668
2010–11	86 085	8.7	73	0.980	551	374-819
2011–12	84 418	11.1	83	0.887	452	323-632
2012–13	83 838	14.8	121	0.976	600	413-904
2013–14	85 110	15.5	159	1.206	379	297-492
2014–15	78 767	17.2	127	0.936	479	352-653
2015–16	78 030	16.6	109	0.840	375	275-521
2016/2017	78 172	17.6	79	0.576	927	
Surface-longline fisheries						
2002–03	10 770 038	20.4	56	0.026	408	289-556
2003–04	7 386 749	21.8	40	0.025	177	127-242
2004–05	3 682 795	21.3	20	0.026	88	58-125
2005–06	3 692 119	19.1	12	0.017	58	33-91
2006–07	3 739 912	27.8	10	0.010	34	19-52
2007–08	2 246 339	18.8	10	0.024	44	25-67
2008–09	3 115 633	30.1	22	0.023	66	45-94
2009–10	2 995 264	22.1	19	0.029	93	60-135
2010–11	3 188 179	21.2	17	0.025	76	48-112
2011–12	3 100 127	23.5	40	0.055	174	127-231
2012–13	2 876 932	19.5	21	0.037	130	82-192
2013–14	2 549 764	30.7	57	0.073	204	156-262
2014–15	2 412 336	30.1	37	0.051	151	109-202
2015–16	2 357 141	13.7	3	0.009	24	8-49
2016/2017	2 092 486	16.5	32	0.093		

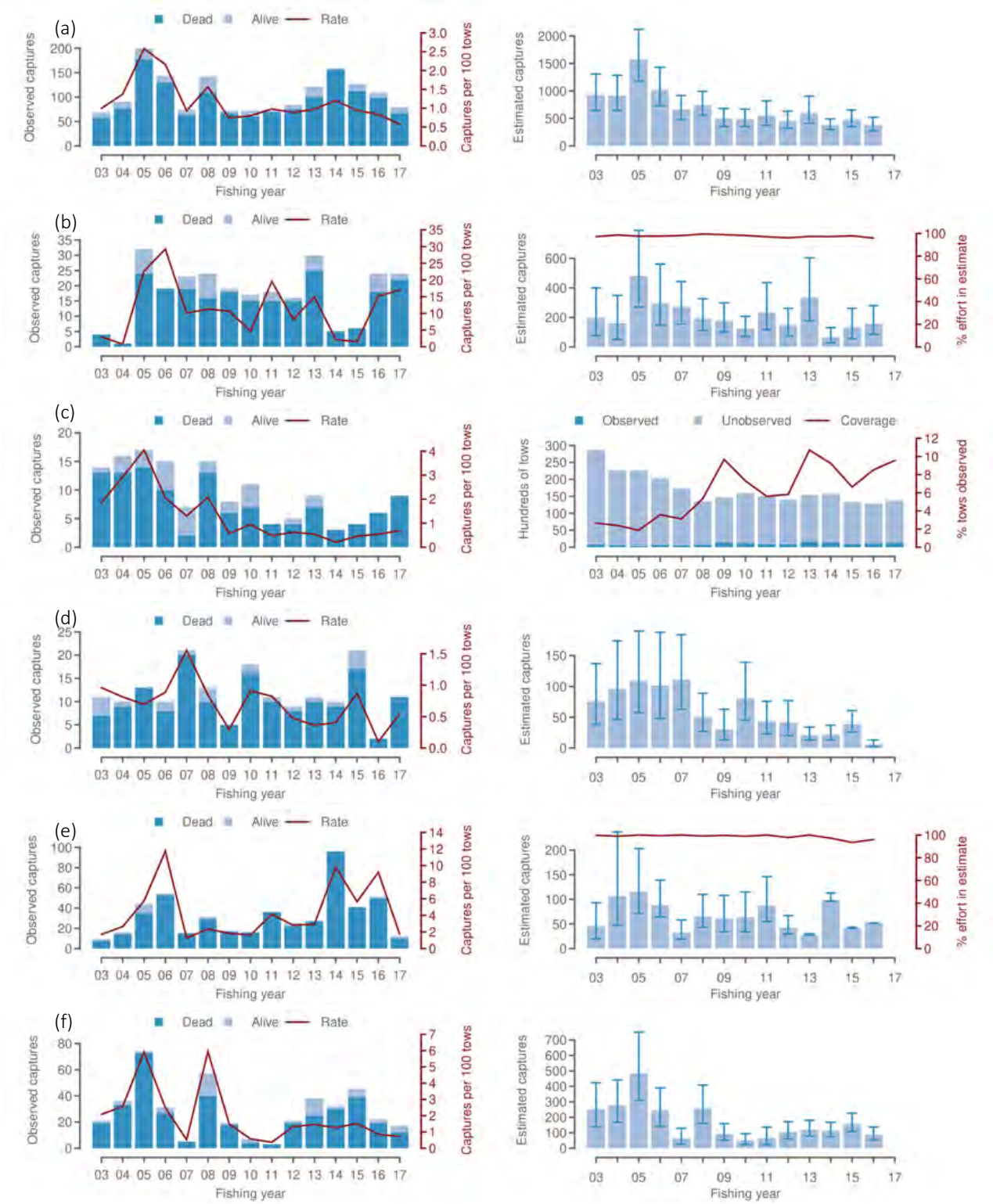


Figure 5.3. Observed captures of New Zealand fur seals (dead and alive) in trawl fisheries, the capture rate (per hundred tows), and the mean number of estimated total captures (with 95% confidence interval) by fishing year for regions with more than 50 observed captures since 2002–03: (a) New Zealand's EEZ; (b) the Cook Strait area; (c) the East Coast South Island area; (d) the Stewart-Snares Shelf area; and (e) the subantarctic area; and (f) the West Coast South Island area (Abraham et al. 2017, and see MPI data analysis at http://data.dragonfly.co.nz/psc, data version v2018001). Percentage effort included in the estimation is shown when it was less than 100%. For more information on the methods used to prepare the data, see Abraham and Thompson (2011). [Updated for AEBAR 2018].

5.4.2 MANAGING FISHERIES INTERACTIONS

The population level impact of direct fisheries mortalities on the New Zealand fur seal population remains somewhat uncertain. However, fishing interactions are considered unlikely to have adverse consequences for New Zealand fur seals at the scale of the entire New Zealand population on the basis of the following evidence: i) the estimated level of bycatch relative to overall New Zealand fur seal abundance; ii) the apparently increasing population and range at most colonies; and iii) the low threat status assigned to this species by both the New Zealand and IUCN threat classification processes. However, fisheries impact and risk may be higher at the scale of particular colonies, or affecting regional subpopulations.

Management has focused on encouraging vessel operators to alter fishing practices to reduce captures, and monitoring captures via the observer programme. A marine mammal operating procedure (MMOP) has been developed by the deepwater sector to reduce the risk of marine mammal captures and is currently applied to trawlers greater than 28 m LOA. It includes a number of mitigation measures supported by annual training, these include managing offal discharge, refraining from shooting the gear when New Zealand fur seals are congregating around the vessel and the introduction of 'trigger' points whereby if two fur seals are captured within 24 hours, or five seals over seven days, then the following procedure is triggered:

1. Advise vessel manager
2. Record capture event including location of capture in ship's log
3. Ensure gear failures are addressed with the gear either onboard or at a depth >50m
4. Report capture to Deepwater Group either directly or via shore management.

The major focus of the MMOP is to reduce the time gear is at or near the surface when it poses the greatest risk. MPI, via observers, monitors and audits vessel performance against this procedure (see the MPI National Deepwater Plan for further details). Research into methods to minimise or mitigate New Zealand fur seal captures in commercial fisheries has focused on fisheries in which New Zealand fur seals are more likely to be captured (trawl fisheries; see Clement and Associates 2009). Finding ways to mitigate captures has proved difficult because the animals are free swimming, can easily dive to the depths of the net when it is being deployed, hauled, or brought to the surface during a turn, and are known to actively and deliberately enter nets to feed. Further, any measures also need to ensure that the catch is not greatly compromised, either in terms of the amount of fish or their condition. Possible fish loss is one potential drawback of using seal exclusion devices (see Rowe 2007). Adhering to current risk mitigation methods (e.g., MMOP) will help to minimise the level of impacts, however bycatch rates are still expected to fluctuate depending on fleet deployment, New Zealand fur seal abundance and local feeding conditions.

5.4.3 MODELLING POPULATION-LEVEL IMPACTS OF FISHERIES INTERACTIONS

Uncertainty about the size of the New Zealand fur seal population limits our ability to estimate the effects of direct fisheries mortalities on sea lions at the scale of the New Zealand population. Potential impacts on specific colonies are best addressed via spatially explicit methods (below). The provenance of New Zealand fur seals caught during fishing is presently unknown. Improved research to understand foraging distributions in relation to colony locations is in progress (PMM2018-04A). In addition, genetics research may help to assign bycaught animals to a specific colony (Robertson & Gemmell 2005).

5.4.4 MULTI-SPECIES MARINE MAMMAL RISK ASSESSMENT

In 2017, the first iteration of a New Zealand Marine Mammal Risk Assessment (NZMMRA) was complete (Abraham et al. 2017) applying a partial implementation of the Spatially Explicit Fisheries Risk Assessment (SEFRA) method formerly applied for New Zealand seabirds and described in Chapter 3.

In the risk assessment outputs fur seals are the seventh-highest at-risk species of marine mammal from New Zealand commercial fisheries. Fisheries risk to fur seals is attributable primarily to 'other trawl' fisheries (i.e., primarily targeting hoki and southern blue whiting), and secondarily to set net fisheries. Estimated annual potential fishery-related deaths for fur seals by fishery group are shown in Figure 5.4.

The estimated cumulative fisheries risk score for fur seals ranges from approximately 0.2 to 0.6 (Figure 5.5), consistent with colony observations indicating a general trend of increasing population size in recent years. Note

that unlike the NZSRA, the NZMMRA does not utilise population monitoring results directly in the risk assessment to inform or constrain total fishery related deaths to be consistent with observed adult survival rates. Introducing this constraint is a priority when a full implementation of the SEFRA framework is delivered for all marine mammal species (PMM2018-07).

An independent external review of the SEFRA method (Lonergan et al. 2017) noted that the reliability and specific applicability of the previous NZMMRA is limited by its reliance on species spatial distributions derived from expert knowledge in which animal densities are assigned to discrete spatial blocks using a Delphi approach. The reviewers recommended that the MMRA should be updated using more reliable species spatial distributions as these become available. Input data layers reflecting finer-scale spatial and seasonal patterns are likely to be especially important for coastal and/or colony-associated species such as fur seals. Where sightings or satellite telemetry data are available, it is likely that these can be used to parameterise predictive spatial foraging models fitted to continuous environmental variables using multivariate statistical approaches, to estimate spatio-temporal species distributions in a more rigorous way. This work has recently been completed to improve available distribution models for cetaceans (under contract PRO2014-01) and for Māui and Hector's dolphins (PRO2017-12). This work is in progress for Auckland Island sea lions (PRO2017-09), for Stewart Island/ South Island sea lions (PMM2018-04B); and for New Zealand fur seals (PMM2018-04A). Because fur seals show sex-specific movement patterns, it is likely that this work will consider male and female distributions and mortalities separately, given that male and female deaths are likely to have very different implications for the population response of harem-breeding mammals

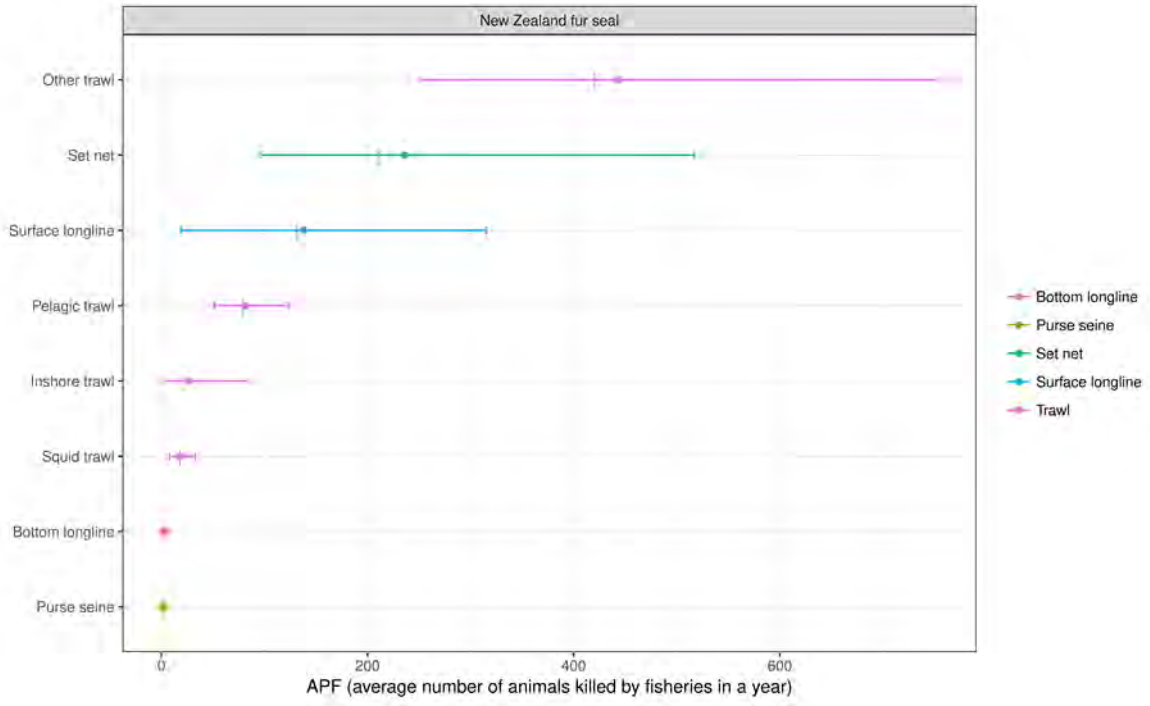


Figure 5.4: Preliminary estimates of annual potential fishery-related deaths of fur seals by fishery group, as estimated by the 2016 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017).

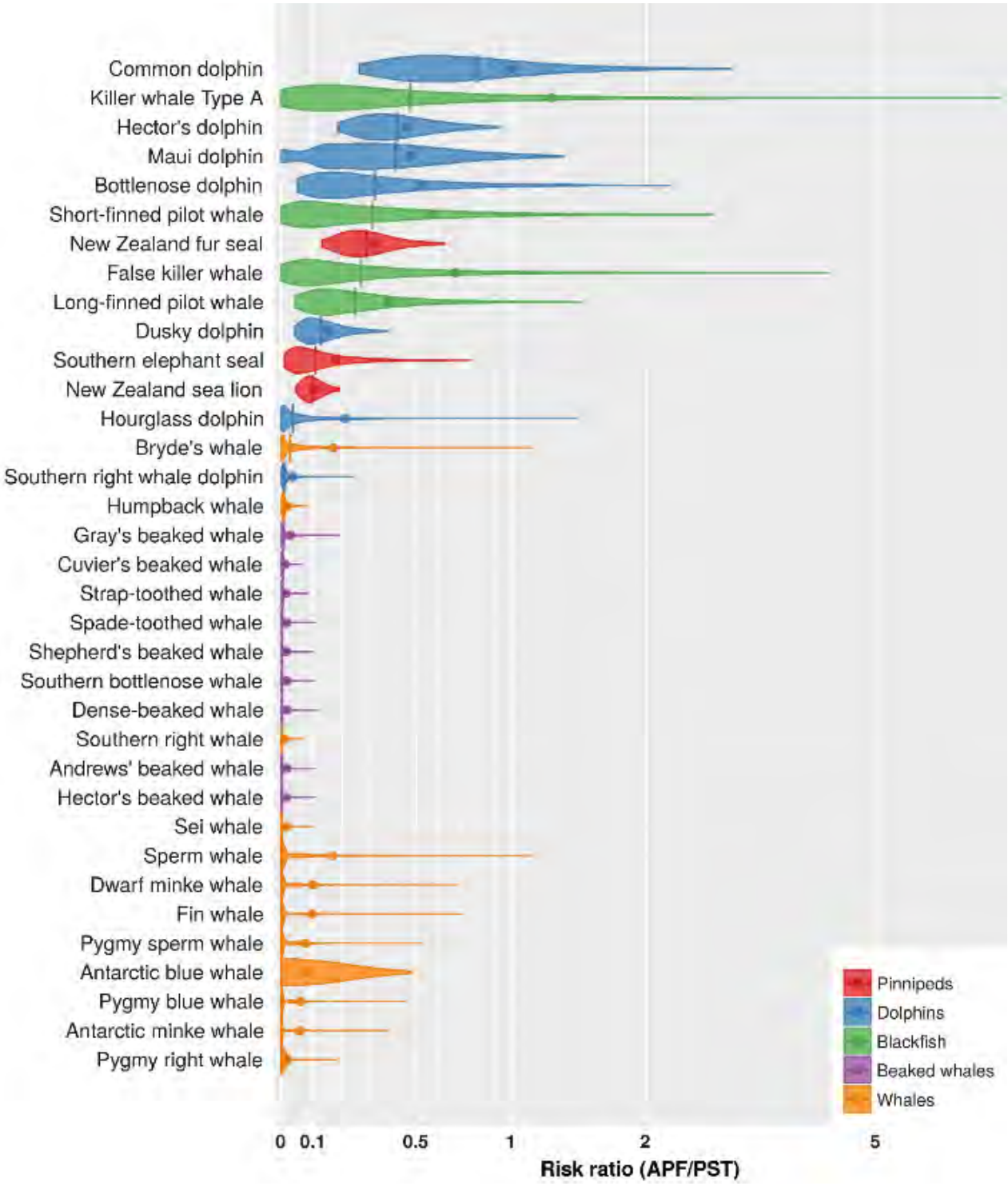


Figure 5.5: Cumulative fishery risk across all fishery groups as estimated by the 2016 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017). Species groups are colour coded.

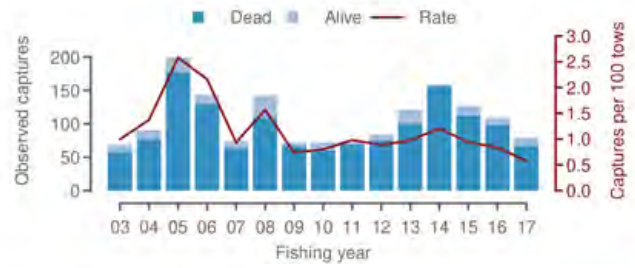
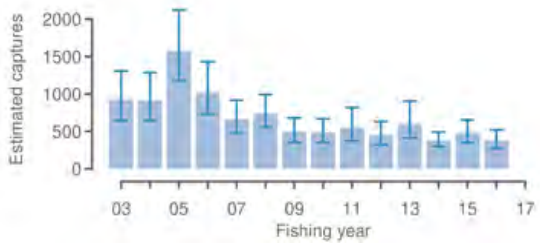
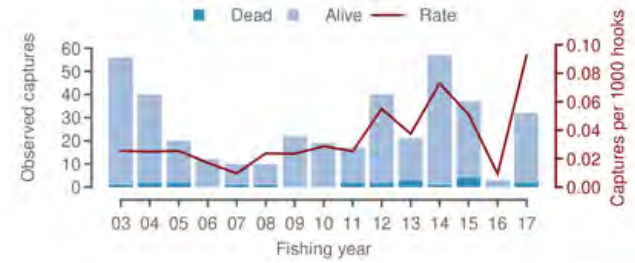
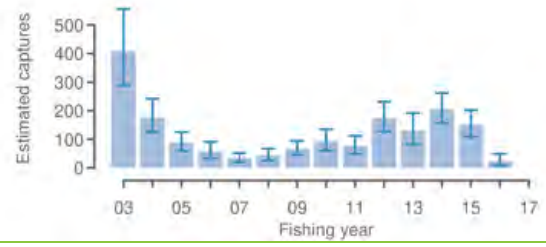
5.4.5 SOURCES OF UNCERTAINTY

Any measure of the effect of New Zealand fur seal mortality from commercial fisheries on New Zealand fur seal populations requires adequate information on the size of the populations at different colonies. Although there is reasonable information about where the main New Zealand fur seal breeding colonies occur, colony size and population dynamics are poorly understood. At present, the main sources of uncertainty are the lack of consistent data on: abundance by colony and in total; population demographic parameters; and at-sea distribution (which would ideally be available at the level of a colony or wider geographic area where several colonies are close together) (Baird 2011). Collation and analysis of existing data, such as that for the west coast South Island, would fill some of these gaps; there is a 20-year time series of pup production from three west coast South Island colonies, a reasonably long data series from the Otago Peninsula, and another from Kaikoura. Maximum benefit could be gained through the use of all available data, as shown by the monitoring of certain colonies of New Zealand fur seals in Australia to provide a measure of overall population stability (see Shaughnessy et al. 1994, Goldsworthy et al. 2003).

Fur seals may forage in waters near a colony or haulout, or may range widely, depending on the sex, age, and individual preferences of the animal (Baird 2011). It is not known whether the New Zealand fur seals around a fishing vessel are from colonies nearby. Some genetic work is proposed to test the potential to differentiate between colonies so that in the future New Zealand fur seals drowned by fishing gear may be identified as being from a certain colony (Robertson & Gemmell 2005).

The low to moderate levels of observer coverage in some fishery-area strata add uncertainty to the total estimated captures. However, the main source of uncertainty in the level of bycatch is the paucity of information from the inshore fishing fleets, which use a variety of gears and methods. Recent increases in observer coverage enabled fur seal capture estimates to include inshore fishing effort. Further increases in coverage, particularly for inshore fisheries, would provide better data on the life stage, sex, and size of captured animals, as well as samples for fatty acid or stable isotope analysis to assess diet and to determine provenance. Information on the aspects of fishing operations that lead to capture in inshore fisheries would also be useful as input to designing mitigation measures.

5.5 INDICATORS AND TRENDS

Population size	Unknown, but potentially ~100 000 in the New Zealand EEZ.[2]
Population trend	Increasing at some mainland colonies but unknown for offshore island colonies. Range is thought to be increasing.
Threat status	New Zealand: Not Threatened, Increasing, Secure Overseas, in 2013.[3] IUCN: Least Concern, in 2015.[4]
Number of interactions	486 estimated captures (95% c.i.: 299–876) in trawl fisheries in 2014–15[6] 179 estimated captures (95% c.i.: 132–237) in surface-longline fisheries in 2014–15[6] 127 observed captures in trawl fisheries in 2014–15[6] 37 observed captures in surface-longline fisheries in 2014–15[6] 949.3 estimated annual potential fatalities (APF) (95% c.i.: 949.3–1 406.5) [7]
Tends in interactions[6]	Trawl fisheries:   Surface-longline fisheries:  

[2] Taylor (1990), Harcourt (2001).

[3] Baker et al. (2016).

[4] Chilvers & Goldsworthy (2015).

[6] For more information, see: http://data.dragonfly.co.nz/psc.

[7] Abraham et al. (2017).

5.6 REFERENCES

Abraham, E R; Berkenbusch, K (2017) Estimated captures of New Zealand fur seal, New Zealand sea lion, common dolphin, and turtles in New Zealand commercial fisheries, 1995–96 to 2014–15. *New Zealand Aquatic Environment and Biodiversity Report No. 188*. 66 p.

Abraham, E R; Berkenbusch, K N; Richard, Y (2010a) The capture of seabirds and marine mammals in New Zealand non-commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 64*. 52 p.

Abraham, E R; Neubauer, P; Berkenbusch, K; Mansfield, R; Knox, C (in prep) Estimating population size and distribution of marine mammals in New Zealand waters using a Delphi survey of expert knowledge. *Draft New Zealand Aquatic Environment and Biodiversity Report*.

Abraham, E R; Richard, Y; Neubauer, P; Berkenbusch, K (2017) Assessment of the risk to New Zealand marine mammals from commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 189*. 123 p.

Abraham, E R; Thompson, F N (2011) Summary of the capture of seabirds, marine mammals, and turtles in New Zealand commercial fisheries, 1998–99 to 2008–09. Final Research Report prepared for Ministry of Fisheries project PRO2007/01. 170 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Abraham, E R; Thompson, F N; Oliver, M D (2010b) Summary of the capture of seabirds, marine mammals, and turtles in New Zealand commercial fisheries, 1998–99 to 2007–08. *New Zealand Aquatic Environment and Biodiversity Report No. 45*. 148 p.

Allum, L; Maddigan, F W (2012) Unusual stability of diet of the New Zealand fur seal (*Arctocephalus forsteri*) at Banks Peninsula, New Zealand. *New Zealand Journal of Marine and Freshwater Research 46*: 91–96.

Baird, S J (Comp.) (1994) Nonfish species and fisheries interactions working group report. *New Zealand Fisheries Assessment Working Group Report 94/1*. Ministry of Fisheries. 54 p. (Unpublished report held in the NIWA library, Wellington.)

Baird, S J (2005) Review of observer comments that relate to captures of New Zealand fur seals (*Arctocephalus forsteri*) during hoki (*Macruronus novaezelandiae*) trawl fishery operations. *NIWA Client Report WLG205-43*. Report prepared for the Hoki Fishery Management Company. 27 p.

Baird, S J (2008) Incidental capture of New Zealand fur seals (*Arctocephalus forsteri*) in longline fisheries in New Zealand waters, 1994–95 to 2005–06. *New Zealand Aquatic Environment and Biodiversity Report No. 20*. 21 p.

Baird, S J (2011) New Zealand fur seals — summary of current knowledge. *New Zealand Aquatic Environment and Biodiversity Report No. 72*. 50 p.

Baird, S J; Bradford, E (2000). Factors that may have influenced the capture of New Zealand fur seals (*Arctocephalus forsteri*) in the west coast South Island hoki fishery, 1991–98. *NIWA Technical Report 92*. 35 p.

Baird, S J; Smith, M H (2007) Incidental capture of New Zealand fur seals (*Arctocephalus forsteri*) in commercial fisheries in New Zealand waters, 2003–04 to 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 14*. 98 p.

Baker, B; Jensz, K; Cawthorn, M; Cunningham, R (2010) Census of New Zealand fur seals on the west coast of New Zealand's South Island. Report to the Deepwater Group Ltd. 22 p. Retrieved from www.latitude-42.com.au.

Baker, C S; Chilvers, B L; Childerhouse, S; Constantine, R; Currey, R; Mattlin, R; van Helden, A; Hitchmough, R; Rolfe, J (2016) Conservation status of New Zealand marine mammals, 2013. *New Zealand Threat Classification Series 14*. Department of Conservation, Wellington. 18 p.

Berkenbusch, K; Abraham, E R; Torres, L G (2013) New Zealand marine mammals and commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 119*. 104 p.

Best, H (2011) West Coast South Island monitoring of New Zealand fur seal pup numbers, 1991–2010 – a summary. Unpublished report for the Department of Conservation. 5 p.

Boren, L J (2005) New Zealand fur seals in the Kaikoura region: colony dynamics, maternal investment and health. A thesis submitted in partial fulfillment of the requirements for a PhD in Biological Sciences, University of Canterbury, University of Canterbury. 261 p.

Boren, L (2010) Diet of New Zealand fur seals (*Arctocephalus forsteri*): a summary. *DOC Research & Development Series 319*. Department of Conservation, Wellington. 19 p.

Boren, L; Morrissey, M; Gemmell, N J (2008) Motor vehicle collisions and the New Zealand fur seal in the Kaikoura region. *Marine Mammal Science 24 (1)*: 235–238.

Boren, L J; Morrissey, M; Muller, C G; Gemmell, N J (2006) Entanglement of New Zealand fur seals in man-made debris at Kaikoura, New Zealand. *Marine Pollution Bulletin 52*: 442–446.

Bradshaw, C J A; Barker, R J; Harcourt, R G; Davis, L S (2003a) Estimating survival and capture probability of fur seal pups using multistate mark-recapture models. *Journal of Mammalogy 84 (1)*: 65–80.

Bradshaw, C J A; Davis, L S; Lalas, C; Harcourt, R G (2000) Geographic and temporal variation in the condition of pups of the New Zealand fur seal (*Arctocephalus forsteri*): evidence for density dependence and differences in the marine environment. *Journal of Zoology 252*: 41–51.

Bradshaw, C J A; Harcourt, R G; Davis, L S (2003b) Male-biased sex ratios in New Zealand fur seal pups relative to environmental variation. *Behavioral Ecology and Sociobiology 53 (5)*: 297–307.

Bradshaw, C J A; Lalas, C; McConkey, S (1998) New Zealand sea lion predation on New Zealand fur seals. *New Zealand Journal of Marine and Freshwater Research 32*: 101–104.

Bradshaw, C J A; Thompson, C M; Davis, L S; Lalas, C (1999) Pup density related to terrestrial habitat use by New Zealand fur seals. *Canadian Journal of Zoology 77 (10)*: 1579–1586.

Carey, P W (1992) Fish prey species of the New Zealand fur seal (*Arctocephalus forsteri* Lesson). *New Zealand Journal of Ecology 16 (1)*: 41–46.

Carey, P W (1998) New Zealand fur seals (*Arctocephalus forsteri*) at the Snares Islands: a stabilised population? *New Zealand Journal of Marine and Freshwater Research 32 (1)*: 113–118.

Chilvers, B L; Goldsworthy, S D (2015) *Arctocephalus forsteri*. The IUCN Red List of Threatened Species 2015: e.T41664A45230026. doi:10.2305/IUCN.UK.2015-2.RLTS.T41664A45230026.en.

Chilvers, B L; Wilson, K-J; Hickling, G J (1995) Suckling behaviours and growth rates of New Zealand fur seals, *Arctocephalus forsteri*, at Cape Foulwind, New Zealand. *New Zealand Journal of Zoology 22*: 263–270.

Clement and Associates (2009) Mitigating incidental captures of fur seals in trawl fisheries. Final Report prepared for Department of Conservation project MIT2006/09. 45 p.

Crawley, M C; Wilson, G J (1976) The natural history and behaviour of the New Zealand fur seal (*Arctocephalus forsteri*). *Tuatara 22 (1)*: 1–29.

Dans, S L; Alonso, M K; Crespo, E A; Pedraza, S N; Garcia, N A (2003) Interactions between marine mammals and high sea fisheries in Patagonia: an integrated approach. *In*: Gales, N; Hindell, M; Kirkwood, R (Eds.), Marine mammals fisheries tourism and management issues. pp 100–115. CSIRO Publishing. 446 p.

Dickie, G S; Dawson, S M (2003) Age, growth, and reproduction in New Zealand fur seals. *Marine Mammal Science 19 (1)*: 173–185.

Dix, B (1993) A new record this century of a breeding colony in the North Island for the New Zealand fur seal *Arctocephalus forsteri*. *Journal of the Royal Society of New Zealand 23*: 1–4.

Dowell, S A; Boren, L J; Negro, S; Muller, C G; Caudron, A K; Gemmell, N J (2008) Rearing two New Zealand fur seal (*Arctocephalus forsteri*) pups to weaning. *Australian Journal of Zoology 56*: 33–39.

Duignan, P J (2003) Disease investigations in stranded marine mammals, 1999–2002. *DOC Science Internal Series 104*. Department of Conservation, Wellington. 32 p.

Duignan, P J; Jones, G W (2007) Autopsy of pinnipeds incidentally caught in commercial fisheries, 2002/03 and 2003/04. *DOC Research & Development Series 280*. Department of Conservation, Wellington. 41 p.

Fea, N I; Harcourt, R; Lalas, C (1999) Seasonal variation in the diet of New Zealand fur seals (*Arctocephalus forsteri*) at Otago Peninsula, New Zealand. *Wildlife Research 26*: 147–160.

Gales, R; Lee, A Y; Pemberton, D; Terauds, A; Irvine, A (2010) Observations of mortality of fur seals between 1998 and 2005 in Tasmania, Australia. *Papers and Proceedings of the Royal Society of Tasmania 144*: 29–35.

Goldsworthy, S D; Bulman, C; He, X; Larcombe, J; Littman, C (2003) Trophic interactions between marine mammals and Australian fisheries: an ecosystem approach. *In*: Gales, N; Hindell, M; Kirkwood, R (Eds.), Marine mammals — fisheries, tourism and management issues. pp 62–99. CSIRO Publishing. 446 p.

Goldsworthy, S; Gales, N (2008) *Arctocephalus forsteri*. The IUCN Red List of Threatened Species. Version 2014.2. IUCN SSC Pinniped Specialist Group. Retrieved from www.iucnredlist.org.

Goldsworthy, S D; Shaughnessy, P D (1994) Breeding biology and haul-out pattern of the New Zealand fur seal, *Arctocephalus forsteri*, at Cape Gantheaume, South Australia. *Wildlife Research 21*: 365–376.

Hamer, D J; Goldsworthy, S D (2006) Seal-fishery operational interactions: Identifying the environmental and operational aspects of a trawl fishery that contribute to bycatch and mortality of Australian fur seals (*Arctocephalus pusillus doriferus*). *Biological Conservation 130*: 517–529.

Harcourt, R G (2001) Advances in New Zealand mammalogy 1990–2000: Pinnipeds. *Journal of the Royal Society of New Zealand 31 (1)*: 135–160.

Harcourt, R G; Bradshaw, C J A; Dickson, K; Davis, L S (2002) Foraging ecology of a generalist predator, the female New Zealand fur seal. *Marine Ecology Progress Series 227*: 11–24.

Harcourt, R G; Schulman, A; Davis, L S; Trillmich, F (1995) Summer foraging by lactating female New Zealand fur seals (*Arctocepbalus forsteri*) off Otago Peninsula, New Zealand. *Canadian Journal of Zoology 73*: 678–690.

Holborrow, J (1999) The diet of New Zealand fur seals (*Arctocephalus forsteri*) in Southern New Zealand. Unpublished MSc thesis, University of Otago, New Zealand.

Hooper, J; Clark, J M; Charman, C; Agnew, D (2005) Sea mitigation measures on trawl vessels fishing for krill in CCAMLR Subarea 48.3. *CCAMLR Science 12*: 195–205.

IUCN (2014) The IUCN Red List of Threatened Species. Version 2014.2. Retrieved from www.iucnredlist.org.

Lalas, C (2008) Recolonisation of Otago, southern New Zealand, by fur seals and sea lions: unexpected patterns and consequences. *In*: Clarkson, B; Kurian, P; Nachowitz, T; Rennie, H (Eds.), Proceedings of the Conser-Vision Conference, University of Waikato. 15 p. Retrieved from www.waikato.ac.nz/wfass/conserv-vision.

Lalas, C; Bradshaw, C J A (2001) Folklore and chimerical numbers: review of a millennium of interaction between fur seals and humans

in the New Zealand region. *New Zealand Journal of Marine and Freshwater Research 35 (3)*: 477–497.

Lalas, C; Harcourt, R (1995) Pup production of the New Zealand fur seal on the Otago Peninsula, New Zealand. *Journal of the Royal Society of New Zealand 25 (1)*: 81–88.

Lalas, C; MacDiarmid, A B (submitted) Rapid recolonisation of south-eastern New Zealand by New Zealand fur seals *Arctocephalus forsteri*. *New Zealand Journal of Marine and Freshwater Research*.

Lalas, C; Murphy, B (1998) Increase in the abundance of New Zealand fur seals at the Catlins, South Island, New Zealand. *Journal of the Royal Society of New Zealand 28 (2)*: 287–294.

Lalas, C; Webster, T (2014) Contrast in the importance of arrow squid as prey of male New Zealand sea lions and New Zealand fur seals at The Snares, subantarctic New Zealand. *Marine Biology 161*: 631–643.

Lento, G M; Mattlin, R H; Chambers, G K; Baker, C S (1994) Geographic distribution of mitochondrial cytochrome b DNA haplotypes in New Zealand fur seals (*Arctocephalus forsteri*). *Canadian Journal of Zoology 72*: 293–299.

Ling, J K (2002) Impact of colonial sealing on seal stocks around Australia, New Zealand and subantarctic islands between 150 and 170 degrees East. *Australian Mammalogy 24*: 117–126.

Lonergan, M E; Phillips, R A; Thomson, R B; Zhou, S (2017) Independent review of New Zealand's Spatially Explicit Fisheries Risk Assessment approach – 2017. *New Zealand Fisheries Science Review 2017/2*. 36 p.

Mattlin, R H (1978a) Population biology, thermoregulation and site preference of the New Zealand fur seal (*Arctocephalus forsteri* (Lesson, 1828) on the Open Bay Islands, New Zealand. Unpublished PhD Thesis, University of Canterbury, Christchurch.

Mattlin, R H (1978b) Pup mortality of the New Zealand fur seal (*Arctocephalus forsteri* Lesson). *New Zealand Journal of Ecology 1*: 138–144.

Mattlin, R H (1981) Pup growth of the New Zealand fur seal, *Arctocephalus forsteri,* on the Open Bay Islands, New Zealand. *Journal of Zoology (London) 193*: 305–314.

Mattlin, R H (1987) New Zealand fur seal, *Arctocephalus forsteri*, within the New Zealand region. *In*: Croxall, J P; Gentry, R L, Status, biology, and ecology of fur seals. Proceedings of an international symposium and workshop, Cambridge, England, 23–27 April 1984. *NOAA Technical Report NMFS-51*.

Mattlin, R H (1994a) Incidental catch of fur seals in the west coast South Island hoki trawl fishery, 1989–92. *New Zealand Fisheries Assessment Research Document 93/19*. 18 p. (Unpublished report held at NIWA library, Wellington.)

Mattlin, R H (Comp. & Ed.) (1994b) Seals and sea birds–fisheries interactions: report of a workshop, Wellington, 1992. *New Zealand Fisheries Occasional Publication No. 8*. 18 p. plus appendices.

Mattlin, R H; Gales, N J; Costa, D P (1998) Seasonal dive behaviour of lactating New Zealand fur seals (*Arctocephalus forsteri*). *Canadian Journal of Zoology 76 (2)*: 350–360.

Mellina, E; Cawthorn, M (2009) New Zealand fur seal (*Arctocephalus forsteri*) population assessment in Fiordland Sounds. Presented to AEWG 7 July 2009, Wellington, New Zealand. 31 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Meynier, L; Chilvers, B L; Muller, C; Virgili, A; McGill, R; Bury, S; Willis, T; Morel, P C H (2013) Combining bio-logging with stable isotope and fatty acid analyses to assess the foraging habits of New Zealand fur seals, west coast, New Zealand. Oral presentation. Proceedings of the 20th Biennial Conference on the Biology of Marine Mammals, December 2013, Dunedin, New Zealand.

Miller, E H (1975) Body and organ measurements of fur seals, *Arctocephalus forsteri* (Lesson), from New Zealand. *Journal of Mammalogy 56 (2)*: 511–513.

Ministry for Primary Industries (2014) Annual Operational Plan for Deepwater Fisheries for 2013/14. *MPI Technical Paper No. 2013/52*. 66 p.

Moore, J E; Wallace, B P; Lewison, R L; Žydelis, R; Cox, T M; Crowder, L B (2009) A review of marine mammal, sea turtle and seabird bycatch in USA fisheries and the role of policy in shaping management. *Marine Policy 33*: 435–451.

Mormede, S; Baird, S J; Smith, M H (2008) Factors that may influence the probability of fur seal capture in selected New Zealand fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 19*. 42 p.

Norman, F I (2000) Preliminary investigation of the bycatch of marine birds and mammals in inshore commercial Fisheries, Victoria, Australia. *Biological Conservation 92*: 217–226.

Read, A J; Drinker, P; Northridge, S (2006) Bycatch of marine mammals in U.S. and global fisheries. *Conservation Biology 20 (1)*: 163–169.

Richard, Y; Abraham, E R; Filippi, D (2011) Assessment of the risk to seabird populations from New Zealand commercial fisheries. Final Research Report for research projects IPA2009-19 and IPA2009-20. 66 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Richards, R (1994) 'The upland seal' of the Antipodes and Macquarie Islands: a historian's perspective. *Journal of the Royal Society of New Zealand 24*: 289–295.

Roberts, J; Best, H (2016) Annual pup production & individual size of New Zealand fur seal pups on the West Coast South Island. NIWA Client Report.

Robertson, B C; Gemmell, N J (2005) Microsatellite DNA markers for the study of population structure in the New Zealand fur seal

Arctocephalus forsteri. *DOC Science Internal Series 196*. Department of Conservation, Wellington. 18 p.

Rowe, S J (2007) A review of methodologies for mitigating incidental catch of protected marine mammals. *DOC Research and Development Series 283*. 47 p. Science and Technical Publishing, Department of Conservation, Wellington.

Rowe, S J (2009) Conservation Services Programme observer report: 1 July 2004 to 30 June 2007. *DOC Marine Conservation Services Series 1*. Department of Conservation, Wellington. 93 p.

Shaughnessy, P D (1999) The action plan for Australian seals. Environment Australia. 62 p.

Shaughnessy, P D; Davenport, S R (1996) Underwater videographic observations and incidental mortality of fur seals around fishing equipment in south-eastern Australia. *Marine and Freshwater Research 47*: 553–556.

Shaughnessy, P D; Gales, N J; Dennis, T E; Goldsworthy, S D (1994) Distribution and abundance of New Zealand fur seals, *Arctocephalus forsteri*, in South Australia and Western Australia. *Wildlife Research 21 (6)*: 667–695.

Shaughnessy, P D; Payne, A I L (1979) Incidental mortality of Cape fur seals during trawl fishing activities in South African waters. *Fisheries Bulletin, South Africa 12*: 20–25.

Smith, I W G (1989) Maori impact on the marine megafauna: pre-European distributions of New Zealand sea mammals. *In*: Sutton, D G (Ed.), 'Saying so doesn't make it so', papers in honour of B. Foss Leach. pp 76–108. New Zealand Archaeological Association, Dunedin.

Smith, I W G (2005) Retreat and resilience: fur seals and human settlement in New Zealand. *In*: Monks, G (Ed.), The Exploitation and Cultural Importance of Sea Mammals. pp 6–18. Oxbow Books, Oxford. 173 p.

Smith, I W G (2011) Estimating the magnitude of pre-European Maori marine harvest in two New Zealand study areas. *New Zealand Aquatic Environment and Biodiversity Report No. 82*. 71 p.

Smith, M H; Baird, S J (2009) Model-based estimation of New Zealand fur seal (*Arctocephalus forsteri*) incidental captures and strike rates for trawl fishing in New Zealand waters for the years 1994–95 to 2005–06. *New Zealand Aquatic Environment and Biodiversity Report No. 40*. 90 p.

Sorensen, J H (1969) New Zealand seals with special reference to the fur seal. *Fisheries Technical Report No. 39*. New Zealand Marine Department. 35 p.

Suisted, R; Neale, D M (2004) Department of Conservation Marine Mammal Action Plan for 2005–2010. Department of Conservation, Wellington.

Taylor, R H (1990) Records of subantarctic fur seals in New Zealand (Note). *New Zealand Journal of Marine and Freshwater Research 24*: 499–502.

Taylor, R H (1996) Distribution, abundance and pup production of the New Zealand fur seal (*Arctocephalus forsteri* Lesson) at the Bounty Islands. *Science for Conservation No. 32*. 14 p.

Taylor, R H; Barton, K J; Wilson, P R; Thomas, B W; Karl, B J (1995) Population status and breeding of New Zealand fur seals (*Arctocephalus forsteri*) in the Nelson-northern Marlborough region. *New Zealand Journal of Marine and Freshwater Research 29 (2)*: 223–234.

Thompson, F N; Abraham, E R (2010) Estimation of fur seal (*Arctocephalus forsteri*) bycatch in New Zealand trawl fisheries, 2002–03 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 61*. 37 p.

Thompson, F N; Abraham, E R; Berkenbusch, K (2011) Marine mammal bycatch in New Zealand trawl fisheries, 1995–96 to 2009–10. Draft Final Research Report for Ministry for Primary Industries project PRO2010-01. 80 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Thompson, F N; Abraham, E R; Berkenbusch, K (2012) Marine mammal bycatch in New Zealand trawl fisheries, 1995–96 to 2010–11. Final Research Report for Ministry for Primary Industries project PRO2010-01. 90 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Townsend, A J; de Lange, P J; Duffy, C A J; Miskelly, C M; Molloy, J; Norton, D A (2008) New Zealand Threat Classification System manual. Department of Conservation, Wellington. 35 p.

Troy, S K; Mattlin, R; Shaughnessy, P D; Davie, P S (1999) Morphology, age and survival of adult male New Zealand fur seals, *Arctocephalus forsteri*, in South Australia. *Wildlife Research 26*: 21–34.

Willis, T J; Triossi, F; Meynier, L (2008) Diet of fur seals *Arctocephalus forsteri* at Tonga Island, Abel Tasman National Park. *NIWA Client Report NEL2008-011*, prepared for the Department of Conservation. 12 p.

Wilson, G J (1981) Distribution and abundance of the New Zealand fur seal, *Arctocephalus forsteri*. *Fisheries Research Division Occasional Publication No. 20*. 39 p.

6 HECTOR'S DOLPHIN (CEPHALORHYNCHUS HECTORI HECTORI) AND MĀUI DOLPHIN (C. H. MAUI)

Status of chapter	This chapter has not been updated for AEBAR 2018. A substantial body of new science was undertaken in 2017-19 to inform the update of the Hector's and Maui dolphin Threat Management Plan in 2019, but the new information was not finalized in time for inclusion in this chapter. Consequently some information in this chapter may be out of date. The most up to date science is described comprehensively in Roberts et al. (2019).
Scope of chapter	This chapter briefly describes: the biology of Hector's dolphin (*Cephalorhynchus hectori hectori*) and Māui dolphin (*C. h. maui*); the nature and extent of potential interactions with fisheries;, management of fisheries interactions; means of estimating fisheries impacts and population level risk; and remaining sources of uncertainty, to guide future work.
Area	West Coast North Island; all coastal areas of South Island.
Focal localities	Areas where significant fisheries interactions are known to have occurred include waters over or close to the continental shelf surrounding the South Island and the west coast of the North Island.
Key issues	Improved estimation of Hector's and Māui dolphin spatio-temporal distributions affecting spatial overlap with fisheries, including in low-density areas; improved population size estimates for the South Coast South Island (SCISI) Hector's dolphin population; improved estimation of demographic parameters affecting potential population growth rates.
Emerging issues	Improved ability to assess risk and apply risk management solutions on a regional subpopulation basis, or at finer spatial and temporal scales; improved understanding of non-fishery threats including disease.
MPI research (current)	PRO2017-12 *Hector's and Māui dolphin risk assessment to support review of the TMP;* PRO2016-09 *Abundance and distribution of Hector's and Māui dolphins on South Coast South Island;* PRO2015-04 *Addressing key information gaps for Māui dolphins*; PRO2014-01 *Improved estimation of the distribution of seabirds and marine mammals (cetaceans);* PRO2013-09 *Population viability of Māui dolphins;* PRO2009-01C *Abundance, distribution and productivity of Hector's (and Māui) dolphins* (ECSI survey).
NZ government research (current)	DOC Marine Conservation Services Programme (CSP): MIT2012-03 *Review of mitigation techniques in set net fisheries*; INT2013-01 *To understand the nature and extent of protected species interactions with New Zealand commercial fishing activities*; INT2013-03 *To determine which marine mammal, turtle and protected fish species are captured in fisheries and their mode of capture*; INT2013-04 *To review the data collected by fisheries observers in relation to understanding the interaction with protected species, and refine efficient protocols for future data collection*; Additional conservancy-level work including aerial and boat surveys in Taranaki, genetic sampling and necropsies of recovered animals.
Other research[1]	Otago University: Long term study of Hector's dolphins at Banks Peninsula, including distribution and abundance, survival rates, reproductive rates, movements, feeding ecology. Auckland University: Population monitoring of Māui dolphins and population genetics of Hector's and Māui dolphins. Massey University: Necropsy of recovered Hector's/Māui dolphins.
Related chapters/issues	Chapter 3 (SEFRA); Chapters 4–5 (sea lions and fur seals); Chapter 7 (common dolphins)

6.1 CONTEXT

Hector's and Māui dolphin [2] (*Cephalorhynchus hectori*), comprising the South Island subspecies referred to as Hector's dolphin (*C. h. hectori*) and the North Island subspecies known as Māui dolphin (*C. h. maui*), is endemic to the coastal waters of New Zealand. Like most other small cetaceans, the species is at risk of fishing-related mortality

[1] Du Fresne et al. (2012) compiled a bibliography of all Hector's and Māui dolphin research completed since 2003 (http://www.doc.govt.nz/documents/science-and-technical/drds332entire.pdf).

[2] In this document, 'Hector's dolphin(s)' refers to the South Island subspecies (*Cephalorhynchus hectori hectori*), while 'Māui dolphin(s)' refers to the North Island subspecies (*C. hectori maui*). 'Hector's and Māui dolphin(s)' refers to both subspecies collectively (*C. hectori*). This approach is taken to avoid confusion and enable distinction between the South Island subspecies and the species as a whole.

(e.g., Read et al. 2006, Reeves et al. 2013a, 2013b, Geijer & Read 2013).

Hector's and Māui dolphin was gazetted as a 'threatened species' by the Minister of Conservation in 1999 and is defined as a 'protected species' according to part 1, s2(1) of the Fisheries Act 1996 and s2(1) of the Marine Mammals Protection Act (MMPA) 1978. Management of fisheries impacts on Hector's and Māui dolphins is legislated under both these acts. The MMPA (1978) allows for the approval of a population management plan for any protected species, within which a maximum allowable level of fishing-related mortality may be imposed. For threatened species, this level 'should allow the species to achieve non-threatened status as soon as reasonably practicable, and in any event within a period not exceeding 20 years' (MMPA 1978, p.11). If a population management plan has been approved, the Fisheries Act (1996) requires that all reasonable steps be taken to ensure that the maximum allowable level of fishing-related mortality is not exceeded, and the Minister may take other measures necessary to further avoid, remedy, or mitigate any adverse effects of fishing on the relevant protected species. In the absence of a population management plan, 'the Minister may, after consultation with the Minister of Conservation, take such measures as he or she considers are necessary to avoid, remedy, or mitigate the effect of fishing-related mortality on any protected species, and such measures may include setting a limit on fishing-related mortality' (Fisheries Act 1996, p.66).

The latest DOC Marine Mammal Action Plan[3] (DOC MMPA; Suisted & Neale 2004) stated that actions required include:

- *'Prepare species plans for both Hector's and Maui's dolphins'*
- *'Consider preparation of Population Management Plans (PMP) for Hector's and Maui's dolphins in accordance with the legal process and the species plans.'*

However, to date no population management plan (PMP) has been produced for Hector's or Māui dolphin and no maximum allowable level of fishing-related mortality has been set. A draft threat management plan (TMP) for Hector's and Māui dolphin was developed jointly by the Department of Conservation (DOC) and the former Ministry of Fisheries (MFish) in 2007. The TMP is not a statutory document, but a management plan identifying human-induced threats to Hector's and Māui dolphin populations and outlining strategies to mitigate those threats. The stated goals of the TMP (DOC & MFish 2007) are:

- *'To ensure the long-term viability of Hector's and Maui's dolphins is not threatened by human activities; and*
- *'To further reduce impacts of human activities as far as possible, taking into account advances in technology and knowledge, and financial, social and cultural implications.'*

These goals were re-stated in the consultation paper on the Review of the Māui dolphin portion of the TMP published in 2012 (MPI & DOC 2012). The review of the Māui portion of the TMP provided a comprehensive overview of information relating to the biology, distribution, threats to, and management of Māui dolphins. To inform the review of the Māui dolphin TMP, a spatially explicit, semi-quantitative risk assessment was conducted using an expert panel, applying an early modification of the SEFRA method (Chapter 3), to identify, analyse and evaluate all threats to Māui dolphins (Currey et al. 2012). The process involved expert panelists mapping dolphin distribution, identifying and characterising threats, scoring the likely impact of each threat, and subsequent quantitative analysis to estimate risk posed by threats. Where these remain relevant, the results of this process are described in the relevant sections below.

A review of the full TMP will begin in 2018. The review will be informed by a comprehensive spatially explicit risk assessment including fisheries and non-fishery threats to Hector's and Māui dolphins, and demographic population models for separate regional subpopulations (MPI project PRO2017-12). The risk assessment will consider updated estimates of population size, updated demographic parameters affecting population growth and recovery potential, and improved estimates of the distribution of the

[3] DOC has confirmed that the Marine Mammal Action Plan for 2005–10 still reflects DOC's priorities for marine mammal conservation.

dolphins to better estimate spatial overlap with threats, adapting methods described in Chapter 3. This information will be used to reassess the risk of fishing- and non-fishing-related threats across Hector's and Māui dolphin subpopulations, to evaluate the effectiveness of the management measures and monitoring programmes currently in place, and to consider the effects of alternative risk management scenarios where these may be required.

6.2 BIOLOGY

6.2.1 TAXONOMY

Hector's and Māui dolphin is one of four species in the genus *Cephalorhynchus*, which are all restricted to cool, temperate, coastal waters in the southern hemisphere. On the basis of morphological differences, and genetic information indicative of reproductive isolation, Hector's and Māui dolphin was divided into two subspecies; Hector's dolphin around the South Island (41˚S to 47˚S) and Māui dolphin, on the west coast of the North Island (WCNI, 36°S to 40°S; Baker et al. 2002). The reproductive isolation of the Māui subspecies is supported by a more recent genetic analysis with a larger sample size (Hamner et al. 2012a) despite genetic analyses having located four Hector's dolphins off the WCNI (Hamner et al. 2014).

6.2.2 DISTRIBUTION

Hector's dolphins are most frequently sighted on the west coast of the South Island (WCSI) between Jackson Bay and Kahurangi Point (Bräger & Schneider 1998, Rayment et al. 2011a), on the east coast (ECSI) between the Marlborough Sounds and Otago Peninsula (Dawson et al. 2004, MacKenzie & Clement 2014) and on the south coast (SCSI) between Toetoes Bay and Porpoise Bay and in Te Waewae Bay (Bejder & Dawson 2001, Dawson et al. 2004). Current population densities are lower in the intervening stretches of coast, e.g., Fiordland (Bräger & Schneider 1998), Golden Bay (Slooten et al. 2001) and the south Otago coast (Jim Fyfe, pers. comm.), resulting in a fragmented distribution. There is significant genetic differentiation among the west, east and south coast populations, with little or no gene flow connecting them (Pichler et al. 1998, Pichler 2002, Hamner et al. 2012a). The observed levels of genetic divergence over such small distances are unusual among cetaceans, especially considering the absence of geographical barriers (Pichler et al. 1998). These genetic differences are thought to result from individuals having small home ranges and high philopatry (Pichler et al. 1998, Bräger et al. 2002, Rayment et al. 2009b). For example, the mean lifetime alongshore home range of the 20 most frequently sighted dolphins at Banks Peninsula was 49.7 km (s.e.: 5.29; ranging from 13.60 km to 101.43 km for individual dolphins) for the period 1985 to 2006 (Rayment et al. 2009b).

Satellite tagging of three Hector's dolphins off the Banks Peninsula in 2004 indicated maximum distances between locations of 50.9 to 66.5 km over deployments lasting from four to seven months (Stone et al. 2005). For photo-identified dolphins, Rayment et al. (2009a) reported distances between extreme sightings for 53 dolphins ranging from 9.34 km to 107.38 km for the period 1985–2006.

Genetic testing of dolphins off the WCNI since 2001 has identified a small number of Hector's dolphins located within the contemporary distribution of Māui dolphin as far north as the Manukau Harbour. These results raise the possibility of at least occasional long distance dispersal by Hector's dolphins (Hamner et al. 2012b). Although some of these dolphins were found in association with Māui dolphins there is currently no evidence of interbreeding (Hamner et al. 2014). Some of the Hector's dolphins sampled on the WCNI could not be unambiguously assigned to one of the three South Island Hector's dolphin populations leading Hamner et al. (2014) to raise the possibility that they may represent a hitherto unsampled population of Hector's dolphins, or indicate interbreeding between the ESCI and WCSI populations.

Māui dolphins are most frequently sighted between Maunganui Bluff and New Plymouth (Slooten et al. 2005, Du Fresne 2010, Hamner et al. 2012a, 2012b). Research surveys since 2003 have sighted Māui dolphins between Kaipara Harbour and Kawhia (Slooten et al. 2005, Du Fresne 2010, Hamner et al. 2012a, 2012b). Historical samples from strandings and museum specimens have allowed genetic identification of Māui dolphins on the WCNI from Dargaville to Wellington (DOC 2017a, 2017b, Hamner, pers. comm.); however there are doubts as to the provenance of a record of a Māui dolphin attributed to the Bay of Islands (Hamner, pers. comm.).

There are reported public sightings of Hector's and Māui dolphins from all around the North Island coast, including the Bay of Islands, Hauraki Gulf, Coromandel Peninsula, Hawkes Bay, Wairarapa and Kapiti Coast (Baker 1978, Cawthorn 1988, Russell 1999, DOC 2017a). Pichler & Baker (2000) reported genetic analysis of samples of Hector's and

Māui dolphins dating back to 1870 and suggest that abundance has declined and geographic range has contracted over the past 140 years. It has also been suggested that Māui dolphin's range has contracted off the WCNI in recent history coincident with a decline in abundance (MPI & DOC 2012).

Small-scale movements by Māui dolphins over up to 80 km of coastline have been revealed by repeated genetic sampling of the same individuals (mean distance between the two most extreme locations for the six individuals sampled at least three times = 35.5 km; s.e.: 4.03 km; Oremus et al. 2012).

Hector's and Māui dolphin densities are highest close to the coast throughout the year. Bräger et al. (2003) used resource selection models to show that Hector's dolphins have a preference for shallow, turbid waters. During systematic aerial surveys on the WCSI (Rayment et al. 2011a, MacKenzie & Clement 2016), east coast (MacKenzie & Clement 2014, Figure 6.2 and Figure 6.3), at Banks Peninsula (Rayment et al. 2010), in Cloudy and Clifford Bays (DuFresne & Mattlin 2009) and on the North Island west coast (Slooten et al. 2005) most sightings were in water depths less than 100 m (e.g., Figure 6.2 and Figure 6.3). Occasional sightings are made beyond the 100 m isobath (e.g., DuFresne & Mattlin 2009, MacKenzie & Clement 2014). Varying bathymetry among these locations meant that most sightings were within 6 nm offshore of the WCSI (Rayment et al. 2011a, MacKenzie & Clement 2016), yet extended at least out to 20 nm from the coast at Banks Peninsula (MacKenzie & Clement 2014). In both these areas, distance offshore best explained dolphin distribution, possibly due to declining prey availability with increasing distance from the coast (Rayment et al. 2010, 2011a). At Banks Peninsula, there was a significant seasonal difference in distribution, with a greater proportion of dolphins close to shore in summer than winter (Rayment et al. 2010, MacKenzie & Clement 2014), a conclusion consistent with nearshore boat-based surveys (e.g., Dawson & Slooten 1988, Bräger 1998) and passive acoustic monitoring (Rayment et al. 2009a). However, the furthest offshore sighting distances were similar in summer and winter (Rayment et al. 2010, MacKenzie & Clement 2014, MacKenzie & Clement 2016). From analysis of passive acoustic data, Dawson et al. (2013a) suggested that Hector's dolphins' use of an inner harbour site in Akaroa Harbour was greater than expected in winter, and that habitat selection was affected by time of day and state of the tide. No such seasonal difference in Hector's dolphin distribution was detected during aerial surveys on the South Island west coast (Rayment et al. 2011a, MacKenzie & Clement 2016).

New aerial surveys of the South Coast South Island population are contracted for the summer of 2017-18 (MPI project PRO2016-09) to reduce uncertainty in the estimation of population size for the SCSI Hector's dolphin subpopulation.

The highest density of Māui dolphins occurs inshore (within 4 nm of the coast) between Manukau Harbour and Port Waikato (Slooten et al. 2005, MPI & DOC 2012, Oremus et al. 2012). Sightings are occasionally made beyond 4 nm from the coast, extending at least to 7 nm offshore (Du Fresne 2010, Thompson & Richard 2012). Sightings of Māui dolphins have been made in three North Island harbours (Kaipara, Manukau and Raglan; see review in Slooten et al. 2005). Passive acoustic monitoring of these three harbours, in addition to Kawhia Harbour, revealed a low-level of episodic use of Kaipara and Manukau Harbours (Rayment et al. 2011b).

A map of Māui dolphin distribution[4] was developed as part of the Māui dolphin risk assessment (Currey et al. 2012). The distribution was generated via generalised additive modelling (Thompson & Richard 2012) of systematic survey data (Ferreira & Roberts 2003, Slooten et al. 2005, 2006, Scali 2006, Rayment & du Fresne 2007, Childerhouse et al. 2008, Stanley 2009, Hamner et al. 2012a) and modification

[4] The map of Māui dolphin distribution was produced using data that included sightings of unknown subspecies identity (e.g., from aerial surveys). Hector's dolphins have been detected off the WCNI. However, they comprised just 4 of the 91 animals genetically identified within the area of mapped distribution since 2001 (two living females, one dead female, one dead male; Hamner et al. 2012a, 2013). The two living Hector's dolphins were found in association with Māui dolphins and three of four dolphins were found in or near Manukau Harbour, close to the core of Māui dolphin distribution (Figure 6.1). Given that the proportion of Hector's dolphins is likely to be small and there was no evidence to suggest that their inclusion would bias the distribution, the risk assessment proceeded with this map on the basis that it provided the best estimate of Māui dolphin distribution available.

to incorporate expert panel feedback regarding the alongshore, offshore and inshore extent (Figure 6.1; see Currey et al. 2012 for further details).

Preliminary results of work currently in progress under project PRO2014-01 suggest that Hector's and Māui dolphin distributions may be more effectively modeled, including potential seasonal changes, using multivariate models fitted to spatial environmental data layers, in particular satellite-derived estimates of turbidity, and/or modeled species distributions of preferred prey. This work will be expanded under project PRO2017-12 and used to inform the update of the Hector's and Māui dolphin Threat Management Plan in 2018.

6.2.3 FORAGING ECOLOGY

Miller et al. (2013) investigated the diet of Hector's and Māui dolphins through the examination of diagnostic prey remains in the stomachs of 63 incidentally captured and beach-cast animals. They concluded that Hector's dolphins take a wide variety of prey throughout the water column (in total 29 taxa were recorded), but that the diet is dominated by a few mid-water and demersal species, particularly red cod (*Pseudophycis bachus*), ahuru (*Auchenoceros punctatus*), arrow squid (*Notodarus* sp.), sprat (*Sprattus* sp.), sole (*Peltorhamphus* sp.) and stargazer (*Crapatulus* sp.). Prey items ranged from an estimated 0.5–60.8 cm in length, but the majority were less than 10 cm in length, indicating that the juveniles of some species were targeted (Miller et al. 2013). The diets of Hector's dolphins from the South Island west and east coasts were significantly different, due largely to the importance of javelinfish (*Lepidorhynchus denticulatus*) on the west coast, and a greater consumption of demersal prey species on the east coast (Miller et al. 2013). Only two samples were derived from Hector's/Māui dolphins off the WCNI, containing only red cod, ahuru, sole and flounder (*Rhomboselea* sp.; Miller et al. 2013). The stomachs of the six smallest dolphins in the sample (standard length under 90 cm) contained only milk, while the next largest (99 cm standard length) contained milk and remains of arrow squid (Miller et al. 2013). Milk was not found in the stomachs of any dolphins longer than 107 cm (Miller et al. 2013).

Hector's dolphins have been observed foraging in association with demersal trawlers at Banks Peninsula, presumably targeting the fish disturbed but not captured by the trawl net (Rayment & Webster 2009). Hector's dolphins are occasionally seen foraging near the sea surface on small fish including sprat, pilchard (*Sardinops neopilchardus*) and yellow-eyed mullet (*Aldrichetta forsteri;* Miller et al. 2013), sometimes in association with white-fronted terns (*Sterna striata*; Bräger 1998). The seasonal changes in distribution of Hector's dolphins at Banks Peninsula described above are presumed to be in response to seasonal movements of their prey species (Rayment et al. 2010), many of which migrate into shallower nearshore waters in the summer months (Paul 2000).

6.2.4 REPRODUCTIVE BIOLOGY

Incidentally captured and stranded Hector's dolphins have provided information on the life history and reproductive parameters of the species. Males reach sexual maturity between six and nine years of age, and females have their first calf between seven and nine years old (Slooten 1991). Examination of the ultrastructure of the teeth from these necropsied animals revealed that females live to at least 19 years (n = 33) and males (n = 27) to at least 20 (Slooten 1991). Photo-ID studies have provided additional data and revealed that the calving interval is two to four years (Slooten 1990) and that longevity is at least 22 years (Rayment et al. 2009b, Webster et al. 2009). Gormley (2009) extended these analyses, estimating mean female fecundity of Hector's dolphins off Banks Peninsula at 0.205 female offspring per capita per annum (s.d.: 0.050) and mean age at first reproduction at 7.5 years (s.d.: 0.42).

Calves are typically born during spring and early summer, with neonatal length estimated to be 60–75 cm (Slooten & Dawson 1994). Calves stay with their mothers for at least one year, more usually two, and the mother does not appear to conceive again until the calf is independent (Slooten & Dawson 1994). Application of the growth models produced by Webster et al. (2010) to the diet data obtained by Miller et al. (2013) suggests that weaning occurs between one and two years of age. Growth is rapid and asymptotic length is reached in 5–6 years (Webster et al. 2010). Sexually mature adults usually fall within the range of 119–145 cm total length and at maturity females are approximately 10 cm longer than males (Slooten & Dawson 1994, Webster et al. 2010). In a sample of 66 female and 100 male known-age Hector's dolphins, the maximum total length measurements were 145 cm and 132 cm respectively (Webster et al. 2010). Māui dolphins are significantly longer than Hector's dolphins, with a maximum recorded total length of 162 cm (Russell 1999). Hector's and Māui dolphins are typically found in small groups of 1–14 individuals (Slooten et al. 2006, Rayment et al. 2010,

2011b, Oremus et al. 2012). Mean group sizes appear to be larger when estimated from boat-based surveys (e.g., Webster et al. 2009, Oremus et al. 2012) compared with aerial surveys (e.g., Slooten et al. 2006, Rayment et al. 2010) possibly due to the species' boat-positive behaviour (e.g., Dawson et al. 2004). Webster et al. (2009) found that Hector's dolphin groups were highly segregated by sex, with 91% of groups of up to five individuals being all male or all female. Using molecular sexing techniques, Oremus et al. (2012) found no evidence of sexual segregation in groups of fewer than eight Māui dolphins. The social organisation of Hector's dolphin groups is characterised by fluid association patterns, with little stability over periods longer than a few days (Slooten et al. 1993). Together with observations of sexual behaviour (Slooten 1990) and the relatively large testis size of males (Slooten 1991), this suggests that Hector's dolphins have a promiscuous mating system, in which males seek encounters with multiple females rather than attempting to monopolise them (Slooten et al. 1993).

These life-history characteristics mean that Hector's dolphins, like many other small cetaceans (Perrin & Reilly 1984), have a low intrinsic population growth rate. Using matrix population models, asymptotic population growth rate for Hector's dolphins was estimated to be -4.2 to +4.9% per year for survivorship schedules based on other mammals (Slooten & Lad 1991). The authors considered that a growth rate of 1.8% was a plausible 'best case' scenario for Hector's dolphin (Slooten & Lad 1991). Estimates of the intrinsic rate of increase from matrix models are sensitive to the particular parameters chosen (Slooten & Lad 1991, Gormley et al. 2012, Baker et al. 2013).

Updated information to estimate relevant demographic parameters for Hector's and Māui dolphins will be considered in the review of the TMP in 2018.

6.2.5 POPULATION BIOLOGY

The earliest survey-based abundance estimate for Hector's and Māui dolphin (3408 animals with a suggested range of 3000 to 4000) was obtained via small boat-based strip transects surveys (Dawson & Slooten 1988; Table 6.1). These surveys were primarily focused on assessing alongshore distribution rather than abundance. Consequently survey effort was concentrated within 800 m of shore and calibrated with a limited number of 5 nm offshore transects. Nationwide line transect surveys of Hector's and Māui dolphin were carried out between 1997 and 2004 (Dawson et al. 2004, Slooten et al. 2004, 2006). These resulted in a population estimate for Hector's dolphin around the South Island and offshore to 4 nm of 7270 (CV = 16%; Slooten et al. 2004) and for Māui dolphin of 111 (CV = 44%; Slooten et al. 2006).

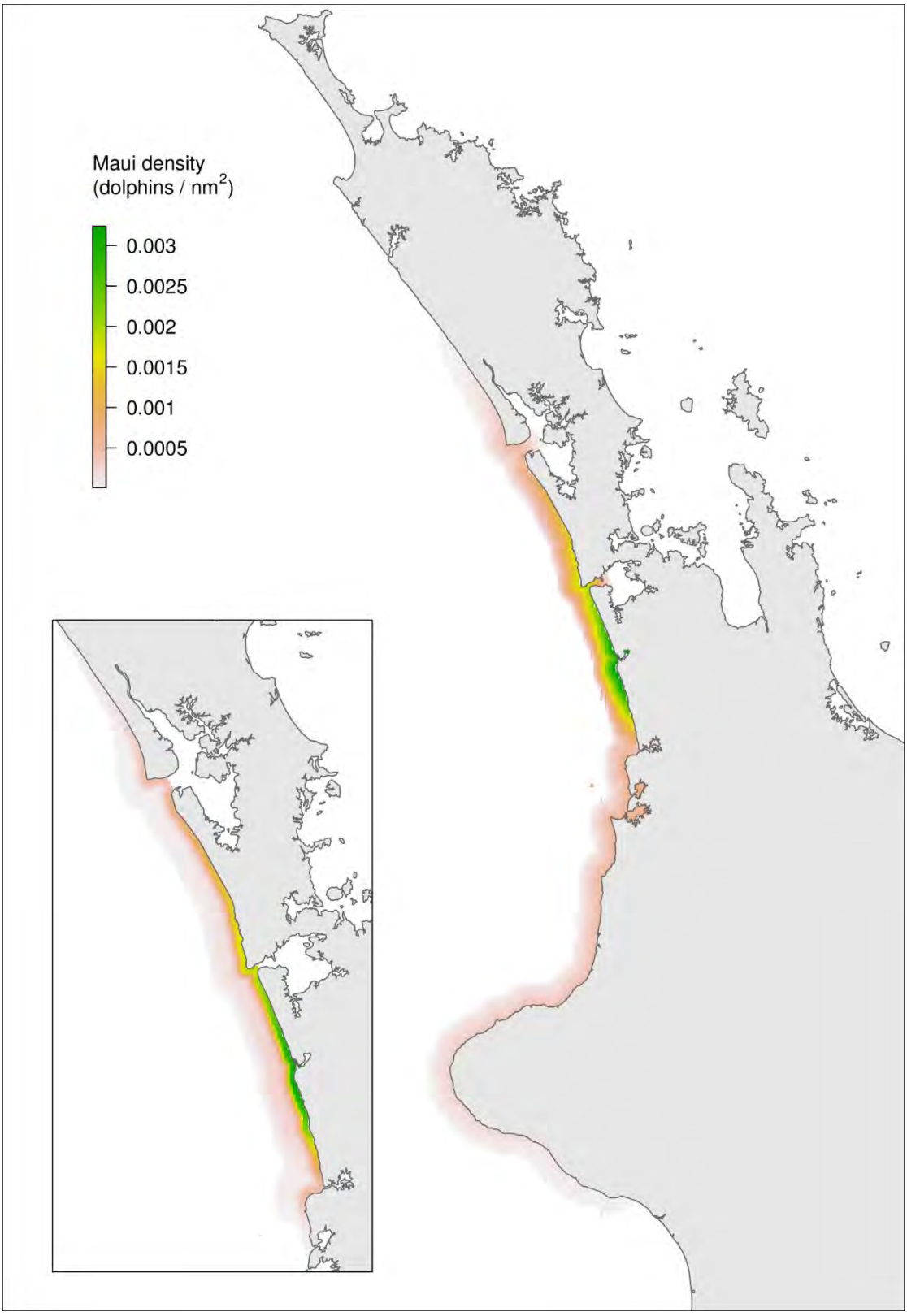


Figure 6.1: Māui dolphin distribution modelled from systematic survey data collected between 2000 and 2012 and modified to incorporate expert panel feedback (Currey et al. 2012). The inset depicts the modelled distribution prior to modification (Thompson & Richard 2012).

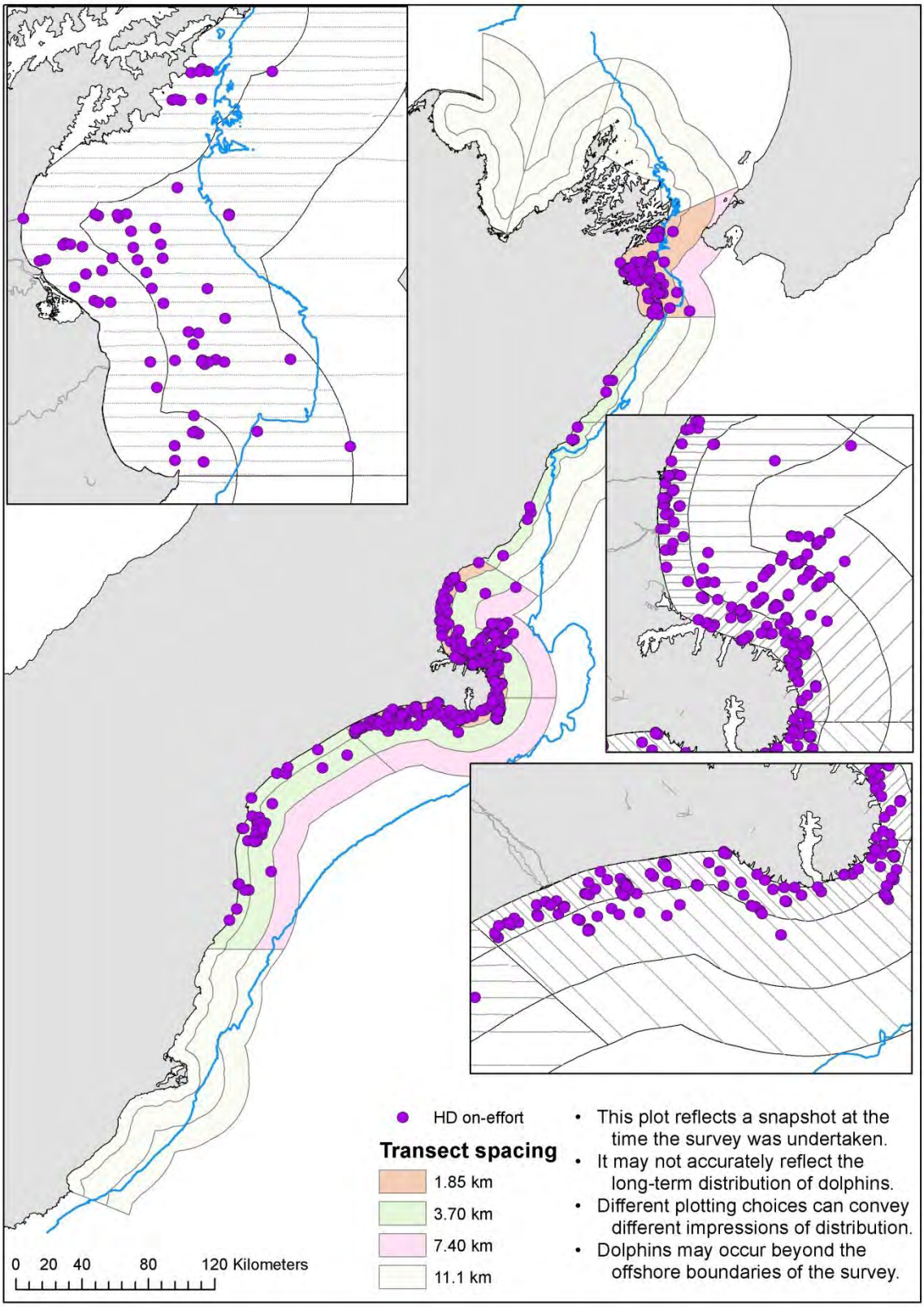


Figure 6.2: The distribution of all on-effort sightings of Hector's dolphins during the summer survey of the ECSI between 28 January and 13 March 2013. Reproduced from MacKenzie & Clement (2014).

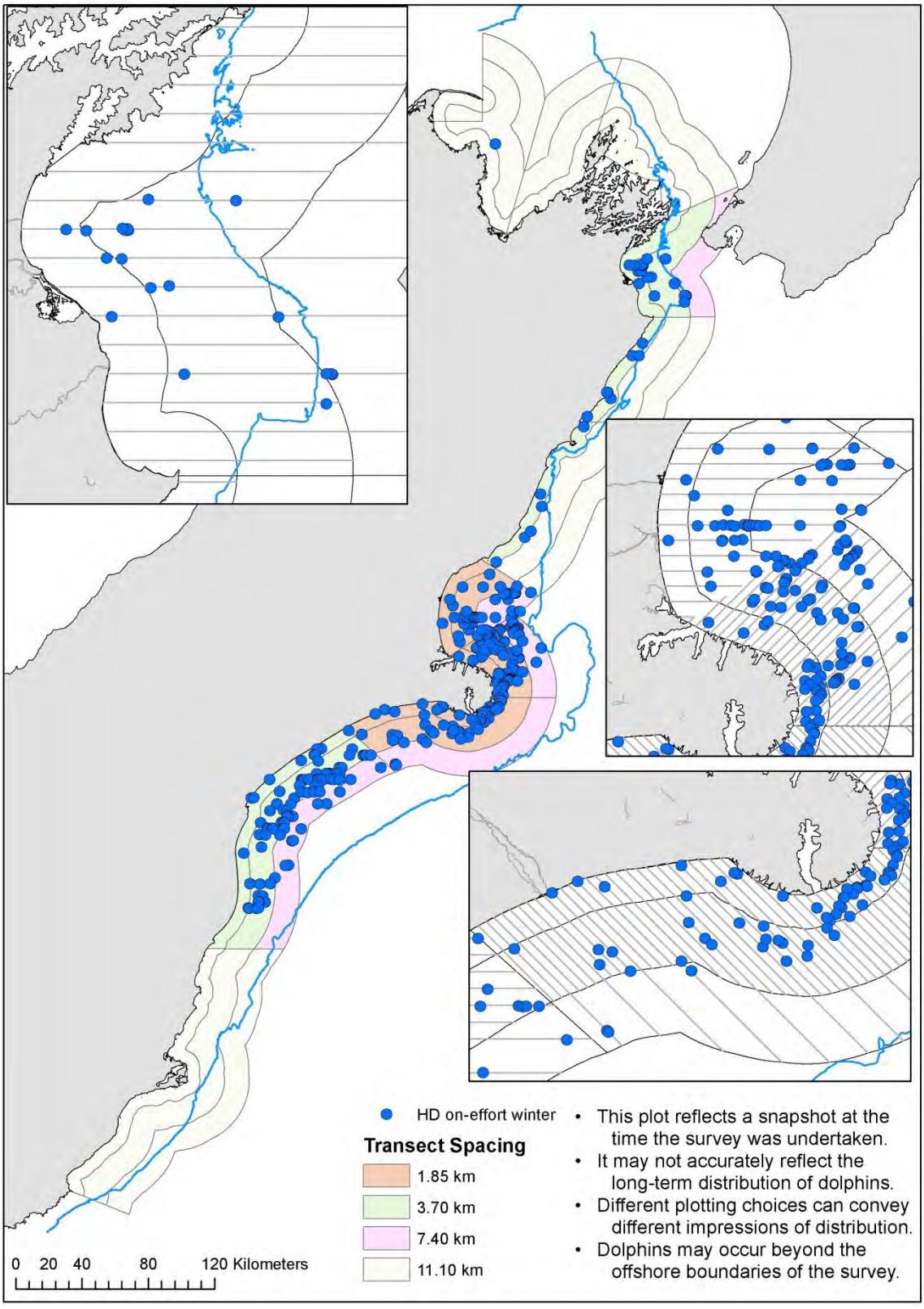


Figure 6.3: The distribution of all on-effort sightings of Hector's dolphins during the winter survey of the ECSI between 1 July and 18 August 2013. Reproduced from MacKenzie & Clement (2014).

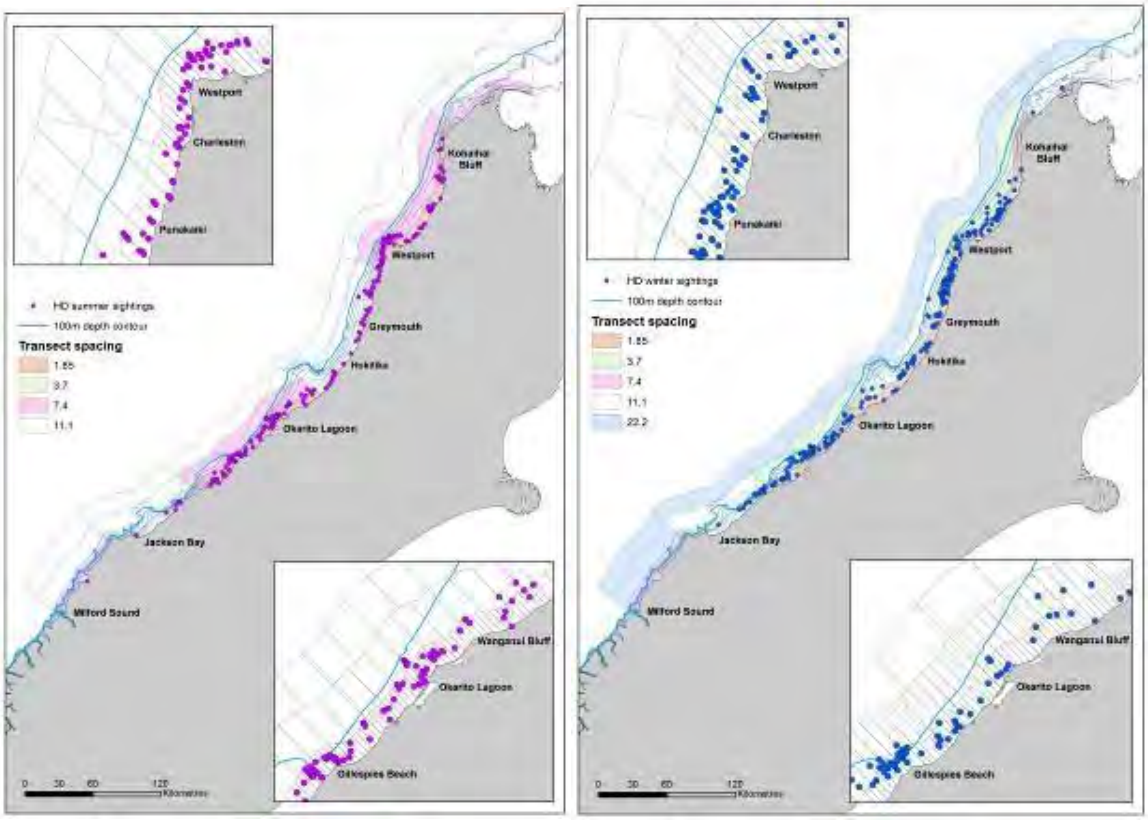


Figure 6.4: Hector's dolphin distributions assessed from aerial line-transect surveys. Panels represent patterns for all on-effort Hector's dolphin sightings in summer (left), and winter (right). Reproduced from MacKenzie & Clement (2016).

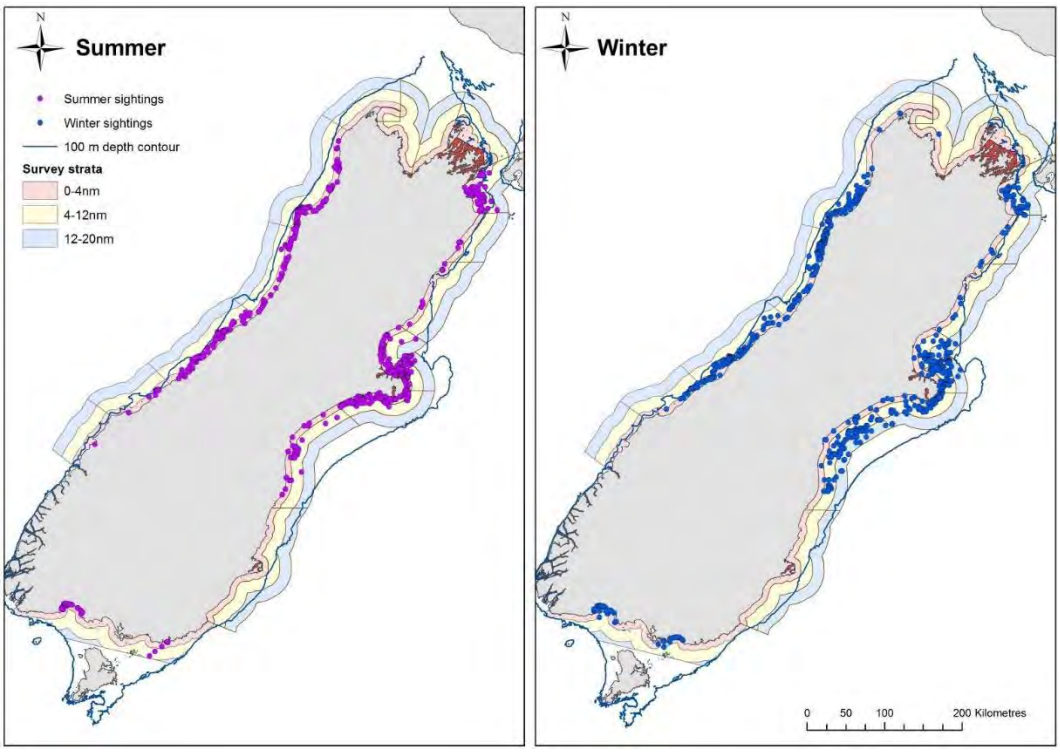


Figure 6.5: Hector's dolphin summer (left) and winter (right) sightings from the three separate abundance surveys: east coast (WCSI) completed 2015, east and north coast (ECSI) completed in 2013 and south coast (SCSI) completed in 2010. SCSI survey will be repeated in early 2018. Reproduced from MacKenzie & Clement (2016).

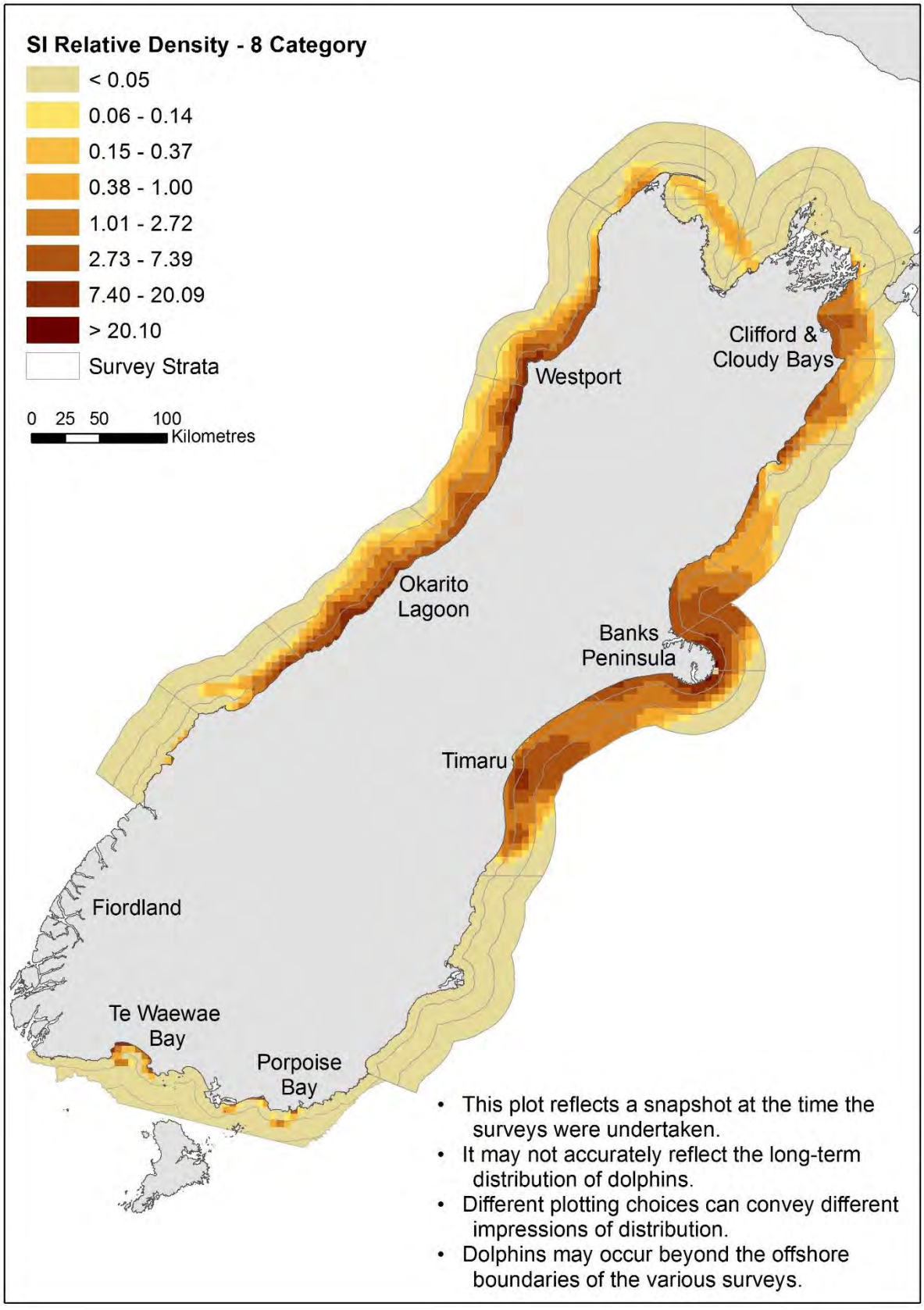


Figure 6.6: The South Island distribution of Hector's dolphin assessed from both summer and winter aerial line-transect surveys. Shows the pattern of the relative density of Hector's dolphins within 5 × 5 km grid cells generated from the Density Surface Models with eight categories. Reproduced from MacKenzie & Clement (2016).

Further aerial surveys focused on assessing seasonal and annual variation in distribution around Banks Peninsula (Rayment et al. 2010) and in distribution and abundance in Cloudy and Clifford Bays (DuFresne & Mattlin 2009). There have also been a number of photo-ID mark-recapture estimates focused on subpopulations of Hector's dolphin (Bejder & Dawson 2001, Gormley et al. 2005, Turek et al. 2013; Table 6.1) and genotype mark-recapture estimates of abundance for Māui dolphins and Hector's dolphins in Cloudy Bay (Hamner et al. 2012b, 2013, Baker et al. 2013, 2016b; Table 6.1). The genetic mark-recapture data yielded estimates of average annual population change for Māui dolphin of -0.13 (i.e., a 13% decrease p.a.; 95% c.i.: -0.40–+0.14) for the period 2001–07 (Baker et al. 2013), and -0.03 (95% c.i.: -0.11–+0.06) for the period 2001–11 (Hamner et al. 2012b). Baker et al. (2016b) estimated an abundance of N = 63 with 95% log-normal CL = 57, 75 for the population of Māui dolphins one year old and older. This estimate is comparable to, but slightly larger than the previous estimate of N = 55 (95% CL = 48, 69) based on the genotype surveys in 2010–11 (Hamner et al. 2012b).

Population trends have also been inferred for Māui dolphins via other methods, including linear regression of the natural logarithm of abundance estimates obtained using a variety of survey methods over the period 1985 to 2011 (-0.032; 90% c.i.: -0.057 to -0.006 for aerial and boat surveys; -0.037; 90% c.i.: -0.042 to -0.032 for boat surveys alone; Wade et al. 2012). Analysis of the Māui dolphin risk assessment expert panel's mortality scores yielded an estimated rate of population decline of 7.6% per annum (95% c.i.: = 13.8% decline to 0.1% increase; Currey et al. 2012). Across methods, estimates of Māui dolphin population trends indicate a high probability that the population is declining, with mean or median estimates suggesting a rate of decline at or above 3% per annum (Currey et al. 2012, Hamner et al. 2012b, Wade et al. 2012, Baker et al. 2013). Based on the more recent genetic mark-recapture data (Baker et al. 2016b), the best-fitting Pradel Survival and Lambda model estimated the annual rate of change to be 0.983 (95% c.i.: 0.940–1.028). Therefore, the Māui dolphin estimates suggest that the population declined by approximately 1.5–2% per year between 2001 and 2016; however, the decline was not confirmed with 95% confidence, as the upper confidence limits span a range up to a population increase of 3% per year (Baker et al. 2016b).

Recently, MPI-funded survey programmes (PRO2009-01A, PRO2009-01B, PRO2009-01C, PRO2013-06) were conducted to assess abundance and distribution of the SCSI, ECSI and WCSI populations of Hector's dolphin (Clement et al. 2011, MacKenzie et al. 2012, MacKenzie & Clement 2014, 2016). The SCSI programme involved two aerial surveys undertaken during March 2010 and August 2010 between Puysegur Point and Nugget Point and out to the 100 m depth contour (PRO2009-01A, Clement et al. 2011). Seven dolphin groups were sighted during summer/autumn surveys and ten groups were observed in winter. Sightings data pooled across seasons were analysed using mark-recapture distance sampling (MRDS) with helicopter-based dive cycle observations used to correct for availability bias. SCSI Hector's dolphin abundance was estimated to be 628 dolphins (CV = 38.9%; 95% c.i.: 301–1311; Clement et al. 2011). MacKenzie & Clement (2016) reanalysed the SCSI survey data from 2014 and produced an annual average estimate for the SCSI of 238 (s.e.: 94; 95% c.i.: 113–503) based on revised figures for availability. Under project PRO2016-09, the SCSI survey will be repeated in early 2018 with higher sampling intensity in the nearshore strata to achieve a lower CV in the estimation of population size.

The ECSI program involved an initial design phase (PRO2009-01B, MacKenzie et al. 2012) followed by two aerial surveys conducted over summer 2012–13 and winter 2013 between Farewell Spit and Nugget Point and offshore to 20 nm (covering about 42 677 km^2; PRO2009-01C; MacKenzie & Clement 2014). A total of 354 dolphin groups were sighted in the summer, along 7156 km of transect lines, and 328 dolphin groups were sighted in the winter, along 7276 km of transect lines (Figure 6.2 and Figure 6.3). Sightings data were analysed using MRDS and density surface modelling techniques to yield estimates of density and total abundance. The estimates of ECSI Hector's dolphin abundance were 9130 dolphins (CV = 19%; 95% c.i.: 6342–13 144) in summer 2012–13 and 7456 dolphins (CV = 18%: 95% c.i.: 5224–10 641) in winter 2013 (MacKenzie & Clement 2014). These estimates were obtained via model averaging four sets of MRDS results for each season; from two different datasets using different truncation distances and two methods of estimating availability (helicopter-based dive cycle and survey aircraft circle-backs). These estimates do not include harbours and bays that were outside of the survey region. MacKenzie & Clement (2016) reanalysed the ECSI survey data from 2014 and produced an annual average estimate for the ECSI of 8968 (s.e.: 1377; 95% c.i.: 6649–12 096), based on revised figures for availability.

A survey programme was specifically designed for sampling the WCSI population using two separate aerial surveys over summer 2014–15 and winter 2015 (MPI project PRO3013-06). The WCSI survey area (about 26 333 km^2 between Farewell Spit and Milford Sound) was stratified into six coastal sections, which were further divided into offshore substrata of 0–4 nm (inner), 4–12 nm (middle) and 12–20 nm (outer). This design was expected to encompass the offshore limits of Hector's dolphin distribution along the WCSI (MacKenzie & Clement 2016).

The WCSI Hector's dolphin summer abundance was estimated to be 5490 (CV: 26%; 95% c.i.: 3319–9079) and 5802 (CV: 21%; 95% c.i.: 3879–8679) in winter. These estimates were obtained by averaging the four sets of results for each season; from two different datasets using different truncation distances and two methods of estimating availability (dive cycle and circle-backs). These estimates are very similar to the previous 2000–01 WCSI estimate of 5388 Hector's dolphins by Slooten et al. (2004) (CV = 21%; 95% c.i.: 3613–8034), even after accounting for differences in offshore survey areas (MacKenzie & Clement 2016).

Summer sightings results consisted of 250 Hector's dolphin groups (115 of which were seen by two observers) sighted within 0.3 km either side of the plane along 4001 km of transect lines. In winter, 272 Hector's dolphin groups (115 of which were seen by two observers) were sighted within 0.3 km either side of the plane along 4307 km of transect lines. Hector's dolphins were observed as far offshore as 12 km (6.5 nm) and 17.7 km (9.5 nm) in summer and winter, and in waters as deep as 160 m and 200 m, respectively. However, the majority of animals in both seasons occurred close to shore (less than 3 nm) and within relatively shallow depths (less than 40 m) (Figure 6.4; MacKenzie & Clement 2016).

Following the reanalysis of the ECSI and SCSI survey data, MacKenzie & Clement (2016) estimate the total Hector's population around the South Island (excluding sounds and harbours) to be 14 849 (CV: 11%; 95% c.i.: 11 923–18 492). This estimate is approximately twice as large as the previous estimate from surveys conducted in the late 1990s – early 2000s (7300; 95% c.i.: 5303–9966) (Slooten et al. 2004), with the difference primarily due to a substantial number of dolphins estimated to be in offshore areas (greater than 4 nm) along ECSI that had not been extensively surveyed previously (Figures 6.4 and 6.5). Densities are similar along ECSI and WCSI (Figure 6.6; MacKenzie & Clement 2016).

For several years questions have been brought forward in International Whaling Commission (IWC) Sub-committee on Small Cetaceans concerning the methods used to derive abundance estimates of Hector's dolphins by New Zealand. The subcommittee agreed at the 2015 IWC meeting to review the abundance estimates intersessionally (International Whaling Commission 2016a). A formal process was established intersessionally following IWC procedures for such review and this included the creation of an Intersessional Expert Group (IEG) and an Intersessional Correspondence Group (ICG). The IEG consisted of independent experts who were asked to review the abundance methodology and estimates produced by MacKenzie and Clement (2014, 2016) (International Whaling Commission 2016b).

The IEG recognised that this study accounted for many difficulties that also affect other small cetacean abundance estimation studies using aerial surveys. It commended the ambitious and often innovative work undertaken by the authors to attempt to deal with all of those issues. After an indepth review of the survey design, analyses and results, the IEG endorsed the abundance estimates and concluded that the estimates accurately reflected the data, were derived from appropriate data collection and analysis methods, and represented the most current abundance estimate for Hector's dolphins around the South Island. Thus, they believed that it follows that it would be reasonable to use them to inform a management plan. The IEG also considered this study to be a step forward in the development of survey methodology more generally (International Whaling Commission 2016b).

Hector's dolphin is one of very few dolphin species for which estimates of survival are available. For long-lived species, a long time series of data is required to robustly estimate survival. The long term photo-ID study at Banks Peninsula has facilitated several survival rate estimates since its inception in 1984 (Slooten et al. 1992, Cameron et al. 1999, Du Fresne 2004, Gormley et al. 2012). The most recent analysis utilises the most data and is therefore arguably the most powerful. Survival rate was estimated as 0.863 (95% c.i.: 0.647–0.971) for the period 1986–88, prior to the designation of the Banks Peninsula Marine Mammal Sanctuary, and 0.917 (95% c.i.: 0.802–0.984) from 1989–2006 after the designation (Gormley et al. 2012). Given the reproductive parameters detailed above, these survival

rate estimates equate to a mean estimated population growth rate of 0.939 (95% c.i.: 0.779–1.025) pre-sanctuary and 0.995 (95% c.i.: 0.927–1.048) post-sanctuary (Gormley et al. 2012). In the post-sanctuary scenario, most of the uncertainty in the population growth estimate is due to uncertainty in the estimate of fecundity (Gormley et al. 2012).

Annual survival of the Māui dolphin has been estimated from the genotype mark-recapture data (Hamner et al. 2012b, Baker et al. 2013, 2016b). The best-fitting Pradel Survival and Lambda model for the data series, 2001–16, estimated the annual survival for age 1+ dolphins to be 0.888 (95% c.i.: 0.842–0.922; Baker et al. 2016b).

Where mark-recapture data are available, population models of Māui and Hector's dolphin subpopulations will be used to inform the update of the Hector's and Māui dolphin TMP in 2018.

6.2.6 KNOWN AND POTENTIAL THREATS

Fishing-related mortality is known to be a potentially serious threat to Hector's and Māui dolphins (DOC & MFish 2007, MPI & DOC 2012). Fisheries risk is described in Section 6.4, below.

Non fishery threats have also been observed but are difficult to quantify. There has been one confirmed death due to boat strike since 1921, a Hector's dolphin calf in Akaroa harbour in 1999 (Stone & Yoshinaga 2000, DOC 2017a). Other known sources of mortality include predation by sharks (e.g., Cawthorn 1988), disease (e.g., Roe et al 2013) and separation of calves from their mothers (DOC 2017a), possibly exacerbated by extreme weather conditions (DOC & MFish 2007, MPI & DOC 2012).

The presence of tourist vessels has been demonstrated to cause behavioural changes (Bejder et al. 1999, Martinez et al. 2012), as has underwater noise. There are potential negative effects due to bioaccumulation of organochlorines and heavy metals (reviewed by Slooten & Dawson 1994). Stockin et al. (2010) reported elevated levels of PCBs and organochlorine pesticides in the tissues of Hector's and Māui dolphins, but noted that no PCB concentrations were over the threshold considered to have immunological and reproductive effects. Additionally, both subspecies face pressures placed on coastal habitat through activities such as aquaculture, seabed mining, dredging and tidal energy installations (DOC & MFish 2007, Currey et al. 2012, MPI & DOC 2012).

A comprehensive list of the threats posed to Māui dolphins was produced as part of the spatially explicit, semi-quantitative risk assessment (Currey et al. 2012). The expert panel was asked to identify, analyse and evaluate all potential threats to Māui dolphins. Working from a previously established list of 47 potential threats to Hector's dolphins from the Hector's and Māui dolphin TMP (DOC & MFish 2007), the expert panel assessed 23 threats potentially relevant to Māui dolphins (i.e., present within their established distribution) in terms of whether these were likely to affect population trends within the next five years (

Table 6.2). For each of these threats, the expert panel provided estimates of the number of Māui dolphin mortalities per year (

Table 6.3).

The panel process resulted in estimated numbers of Māui dolphin mortalities from commercial set net fisheries of 2.33 (95% c.i.: 0.02–4.26) per annum, with spatial disaggregation of the estimates indicating that Māui dolphins are exposed to the greatest level of risk from set net fisheries in the area of the northern Taranaki coastline out to 7 nm offshore, and at the entrance to the Manukau Harbour. Subsequent interim measures banned set-net fishing within 2 nm of the Taranaki coast (between Pariokariwa Point and Hawera) and required full observer coverage of commercial set net fishing out to 7 nm. No Māui dolphins have been captured or sighted by observers in the Taranaki set-net fishery since observer coverage began in July 2012.

The expert panel's assessment of mortalities can be treated as testable hypotheses reflecting the limitations of available knowledge at that time and should be updated using new information. In particular, Roe et al.'s (2013) finding that 2 of 3 Māui dolphins tested in the period 2007 to 2011 had died as a result of *Toxoplasma gondii* infection, possibly as a result of run-off from terrestrial sources, indicates that the panel results (

Table 6.3) may have underestimated mortality from this source. Roe et al. (2013) note that toxoplasmosis may have other effects beyond direct mortality and could be an important cause of neonatal loss. New work to investigate the risk of toxoplasmosis to Hector's and Māui dolphins is ongoing (W. Roe, pers. comm.) and will inform the update of the TMP in 2018.

6.2.7 CONSERVATION BIOLOGY AND THREAT CLASSIFICATION

Threat classification is an established approach for identifying species at risk of extinction (IUCN 2013). The risk of extinction for Hector's and Māui dolphin has been assessed under two threat classification systems: the New Zealand Threat Classification System (Townsend et al. 2008) and the International Union for the Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2013).

The IUCN classifies Māui dolphin as Critically Endangered under criteria A4c,d and C2a(ii)[5] due to an ongoing and projected decline of greater than 80% over three generations, and there being fewer than 250 mature individuals remaining (Reeves et al. 2013a). Critically Endangered is the most threatened status before 'Extinct in the Wild'. Hector's dolphin is classified by the IUCN as Endangered under criterion A4d[6] due to an ongoing and projected decline of greater than 50% over three generations (Reeves et al. 2013b).

Under the New Zealand Threat Classification System (Baker et al. 2016a), Māui dolphin is classified as Nationally Critical, the most threatened status, under criterion A(1), with the qualifier Conservation Dependent (CD)[7] and Hector's dolphin as Nationally Endangered, the second most threatened status, under criterion C(1/1), with the qualifier Conservation Dependent (CD).[8]

Table 6.1: Abundance estimates for Hector's and Māui dolphin. N = estimated population size. * applies to individuals more than 1 year of age and includes two individuals genetically identified as Hector's dolphins. [Continued on next pages]

Sampling period	Subspecies	Survey area	Survey method	Analysis method	N	CV	95% c.i.	Reference
1984–85	Hector's and Māui dolphin	North and South Islands	Small boat based strip-transect	Distance sampling	3 408		3 000–4 000	Dawson & Slooten 1988
1989–97	Hector's dolphin	Banks Peninsula	Photo-ID	Mark-recapture	1 119	0.21	744–1682	Gormley et al. 2005
1995–97	Hector's dolphin	Porpoise Bay	Photo-ID	Mark-recapture	48		44–55	Bejder & Dawson 2001

[5] A taxon is listed as 'Critically Endangered' if it is considered to be facing an extremely high risk of extinction in the wild. A4c,d refers to a reduction in population size (A), based on an observed, estimated, inferred, projected or suspected reduction of ≥ 80% over any 10-year or three-generation period (whichever is longer up to a maximum of 100 years (3); with the reduction being based on a decline in area of occupancy, extent of occurrence and/or quality of habitat (c); or actual or potential levels of exploitation (d; IUCN 2010). C2a(ii) refers to a population size estimated to number fewer than 250 mature individuals (C); with a continuing decline, observed, projected, or inferred, in numbers of mature individuals (2); and a population structure (a) with at least 90% of mature individuals in one subpopulation (ii; IUCN 2013).

[6] A taxon is listed as 'Endangered' if it is considered to be facing a very high risk of extinction in the wild. A4d refers to a reduction in population size (A), based on an observed, estimated, inferred, projected or suspected reduction of ≥ 80% over any 10-year or three-generation period (whichever is longer up to a maximum of 100 years (3); with the reduction being based on actual or potential levels of exploitation (d, IUCN 2013).

[7] A taxon is listed as 'Nationally Critical' under criterion A(1) when evidence indicates that there are fewer than 250 mature individuals, regardless of population trend and regardless of whether the population size is natural or unnatural (Townsend et al. 2008).

[8] A taxon is 'Nationally Endangered' under criterion C(1/1)when evidence indicates that the total population size is 1000–5000 mature individuals and there is an ongoing or predicted decline of 50–70% in the total population due to existing threats, taken over the next 10 years or three generations, whichever is longer (Townsend et al. 2008).

Table 6.1 [Continued]:

Sampling period	Subspecies	Survey area	Survey method	Analysis method	N	CV	95% c.i.	Reference
1997–98	Hector's dolphin	Motunau–Timaru (0–4 nm)	Boat based line-transect	Distance sampling	1 198	0.27	848–1 693	Dawson et al. 2004
1998–99	Hector's dolphin	Timaru–Long Point (0–4 nm)	Boat based line-transect	Distance sampling	399	0.26	279–570	Dawson et al. 2004
1999–2000	Hector's dolphin	Farewell Spit–Motunau (0–4 nm)	Boat based line-transect	Distance sampling	285	0.39	137–590	Dawson et al. 2004
2000–01	Hector's dolphin	Farewell Spit–Milford Sound (0–4 nm)	Aerial line-transect	Distance sampling	5 388	0.21	3 613–8 034	Slooten et al. 2004
2001–07	Māui dolphin	Kaipara Harbour–Tirua Point	Biopsy	Mark-recapture	59		19–181	Baker et al. 2013
2004	Māui dolphin	Maunganui Bluff–Pariokariwa Point (0–4 nm)	Aerial line-transect	Distance sampling	111	0.44	48–252	Slooten et al. 2006
2004–05	Hector's dolphin	Te Waewae Bay	Photo-ID	Mark-recapture	251 (autumn)	0.16	183–343	Green et al. 2007
					403 (summer)	0.12	280–488	
2006–09	Hector's dolphin	Cloudy and Clifford Bays (100 m contour)	Aerial line-transect	Distance sampling	951 (summer)	0.26	573–1 577	DuFresne & Mattlin 2009
					927 (autumn)	0.30	520–1 651	
					315 (winter)	0.31	173–575	
					188 (spring)	0.33	100–355	
2010	Hector's dolphin	Puysegur Point–Nugget Point (100 m contour)	Aerial line-transect	Distance sampling	628	0.39	301–1 311	Clement et al. 2011
2010 (reanalysis)	Hector's dolphin	Puysegur Point–Nugget Point (100 m contour)	Aerial line-transect	Distance sampling	238		113-503	MacKenzie & Clement 2016
2010–11	Māui dolphin	Kaipara Harbour–New Plymouth	Biopsy	Mark-recapture	57*		49–71	Hamner et al. 2012b
2010–11	Hector's dolphin	Taiaroa Head–Cornish Head	Photo-ID	Mark-recapture	42	0.41	19–92	Turek et al. 2013
2011–12	Hector's dolphin	Cloudy Bay	Biopsy	Mark-recapture	272	0.12	236–323	Hamner et al. 2013

Table 6.1 [Continued]:

Sampling period	Subspecies	Survey area	Survey method	Analysis method	N	CV	95% c.i.	Reference
2012–13	Hector's dolphin	Farewell Spit–Nugget Point (0–20 nm)	Aerial line-transect	Mark-recapture distance sampling	9 130 (summer)	0.19	6 342–13 144	MacKenzie & Clement 2014
					7 456 (winter)	0.18	5 224–10 641	
2012–13 (reanalysis)	Hector's dolphin	Farewell Spit–Nugget Point (0–20 nm)	Aerial line-transect	Mark-recapture distance sampling	9 728 (summer)	0.17	7 001–13 517	MacKenzie & Clement 2016
					8 208 (winter)	0.27	4 888–13 785	
2014–15	Hector's dolphin	Farewell Spit–Nugget Point (0–20 nm)	Aerial line-transect	Mark-recapture distance sampling	5 490 (summer)	0.26	3 319–9 079	MacKenzie & Clement 2016
					5 802 (winter)	0.21	3 879–8 679	
2015–16	Māui dolphin	Kaipara Harbour–New Plymouth	Biopsy	Mark-recapture	63		57–75	Baker et al. 2016b

6.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Coastal cetaceans are impacted by incidental capture in fisheries throughout the world (Read et al. 2006, Read 2008, Reeves et al. 2003). Read et al. (2006) estimated that global incidental captures of cetaceans exceeded 270 000 p.a. in the mid-1990s and that more than 95% of incidental captures occurred in set nets. Hector's and Māui dolphins are endemic to New Zealand and hence discussion of fisheries interactions for the species is detailed below under state of knowledge in New Zealand.

6.4 STATE OF KNOWLEDGE IN NEW ZEALAND

It is widely accepted that incidental mortality in coastal fisheries, notably set nets and to a lesser extent trawls, is the most significant threat to Hector's and Māui dolphins (MFish & DOC 2007, Slooten & Dawson 2010, Currey et al. 2012; see

Table 6.3). Hector's and Māui dolphins have been caught in inshore commercial and recreational set net fisheries since at least the early 1970s (Taylor 1992). Incidental mortalities have been documented throughout the species' range (Table 6.4). Beach-cast carcasses are frequently reported by members of the public, with the greatest number of reports coming from the east coast of the South Island (DOC 2017a). The numbers reported in the DOC Incident Database are not representative of the total magnitude or relative scale of incidental capture (DOC 2017a, Slooten 2013) because carcasses may not be reported by fishers, may not wash ashore, may not be recovered or may not show evidence of interaction with fishing gear. Carcass reporting is also likely to be correlated with proximity to major population centres and thoroughfares. The information in the incident database (Table 6.4) provides only a biased indication of incidental captures. It is clear from this information, however, that incidental captures occur in all areas where the distribution of Hector's and Māui dolphins overlaps with the distribution of fishing effort. Where overlap occurs, the rate at which dolphins are captured per unit of overlap (as a proxy for encounter rate) can be estimated using fisheries observer programmes, and potentially video monitoring (see below).

Incidental captures have most frequently occurred in commercial set nets targeting rig (*Mustelus lenticulatus*), elephant fish (Callorhynchus milli) and school shark (*Galeorhinus australis*, Dawson 1991, Baird & Bradford 2000), and in recreational nets set for flounder (*Rhomboselea* sp.) and moki (*Latridopsis ciliaris*, Dawson 1991).

Nineteen individual Hector's dolphins were reported caught in trawl fisheries between 1921 and 2008 and one since 2008 (Table 6.4; DOC 2017a). The first report of incidental capture in the commercial trawl fishery dates back to 1973 (Baker 1978). Note however that in the application of fisheries risk assessment methods used by MPI, only fishing effort that has been independently observed is used to estimate capture rates per encounter with fishing.

There have been three known incidents of Hector's dolphins becoming entangled in buoy lines of pots set for crayfish (*Jasus edwardsii*), all from Kaikōura (DOC & MFish 2007, DOC 2017a). From July 2008 to November 2017, there have been seven additional incidents of known entanglements in commercial set nets (six from the ECSI, one from WCNI) and five incidents of probable entanglements (three from ECSI, one from WCSI and one from the north coast of the South Island). These additional data are valid as of December 2017 (Table 6.4; DOC 2017a).

There are discrepancies between the data presented in the DOC Incident Database (2017a) and elsewhere in the published literature. Dawson (1991) collated reports of known incidental captures in Canterbury between 1984 and 1988 based on interviews with fishers. The minimum estimate of incidental captures in commercial set nets was 200 and in amateur nets was 24 (Dawson 1991), both of which are appreciably higher than the numbers presented in Table 6.4. These interview estimates were reviewed by Voller (1992) who reported a total of 112 entanglements in commercial nets from Timaru to Motanau in the period 1984–88 and attributed the difference from Dawson's results to the assumptions made about information provided by three individuals. There are a number of reasons why the people who were interviewed multiple times may have provided different information regarding incidental captures.

Table 6.2: Characterisation of threats evaluated as relevant to Māui dolphins and likely to affect population trends within the next five years. Reproduced from Currey et al. (2012). [Continued on next page]

Threat class	Threat	Mechanism	Type	Population component(s) affected
Fishing	Commercial trawl	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Commercial set net	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Recreational set net	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Recreational driftnet	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Customary set net	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Trophic effects	Competition for prey, changes in abundance of prey and predator species	Indirect	Fecundity, juvenile or adult survival
	Vessel noise: displacement, sonar	Displacement from habitat, masking biologically important behaviour	Indirect	Fecundity, juvenile or adult survival
Vessel traffic	Boat strike	Physical injury/mortality	Direct	Juvenile or adult survival
	Disturbance	Displacement from habitat, masking biologically important behaviour	Indirect	Fecundity, juvenile or adult survival
Pollution	Agricultural run-off	Compromising dolphin health, habitat degradation, trophic effects	Indirect	Fecundity, juvenile or adult survival
	Industrial run-off	Compromising dolphin health, habitat degradation, trophic effects	Indirect	Fecundity, juvenile or adult survival
	Plastics	Compromising dolphin health, ingestion and entanglement	Both	Fecundity, juvenile or adult survival
	Oil spills	Compromising dolphin health, ingestion (direct and prey) and inhalation	Both	Fecundity, juvenile or adult survival
	Trophic effects	Changes in abundance of prey and predator species	Indirect	Fecundity, juvenile or adult survival
	Sewage and stormwater	Compromising dolphin health, habitat degradation, trophic effects	Indirect	Fecundity, juvenile or adult survival
Disease	Natural	Compromising dolphin health	Both	Fecundity, juvenile or adult survival
	Stress-induced	Compromising dolphin health	Both	Fecundity, juvenile or adult survival
	Domestic animal vectors	Compromising dolphin health	Both	Fecundity, juvenile or adult survival

Table 6.2 [Continued]:

Threat class	Threat	Mechanism	Type	Population component(s) affected
Small population effects	Stochastic and Allee effects	Increased susceptibility to other threats	Indirect	Fecundity, juvenile or adult survival
Mining and oil activities	Noise (non-trauma)	Displacement from habitat, masking biologically important behaviour	Indirect	Fecundity, juvenile or adult survival
	Noise (trauma)	Compromising dolphin health	Direct	Fecundity, juvenile or adult survival
	Pollution (discharge)	Compromising dolphin health	Indirect	Fecundity, juvenile or adult survival
	Habitat degradation	Displacement from habitat, reduced foraging efficiency, trophic effects	Indirect	Fecundity, juvenile or adult survival

Table 6.3: Estimated number of Māui dolphin mortalities per year, the risk ratio of estimated mortalities to PBR and the likelihood of exceeding PBR for each threat, as scored by the expert panel. Individual threat scores were bootstrap resampled from distributions specified by the panel members and aggregated to generate medians and 95% confidence intervals. Modified from Currey et al. (2012).

Threat	Estimated mortalities		Risk ratio		Likelihood of exceeding PBR
	Median	95% c.i.	Median	95% c.i.	Median percentage
Fishing*	4.97	0.28–8.04	71.5	3.7–143.6	100.0
Commercial set net fishing*	2.33	0.02–4.26	33.8	0.3–74.3	88.9
Commercial trawl fishing*	1.13	0.01–2.87	16.7	0.1–48.5	88.9
Recreational/customary set net fishing	0.88	0.02–3.14	12.8	0.3–50.9	88.7
Recreational driftnet fishing	0.05	0.01–0.71	0.7	0.1–10.9	41.3
Trophic effects of fishing	0.01	<0.01–0.08	0.1	<0.1–1.2	4.7
Vessel noise/disturbance from fishing	<0.01	<0.01–0.10	<0.1	<0.1–1.6	9.0
Mining and oil activities	0.10	0.01–0.46	1.5	0.1–7.4	61.3
Habitat degradation from mining and oil activities	0.03	<0.01–0.17	0.4	<0.1–2.7	26.4
Noise (non-trauma) from mining and oil activities	0.03	<0.01–0.23	0.5	<0.1–3.6	28.6
Noise (trauma) from mining and oil activities	0.01	<0.01–0.13	0.2	<0.1–2.0	8.8
Pollution (discharge) from mining and oil activities	<0.01	<0.01–0.13	0.1	<0.1–2.2	13.4
Vessel traffic	0.07	<0.01–0.19	1.0	0.1–3.1	47.8
Boat strike from all vessels	0.03	<0.01–0.10	0.5	<0.1–1.6	17.9
Vessel noise/disturbance from other vessels	0.02	<0.01–0.12	0.3	<0.1–1.9	14.4
Pollution	0.05	<0.01–0.36	0.8	<0.1–5.9	40.2
Oil spills	0.02	<0.01–0.15	0.4	<0.1–2.4	20.4
Agricultural run-off	<0.01	<0.01–0.12	<0.1	<0.1–1.9	9.6
Industrial run-off	<0.01	<0.01–0.11	<0.1	<0.1–1.7	7.6
Sewage and stormwater	<0.01	<0.01–0.11	<0.1	<0.1–1.6	7.3
Trophic effects of pollution	<0.01	<0.01–0.06	<0.1	<0.1–0.9	2.1
Plastics	<0.01	<0.01–0.01	<0.1	<0.1–0.1	<0.1
Disease	<0.01	<0.01–0.36	<0.1	<0.1–5.5	29.5
Stress-induced diseases	<0.01	<0.01–0.35	<0.1	<0.1–5.2	20.7
Domestic animal diseases	<0.01	<0.01–0.07	<0.1	<0.1–1.1	3.9
Total	**5.27**	**0.97–8.39**	**75.5**	**12.4–150.7**	**100.0**

*Note that since the completion of the Marine Mammal Risk Assessment (Abraham et al. 2017 and Section 6.4.4, below) subjective estimates of commercial fisheries risk from Currey et al. (2012) are no longer considered best available information, but are retained for completeness until a comprehensive multi-threat assessment is available. Information in this table will be replaced following the update of the TMP in 2018.

Table 6.4: Fishing-related cause of death of Hector's and Māui dolphins 1921–2008 and 2008–16 by region as listed in the DOC Incident Database (2017a). ECSI = East Coast South Island, WCSI = West Coast South Island, SCSI = South Coast South Island, WCNI = West Coast North Island. See footnotes for explanation of probability categories as detailed in the database.

	Cause of death	ECSI	WCSI	SCSI	WCNI	Unknown population
From 1921 to 2008						
Known entanglement[9]	Commercial set net	41	2	0	0	2
	Recreational set net	12	9	0	0	0
	Unknown set net	15	6	0	2	1
	Trawl net	15	4	0	0	0
Probable entanglement[10]	Commercial set net	0	0	0	0	0
	Recreational set net	0	0	0	0	0
	Unknown set net	1	4	0	0	0
	Unknown net	8	4	1	1	0
Possible entanglement[11]	Commercial set net	0	0	0	0	0
	Recreational set net	1	0	0	0	0
	Unknown set net	16	10	0	0	0
	Unknown net	16	7	1	2	0
From July 2008 to December 2017						
Known entanglement[12]	Commercial set net	5	0	0	1	0
	Recreational set net	1	1	0	0	0
Probable entanglement[13]	Recreational set net	3	0	0	0	1
	Unknown set net	0	1	0	0	0
Possible entanglement[14]	Commercial set net	1	0	0	0	0

[9] Animal was known (from incident report) to have been entangled and died.
[10] As read from pathology report, or presence of net marks on body and a mention of this in incident report.
[11] As read from pathology report, or presence of net marks on body and a mention of this in incident report.
[12] Animal was known (from incident report) to have been entangled and died.
[13] As read from pathology report, or presence of net marks on body and a mention of this in incident report.
[14] As read from pathology report, or presence of net marks on body and a mention of this in incident report.

Table 6.5: Summary of observed inshore set-net and trawl events, and Hector's and Māui dolphin captures, 1997–2012 (see also Baird & Bradford 2000, Blezard 2002, Fairfax 2002, Rowe 2009, 2010, Ramm 2010, 2012a, 2012b). Observed fishing effort, measured in kilometres of net set, or number of trawl tows. Fishing effort numbers are taken from linked fisher reports where possible. The inshore trawl effort is defined as being vessels less than 28 m, targeting flat fish (FLA, LSO, ESO, SFL, YBF, FLO, GFL, TUR, BFL, PAD) or inshore species (TAR, SNA, GUR, RCO, TRE, JDO, STA, ELE, LEA, QSC, MOK, SCH, SPO, BCO, RSK, HPB, LDO). FMAs include areas within and outside Hector's and Māui dolphin distribution (within: 3, 5, 7, 8 and 9; outside: 1, 2 and 10).

Fishing year	Set net					Inshore trawl			
	Areas (FMAs)	Total effort (sets)	Total effort (kms)	Observed effort (%)	Observed captures	Areas (FMAs)	Effort (tows)	Observed effort (%)	Observed captures
1997–98	3	214	260	0.87	8	3, 5, 7, 10	403	0.5	1
1998–99						2	15	0.02	0
1999–00						2, 3, 9	24	0.04	0
2000–01	3	535	24	0.08	0	2, 3	47	0.08	0
2001–02						1, 3, 9	25	0.04	0
2002–03						1	1	0	0
2003–04						3	4	0.01	0
2004–05						3	2	0	0
2005–06	3, 5, 7, 8	458	139	0.57	0	2, 7, 9	49	0.08	0
2006–07	3, 5, 7, 8	413	167	0.69	1	1, 3, 5, 7, 8, 9	260	0.46	0
2007–08	3, 5, 7, 8, 9	821	295	1.4	1	1, 3, 7, 8, 9	102	0.22	0
2008–09	3, 5, 7, 9	1 829	504	2.41	1	1, 3, 5, 7, 8, 9	1 682	3.46	0
2009–10	1, 3, 5, 7	1 927	580	2.61	2	1, 3, 5, 7	788	1.47	0
2010–11	2, 3	514	174	0.81	0	1, 2, 5, 7, 8	744	1.52	0
2011–12	7, 8, 9	161	75	0.37	0	1, 3, 7	328	0.67	0

6.4.1 OBSERVED FISHERIES INTERACTIONS

Prior to 2012, the only observer programme with sufficient coverage to yield a robust estimate of the rate of incidental capture of Hector's dolphins in inshore commercial set nets (Baird & Bradford 2000) was an observer programme in Statistical Areas 018, 020 and 022 (FMA 3) on the east coast of the South Island in the 1997–98 fishing year, which observed 214 inshore set net events, targeting shark species and elephant fish. Eight Hector's dolphins were caught in five sets, of which two were released alive. Capture rates were most precise in Area 022, where six of the catches were reported, following observer coverage of 39% (Baird & Bradford 2000). Capture rate was estimated at 0.064 dolphins per set (CV = 43%) in Area 022 and 0.037 dolphins per set (CV = 39%) in Areas 020 and 022 combined (Baird & Bradford 2000). A total of 16 dolphins (CV = 43%) were estimated caught in Area 022 with 18 dolphins (CV = 38%)[15] estimated caught in Areas 020 and 022 combined (Baird & Bradford 2000). The authors stress that the preceding estimates are of dolphins caught, and not necessarily of mortalities (Baird & Bradford 2000). Note also that these estimates are from Statistical Areas containing the Banks Peninsula Marine Mammal Sanctuary, which at that time effectively prohibited commercial set netting between Sumner Head and the Rakaia River out to 4 nm from the coast (Dawson & Slooten 1993).

The spatial distribution of inshore set-net and trawl fishery effort is presented in Figure 6.7. The level of observation of inshore set-net fisheries since 1998 has been low (Table 5.5). Slooten & Davies (2012) used the observed set-net data from 2009–10 to estimate total captures on the ECSI of 23 dolphins (CV = 0.21). This was the first published capture estimate since extensive protection measures to mitigate Hector's dolphin risk were introduced in 2008 (see below). While this analysis has not been reviewed by the MPI Aquatic Environment Working Group (AEWG), a similar analysis extrapolating a capture rate estimated around Kaikōura across the ECSI was previously presented to an

[15] This was reported as either 16 or 18 dolphins in the cited reference, but has been confirmed as 18 dolphins by correspondence with the author (S. Baird, pers. comm.).

AEWG and rejected given the unrepresentative nature of the observer coverage.

In the 2012–13 year, the inshore set-net fishery operating in Statistical Areas 022 and 024 was observed by human observers and electronic monitoring. During that time, at least two Hector's dolphins were captured, with one released alive.

Hector's dolphin captures in trawl nets include an individual caught in a trawl targeting red cod (*Pseudophycis bacchus*) in area 022 in 1997–98 (Starr & Langley 2000) and the capture of three Hector's dolphins in a trawl in Cloudy Bay in 2006 (DOC & MFish 2007). Baird & Bradford (2000) noted that the lack of information on the depth and position of commercial trawl effort and low observer coverage precluded any estimation of the total number of Hector's dolphins caught in trawl nets. While there have been ongoing attempts to increase the level of observer coverage in inshore trawl fisheries, it remains low (Table 6.5). A simple extrapolation using capture rate and total fishing effort suggests that the number of dolphins caught in trawl fisheries could be as high as the number caught in set nets (Slooten & Davies 2012).

In addition to data gathered by human observers, electronic monitoring of inshore set net and trawl fisheries has been trialed (McElderry et al. 2007). The trial monitored 89 set-net events and 24 trawls off the Canterbury coast in the 2003–04 fishing year. Two Hector's dolphin captures were recorded in the set nets (McElderry et al. 2007), reflecting a similar catch rate to previous estimates. Observers and electronic monitoring were deployed in the Timaru set-net fishery in 2012–13 and observers were deployed again in 2013–14. One confirmed and one probable capture of Hector's dolphins were observed (Archipelago 2013, MPI 2016).

Until recently, no attempt to quantify total captures of Māui dolphins in set nets or trawls using population-specific observer data was possible, because historical observer coverage was low and at low population sizes even very rare events may constitute considerable risk. In the absence of an empirical estimate, the government relied instead on a semi-quantitative risk assessment relying on estimates provided by a panel of nine domestic and international experts (Currey et al. 2012). The panel estimates attributed 95.5% of the mortality risk to fishing-related activities and 4.5% to non-fishing related threats, with captures in commercial set nets assessed as posing the greatest risk (

Table 6.3; Currey et al. 2012). This risk assessment was conducted before the introduction of interim measures off the west coast of the North Island in 2012, so reflects historical rather than current risk. Since the introduction of interim measures, commercial set-net vessels have been required to carry an MPI observer when operating off the Taranaki coastline from 2 to 7 nm offshore between Pariokariwa Point and Hawera (i.e., outside the existing set net closure area). There have been no observed captures and no observations of dolphins in this area since July 2012 when this observer coverage began.

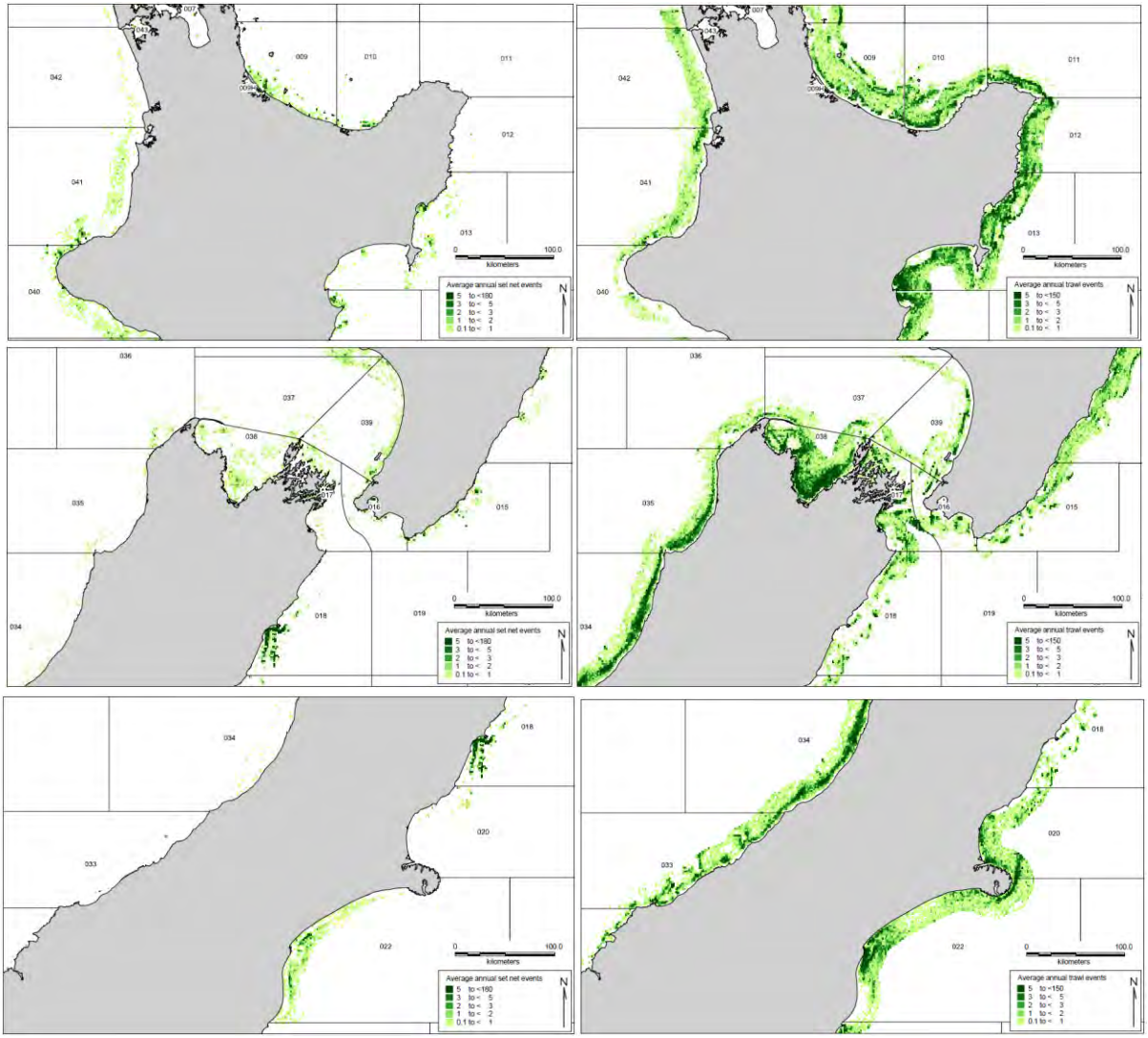


Figure 6.7: The distribution of set-net (left) and trawl (right) fishing events 2007–08 to 2009–10 to show the general spatial pattern of fishing activity. The annual average number of events (start positions) is shown for each 1 nm grid cell for events reporting coordinates (about 33% of set-net events, almost 100% of trawl events). Black lines show general statistical areas. Fishing returns are subject to occasional errors in method codes and coordinates; where possible, these errors have been corrected.

6.4.2 MANAGEMENT OF FISHERIES INTERACTIONS

Broadly, there are three potential solutions to managing incidental captures: gear modifications, mortality limits and spatial closures (Dawson & Slooten 2005). Gear modifications aimed at reducing cetacean captures include changing the way that fishing gear is deployed to reduce the risk of entanglement (e.g., Hembree & Harwood 1987) or adding acoustic alarms (pingers) to make its presence more obvious (Dawson et al. 2013b). Setting mortality limits involves determining a level of mortality that is sustainable (e.g., Wade 1998), and closing the fishery when it is reached. Both these approaches have been used as Hector's dolphin management tools. Most ECSI set-net fishermen voluntarily use pingers under a Code of Practice (Southeast Finfish Management Company 2000), and for a period had an annual mortality limit of three Hector's dolphins for the Canterbury gillnet fishery, which is no longer in effect (Hodgson 2002). Although the effectiveness of pingers has been demonstrated in some experimental trials for other small cetaceans (e.g., Kraus et al. 1997, Trippel et al. 1999, Bordino et al. 2002; see review in Dawson et al. 2013b), cetaceans can become habituated to the presence of pingers (Cox et al. 2001) and fishers do not necessarily deploy them correctly in real fisheries (Cox et al. 2007, Dawson et al. 2013b). Further, a trial reporting that 10 kHz pingers were avoided by Hector's dolphins (Stone et al. 1997) was analytically flawed and hence its conclusion is not correct (Dawson & Lusseau 2005). While setting mortality limits is an effective solution in some fisheries, it requires sufficient observer coverage to provide credible data on how many dolphins are caught, and hence when the fishery should be closed. Baird & Bradford (2000), who analysed the data from the Canterbury observer programme, estimated that the level of observer coverage would need to be 56–83% (depending on the fisheries area) to achieve a CV of 30% on the capture estimate, and 74–100% to achieve a CV of less than 20%. The third solution, creation of spatial closures where harmful activities are restricted or regulated, is the only management approach for which there has been an apparent associated improvement in a vital rate for Hector's and Māui dolphins. Gormley et al. (2012) estimated a 90% probability of increased annual survival rate following the designation of the Banks Peninsula Marine Mammal Sanctuary (see below).

The first spatial closure implemented to mitigate the risk of Hector's dolphin incidental capture was designated at Banks Peninsula in 1988 (Dawson & Slooten 1993). Commercial set netting was effectively prohibited out to 4 nm from the coast and recreational set netting was subject to seasonal restrictions (Dawson & Slooten 1993). A second was designated off the WCNI in 2003. All set nets were prohibited to 4 nm offshore (DOC & MFish 2007). In 2008, a more extensive package of spatial closures was implemented by the Minister of Fisheries (see review by Slooten 2013), providing some protection in most of the areas where Hector's and Māui are found and largely superseding the two existing discrete closures. The set-net restrictions on the WCNI were extended to 7 nm offshore between Maunganui Bluff and Pariokariwa Point (including the entrances to the Kaipara, Manukau and Raglan Harbours and the entrance to the Waikato River) (Figure 6.8), most set netting was prohibited within 4 nm of the coast on the ECSI and SCSI (Figure 6.9 and Figure 6.10), and recreational set netting was banned on the WCSI within 2 nm of the coast and commercial set netting was subject to a seasonal restriction (Figure 6.8). Trawling was banned on the WCNI to 2 nm offshore between Maunganui Bluff and Pariokariwa Point and 4 nm offshore between Manukau Harbour and Port Waikato, and restricted within 2 nm offshore on the ECSI and SCSI[16] (Figure 6.11). In 2012, the set-net restrictions on the WCNI were extended further south, banning commercial and recreational set netting to 2 nm offshore from Pariokariwa Point to Hawera and requiring an MPI observer on any commercial set net vessel operating between 2 and 7 nm. In 2013, the set-net restrictions were extended again, banning commercial and recreational set netting between 2 and 7 nm from Pariokariwa Point to the Waiwhakaiho River mouth (Figure 6.8).

[16] Detailed descriptions of the restrictions can be found at: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

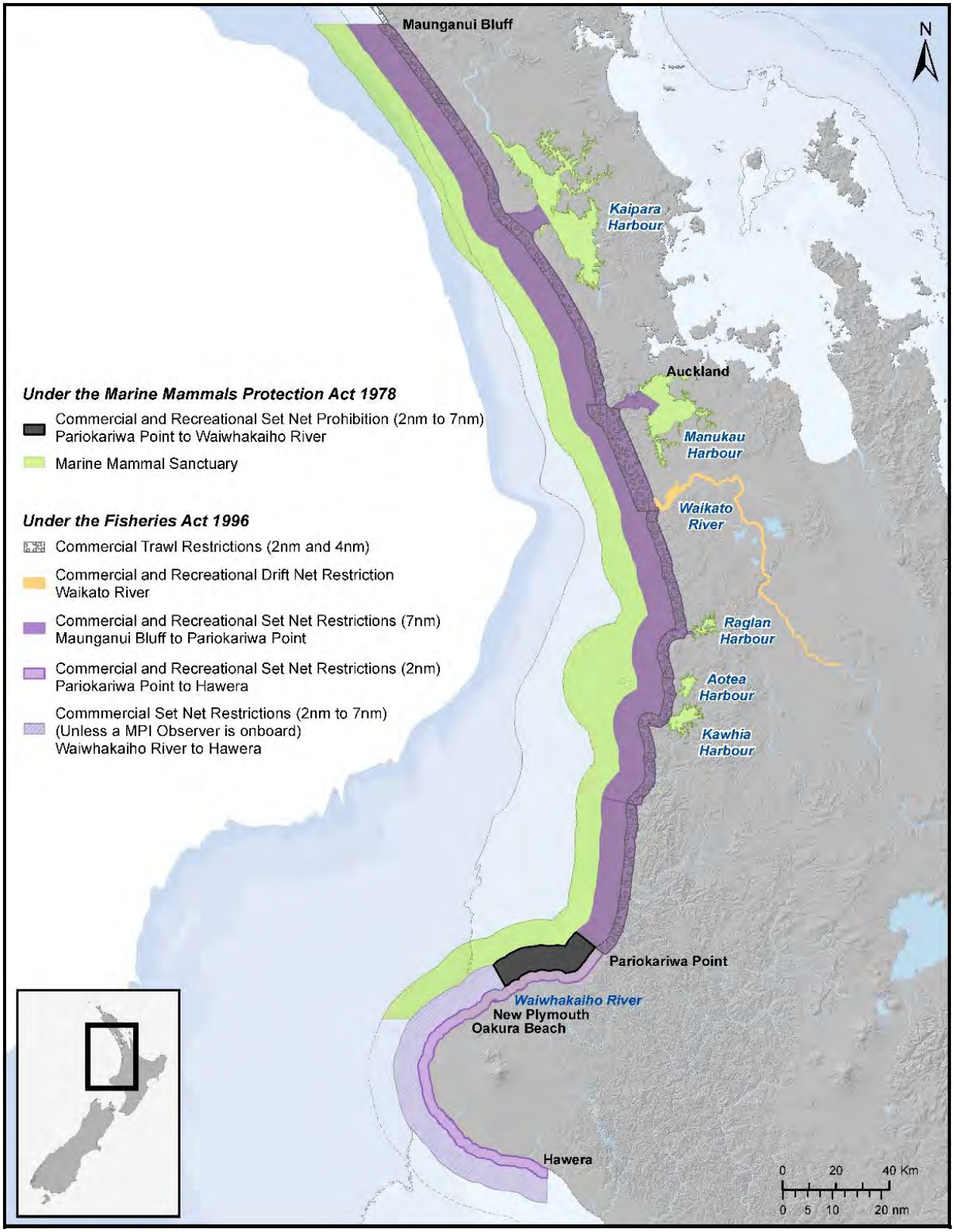


Figure 6.8: Summary of restrictions on commercial and amateur set netting on the WCSI. For a full description of the restrictions, see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

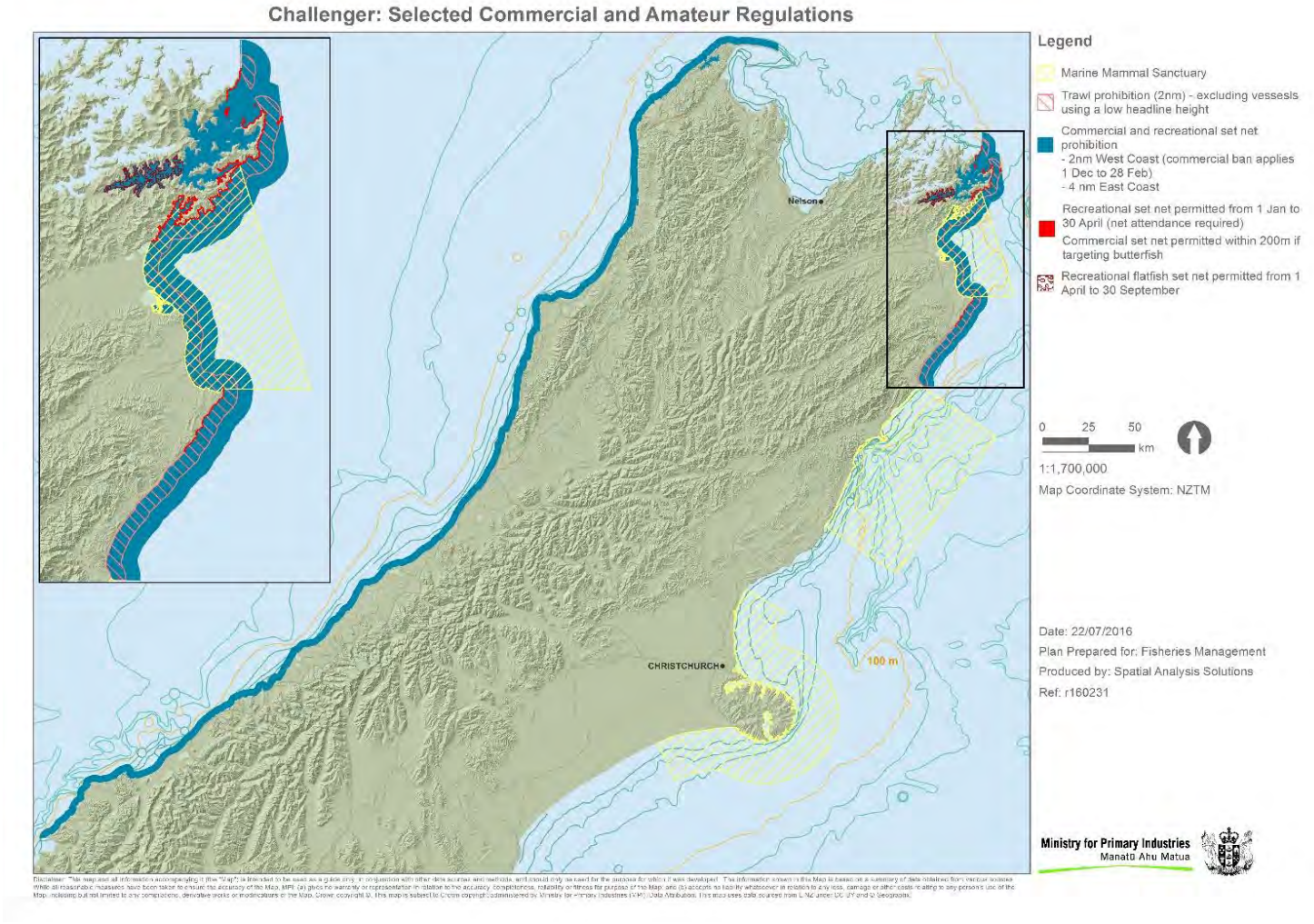


Figure 6.9: Summary of restrictions on commercial and amateur set netting, and commercial trawling in the Challenger area, west coast and northern east of New Zealand. For a full description of the restrictions, see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

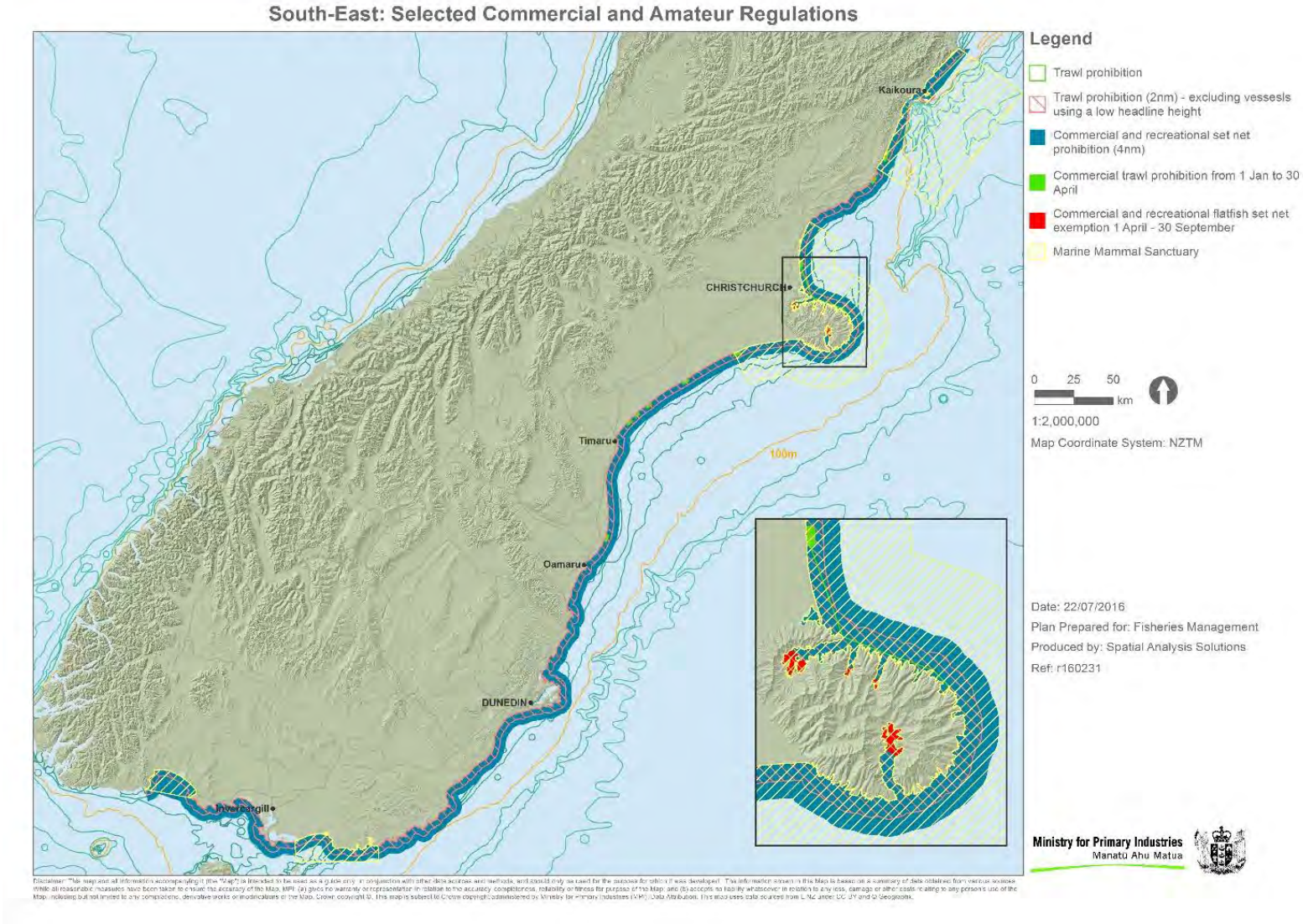


Figure 6.10: Summary of restrictions on commercial and amateur set netting, and commercial trawling in the south-east of New Zealand. For a full description of the restrictions, see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

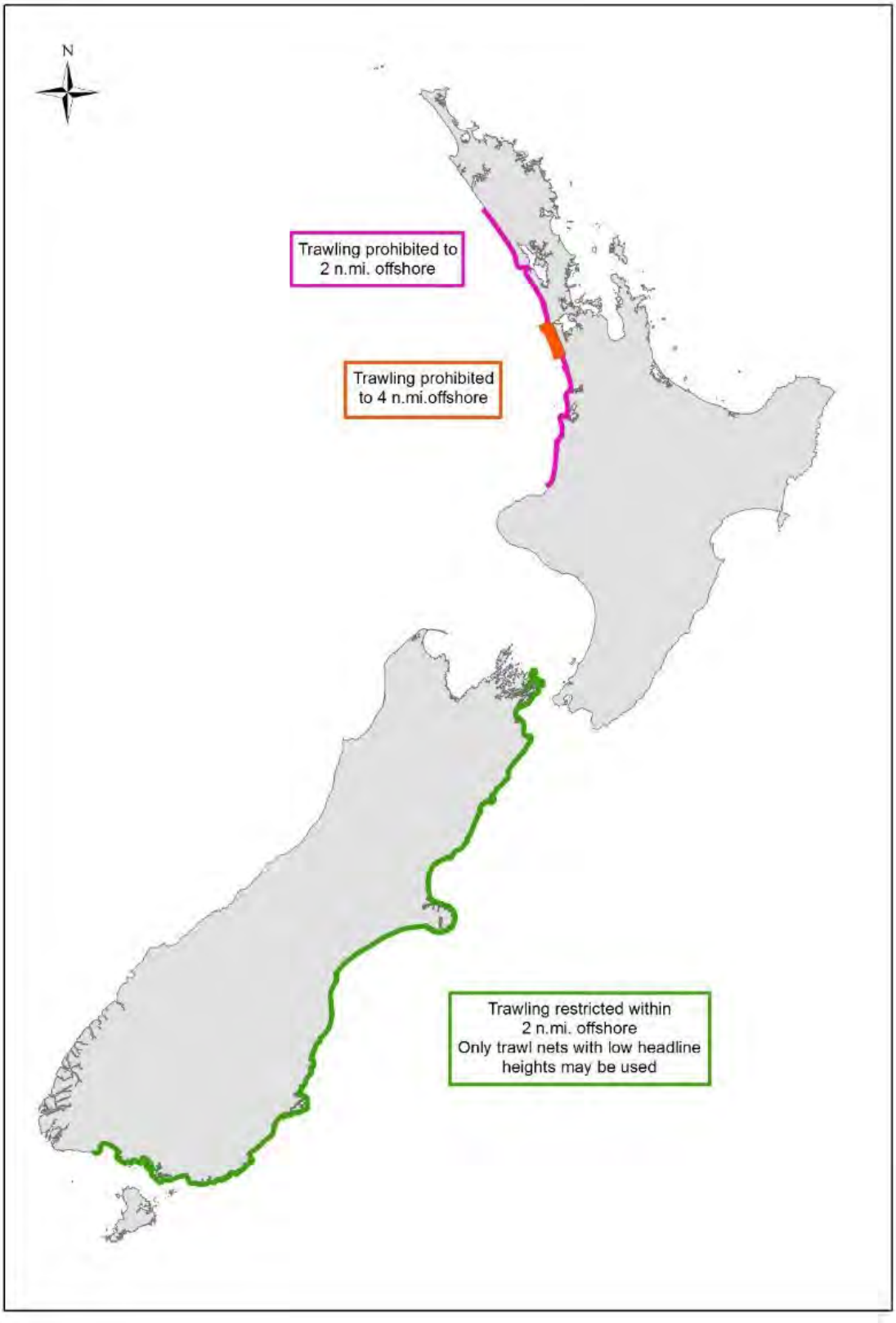


Figure 6.11: Summary of restrictions on trawling. For a full description of the restrictions see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

Assessing the degree of coverage of Hector's and Māui dolphin distribution afforded by spatial management measures is not straightforward as dolphin distributions are dynamic. Aerial surveys can be used to provide a broad-scale indication of dolphin distribution; however they only provide a static picture, strictly relevant to the time of the survey. Notwithstanding this limitation, it is possible to gain an indication of the proportion of a population that was within or outside a particular area at the time of an aerial survey from the proportion of on-effort sightings that were made inside or outside the area. For example, Rayment et al. (2010; Figure 6.12) conducted aerial surveys of Hector's dolphins at Banks Peninsula from the coast to 15 nm offshore over three summers and winters. A significantly larger proportion of the population was sighted inside the 4 nm set-net restriction in summer (mean = 81%; s.e.: 3.60) than in winter (mean = 44%; s.e.: 3.60). Similar seasonal differences in distribution were observed during the recent ECSI aerial surveys (MacKenzie & Clement 2014; Figure 6.13). In the Banks Peninsula (BP) stratum, 45% of the local summer population and 26% of the local winter population were within the set-net fisheries restriction zones. In the Clifford and Cloudy Bay (CCB) stratum, 47% of the local summer population and 14% of the local winter population were within the set-net fisheries restriction zones Although a sizeable proportion of the sightings occurred within areas closed to set-net fishing during both surveys (Rayment et al. 2010, MacKenzie & Clement 2014), many sightings in summer and most sightings in winter occurred outside these areas.

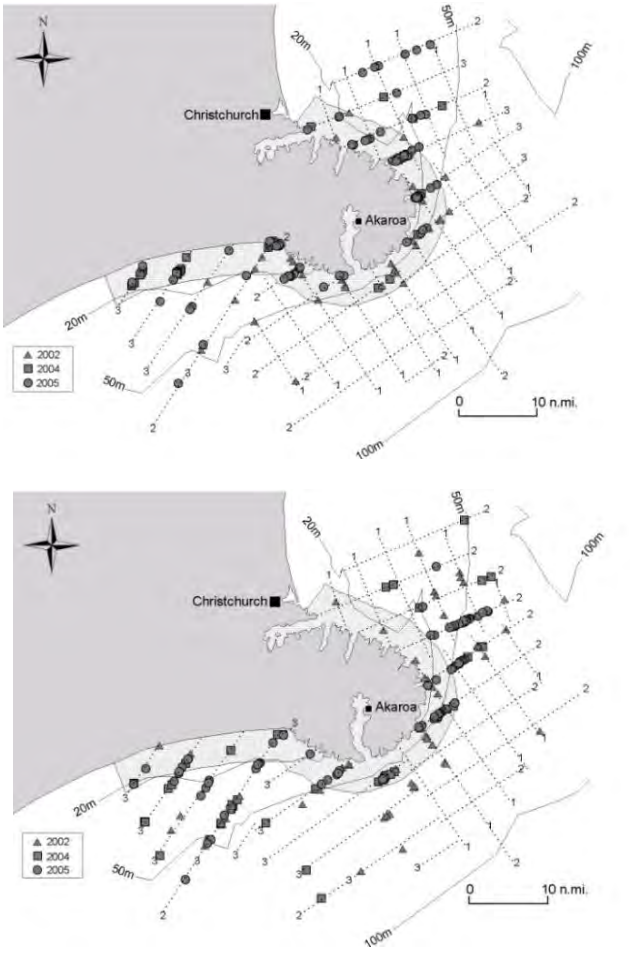


Figure 6.12: Transects and Hector's dolphin sightings on (top) three summer surveys, and (bottom) three winter surveys around Banks Peninsula. Numbers at the end of transect lines are the number of years each line was surveyed. Reproduced from Rayment et al. (2010).

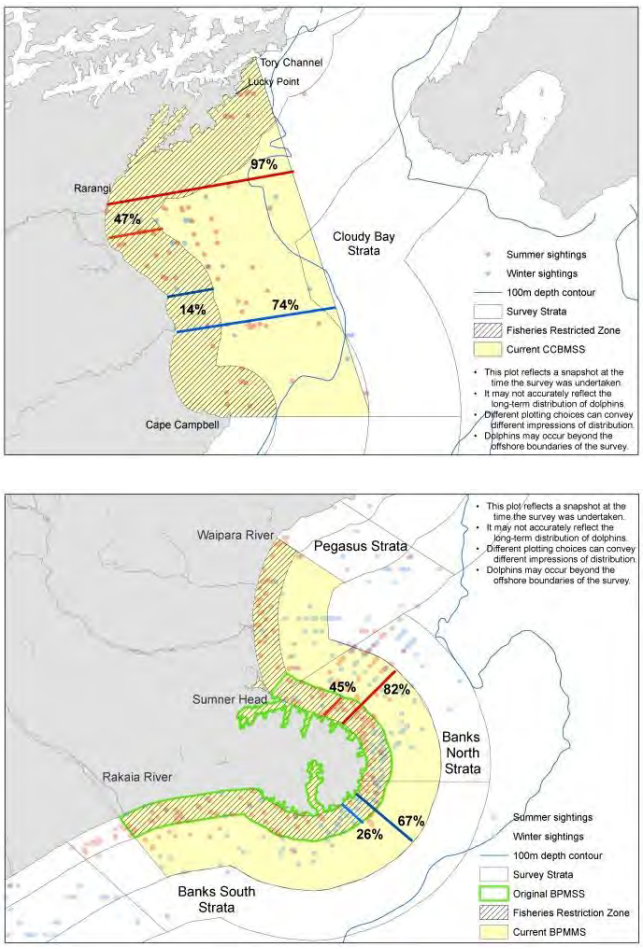


Figure 6.13: The location of summer (red) and winter (blue) survey sightings in relation to fisheries restriction zones and marine mammal sanctuary (MMS) boundaries around Clifford and Cloudy Bays (CCB, top) and Banks Peninsula (BP, bottom). Lines and associated percentages represent proportion of the local population found within 4 nm and 12 nm in summer (red) and winter (blue). Reproduced from MacKenzie & Clement (2014).

6.4.3 PREVIOUS MODELS OF POPULATION-LEVEL FISHERIES IMPACTS

A number of modelling exercises have aimed to assess the effect of various proposed management approaches on the future population trajectory of Hector's and Māui dolphins. Most of this work has been published in science journals (Martien et al. 1999, Burkhart & Slooten 2003, Slooten 2007a, Slooten & Dawson 2010) using their respective peer-review processes, but Davies et al.'s (2008) analysis was reviewed by the AEWG and published as a research report.

The various models share some necessary similarities given the available information:

- Each assumes a particular form of population model and uses this to project dolphin numbers forward and backward from a single population estimate;
- None of the models used the most recent survey estimates of abundance and distribution in SCSI and ECSI;
- None of these models used spatially explicit estimates of overlap with fisheries to estimate encounter rate and capture rate per encounter, instead a single estimate of dolphin capture rate from the ECSI was applied to historical fishing effort patterns to estimate fishing-related dolphin mortalities for all four populations.

Martien et al. (1999) employed a simple logistic ('Schaefer') population model and projected numbers back to 1970, and forward 200 years, from the 1985 abundance estimate published by Dawson & Slooten (1988). Three separate populations were modelled (WCNI, WCSI and a population that included both ECSI and SCSI populations). Using Dawson's (1991) estimates of mortality from the ECSI area, the back calculation suggested a total of 7077 dolphins across the three populations in 1970, if maximum

population growth rate was 4.4%, and 7957 if maximum population growth rate was 1.8% per annum. Martien et al. (1999) considered that the 1985 estimate of abundance was likely to be a slight underestimate (because transects to assess offshore distribution extended only 5 nm offshore), but suggested that any resulting bias in the estimate of the level of the population as a proportion of carrying capacity was likely to be small. The ESCI population was projected to increase for all combinations of parameters except when the maximum growth rate was set to 1.8%.

Davies & Gilbert (2003) conducted a risk assessment for Māui dolphins using a spatially and temporally stratified, age-structured, Bayesian population model for ECSI Hector's dolphins, a population thought to have similar biological and productivity characteristics to Māui dolphin. Estimated population productivity was highly uncertain and largely driven by the priors. Strong assumptions were needed to translate the ECSI model to a model for Māui dolphin and to model population distribution and abundance off the WCNI. Davies & Gilbert found the probability of population decline to be high (50 to 90%) assuming the distribution and intensity of fishing effort pertaining at the time, but the predicted performance of alternative management strategies was sensitive to assumptions about movement, adult survival rate, and set-net catchability. In February 2003 the Ministry of Fisheries introduced closures off WCNI to reduce the risk to Māui dolphins.

Burkhart & Slooten (2003) developed a stochastic version of the logistic model to include a wider range of parameters, variation in fishing effort and population growth, and smaller population units (16 closed populations). Using the same survey and mortality estimates as Martien et al. (1999) yielded similar estimates of the total 1970 population size, but disaggregation of the population into smaller units allowed a conclusion that only the Banks Peninsula subpopulation was likely to increase.

Slooten (2007a) used the stochastic version of the logistic model, the 1998–2003 series of abundance estimates, and catch rates from a 1998 observer programme and concluded a markedly higher estimate of 29 316 individuals in 1970 (CV = 0.26). Slooten's (2007a) projections under status quo management suggested that populations in many areas, including Banks Peninsula, would decrease, but that the WCNI population would increase. Middleton et al. (2007) criticised the high level of confidence ascribed by Slooten (2007a) to her model results without acknowledging that (i) these were dependent on particular model assumptions and (ii) failed to consider other relevant data. In response, Slooten (2007b) gave more detail of her modelling choices, suggested that they were unlikely to lead to overestimation of the impact of fishing, and pointed to similarities between her results and those of other work that was close to being finalised at the time (Davies et al. 2008).

The modelling conducted by Davies et al. (2008) built on the work by Davies & Gilbert (2003) and comprised a Bayesian age-structured population model for the Banks Peninsula subpopulation and 100-year projection simulations for all four subpopulations under different assumed management regimes. The BP population model was structured by age, area, and seasonally to account for the behaviour of the dolphins and the fishery, had a density-dependent calving rate (maximum one calf per female every two years). It was fitted to an absolute abundance estimate from the 1998–2000 surveys of the ECSI, a time series of relative abundance indices for 1990 to 1996 from mark-recapture analyses of dolphin re-sightings around Banks Peninsula, an estimate of average annual adult survival rate 1985–2002, information on the age at first reproduction, the age composition of entangled dolphins, the catch of dolphins recorded by relevant observers, and the amount and distribution of relevant commercial set-net fishing since 1970. Sensitivity to key assumptions was explored by fitting models based on alternative assumptions and by omitting some datasets.

Because so few data were available on the dolphin population and bycatch, Davies et al. (2008) required informative priors to fit their Banks Peninsula model. Even so, the posterior distributions of most parameters were broad and were sensitive to key assumptions, suggesting great uncertainty in our understanding of historical dolphin population dynamics and current population status. Estimates of potential population growth rate ranged from close to zero to the upper bound of what is biologically feasible. The stochastic 100-year projections for each subpopulation entail additional uncertainty, only some of which could be captured in the simulations.

The AEWG agreed that:

- The outcomes of different management strategies could not be predicted with any certainty and, for all subpopulations and management strategies

modelled, future population increases and decreases were both plausible.

- Taking the modelling results at face value, all three subpopulations of Hector's dolphin were more likely to decline than increase under set-net fishing effort pertaining at the time, and the decline could be substantial. Conversely, under all alternative strategies simulated, all three subpopulations of Hector's dolphins were more likely to increase than decrease.
- The results for ECSI, including Banks Peninsula, were likely to be more reliable.
- The predicted rates of increase or decrease of all subpopulations were sensitive to the assumed level of productivity.
- For Māui dolphins, the management regime at that time included substantial protection, and the likelihood of continued decline depended strongly on the assumed level of productivity.
- The available data had been used in the best possible way and had been found not to be sufficient to support a definitive analysis. However, the modelling provided helpful guidance on areas where new information should be collected to reduce our uncertainty.
- If the risk analysis was to be communicated to managers, it should be with appropriate caveats around its shortcomings and uncertainty.

The AEWG could not agree whether it was reasonable to adopt all the assumptions required but, consistent with the Terms of Reference, the Chair of the AEWG decided that the modelling could provide qualitative guidance to managers as a risk assessment. He added that the predicted rates of change for all Hector's and Māui subpopulations were sensitive to the assumed level of productivity but, except at the lowest level of productivity, the differences between the predicted outcomes of strategies other than status quo were modest. He noted that, at the lowest assumed level of productivity, projections suggested that the small SCSI subpopulation was more likely to decrease than increase under all simulated management measures other than zero fishing mortality, and that population was also quite likely to be affected by depensation (increasingly low population productivity as abundance decreases, also called an Allee effect).

The stochastic logistic model was used by Slooten & Dawson (2010) to assess the effect of management options developed for the Hector's and Māui Dolphin Threat Management Plan (although the options evaluated differed from the final proposals). The input data were similar to those of Slooten (2007a, 2007b). Slooten & Dawson's (2010) population estimates for 1970 (their figure 1) were similar to those reported by Slooten (2007a), but showed some regional differences. Both Slooten (2007a) and Slooten & Dawson (2010) suggested that the WCNI population would increase under management pertaining at the time, whereas the other three populations would decline. Slooten & Dawson (2010) further suggested that their option B (similar to the 2008 measures) would lead to the ECSI and SCSI populations increasing on average, whereas the WCSI population would continue to decline.

Slooten & Davies (2012) published a new estimate of 23 captures from the ECSI population between May 2009 and April 2010 based on observer records (although their description of the methods suggests that their reported CV of 21% is greatly underestimated). They used this and an estimate of 110–150 dolphins caught annually around the South Island before 2008, including 35 to 50 dolphins caught off the ECSI (Davies et al. 2008) to update the two most recent modelling approaches (Davies et al. 2008, Slooten & Dawson 2010). Slooten & Davies (2012) found the consistent predictions from all population models used to date surprising, given the substantial differences in their structural assumptions. They noted that all population models indicated that substantial declines had occurred and were likely to continue, and concluded that this consistency should add confidence to the predictions about the consequences of the different management options. In addition, they also cited a number of reasons why the conclusions might be optimistic, notably that most only include incidental captures in commercial set nets, as the other forms of fisheries-related mortality have yet to be quantified (Davies et al. 2008, Slooten & Dawson 2010, Slooten & Davies 2012).

6.4.4 MULTI-SPECIES MARINE MAMMAL RISK ASSESSMENT

At the time of the Māui dolphin risk assessment described in Currey et al. (2012), fishery threats were only estimated subjectively, because comprehensive spatial distributions had not yet been estimated for Māui and Hector's dolphins, and fisheries observer coverage was low, especially for set-net fisheries. However with the subsequent availability of aerially or expert derived spatial distributions (Figures 6.1 and 6.6) and development of the SEFRA method (Chapter 3) fisheries risk can now be estimated empirically based on

spatial overlap between dolphins and fishing effort, and observed capture rates.

The first implementation of the New Zealand Marine Mammal Risk Assessment (MMRA) is described in Abraham et al. (2017). The MMRA estimates that Māui and Hector's dolphins are the third and fourth most at-risk subspecies of marine mammal, respectively, from New Zealand commercial fisheries (Figure 6.14). For Māui dolphin the mean estimated risk ratio is 0.47 (95% c.i.: 0.00–1.33). For Hector's dolphin the estimated mean risk ratio is 0.45 (95% c.i.: 0.18–0.92).[17]

Fisheries risk to Māui and Hector's dolphins is attributable primarily to set-net fisheries, and secondarily to inshore trawl (Figure 6.15). For Hector's dolphins, the mean estimate is of 9.0 fatalities per year in trawl fisheries (95% c.i.: 1.1–26.6) and 32.3 fatalities per year in set-net fisheries (95% c.i.: 13.8–65.8). For Māui dolphins, the mean estimate is of 0 (0–0.1) fatalities per year in trawl fisheries, and 0.2 (0.0–0.5) fatalities per year in set-net fisheries. These numbers include estimated observable captures and an allowance for cryptic mortality.

That estimated fishery deaths are considerably lower than previous estimates likely reflects: i) the improved methods employed, allowing empirical estimation with explicit consideration of spatially unrepresentative observer coverage; ii) improved data inputs with respect to population size and species spatial distribution, arising from South Island aerial surveys; and iii) changes in actual fisheries risk since the adoption of spatial fisheries closures in areas of highest dolphin density (noting that subjective expert estimates from Currey et al. (2012) likely also reflected risk in areas that are now closed to fishing).

The following sections are reproduced from Abraham et al. (2017).

6.4.4.1 HECTOR'S DOLPHIN

The estimated annual potential fatalities of Hector's dolphin were 41.3 (95% c.i.: 19.1–77.7), with 32.3 (95% c.i.: 13.8–65.8) estimated captures in set-net fisheries. The Hector's dolphin subspecies has three genetically distinct subpopulations: East Coast South Island, West Coast South Island, and South Coast South Island (Baker et al. 2002). The overlap between set-net fisheries and Hector's dolphin was almost entirely on the East Coast South Island. The current estimate of captures in set-net fisheries overlaps with a previous estimate of 23 Hector's dolphin (CV: 0.21) caught in east coast South Island set-net fisheries during 2009–10 (Slooten & Davies 2012). The risk to Hector's dolphin was entirely less than one, however, there was a 9.4% probability that the risk to the east coast South Island Hector's dolphin population from set-net fishing in that area exceeded one.

There are already extensive areas of South Island waters where set-net fishing is prohibited, including restrictions within 4 nm (7.4 km) off the coast, and a larger marine mammal protection area surrounding Banks Peninsula. The effect of these restrictions is evident on the map of overlap between Hector's dolphin and set-net fisheries, as there is no overlap close to shore along most of the South Island coast. The set-net closures are reflected in the fishing effort used for the risk assessment (from the period 2012–13 to 2014–15).

For Hector's dolphin, the uncertainty in the risk was high. This uncertainty were partly due to the low observer coverage of set-net fisheries: around half of the observed captures were from a dedicated programme that was carried out in the late 1990s (Baird & Bradford 2000, Starr & Langley 2000). Trials of video monitoring on set-net vessels have demonstrated that Hector's dolphin bycatch can be recorded by video cameras (e.g., McElderry et al. 2007), and it appeared that the video was able to record captures that would not have been seen by observers on the vessels. Expanding observer coverage (either via human observers or video monitoring) would help to reduce uncertainty in the estimated captures. In addition, improving knowledge of cryptic mortality would reduce uncertainty, as estimated annual potential fatalities are in part associated with an assumed cryptic mortality. For set-net fisheries, the probability that a capture incident is observed was assumed to be uniformly distributed

[17] These risk ratios reflect a Population Sustainability Threshold in the risk ratio formula applying a default population reference outcome for the multi-species risk assessment, i.e., 'recovery to or stabilisation at a level higher than 50% of carrying capacity, with 90% certainty'. Note however that in other species-specific implementations of the SEFRA method, alternative population reference outcomes may be adopted, reflecting species-specific policy choices.

between one-third and one, with a mean of two-thirds. Obtaining better information on the drop-out rates, i.e., the proportion of dolphin that are caught but not recovered onboard the vessel, would help reduce the uncertainty. Another source of uncertainty in the estimation of risk to Hector's dolphin was the estimation of the maximum growth rate (r_{max}; mean 0.026; 95% c.i.: 0.018–0.036), which relied on expert judgement. Research to estimate r_{max} empirically from demographic data or life-history parameters may be useful to reduce this uncertainty. The analysis of overlap relied on surveys conducted during two seasons (winter and summer), and no information was available on the variation in the distribution of Hector's dolphin within those seasons.

From the observed set-net captures, the model estimated a live-release probability of 12.8% (95% c.i.: 3.7–26.4%) for small dolphins in set-net fisheries. The post-release survival of these animals was unknown, and so was assumed to be uniformly distributed between zero and one, with a mean of one-half. Taken together, around 6% of the capture incidents were assumed to not have resulted in a fatality.

6.4.4.2 MĀUI DOLPHIN

The mean and median values of the risk ratio for Māui dolphin were below 0.5, but the upper credible limit extended above one. Set-net fisheries were the only fisheries with a mean of more than 0.05 annual potential fatalities of Māui dolphin, highlighting that efforts to reduce the potential capture of this species in fisheries need to focus on set-net fishing.

The estimated overlap between set-net fisheries and Māui dolphin was concentrated inside harbours on the North Island west coast. In the region of overlap, much of the fishing effort had the location imputed. To help refine the overlap, and consequently the estimated captures, it is necessary to clarify where fishing effort in these harbours is occurring, and how the distribution of Māui dolphin extends into them. Uncertainty in estimates of risk to Māui dolphin also reflected uncertainty in the maximum growth rate, r_{max}, which was estimated by experts as mean 0.023 (95% c.i.: 0.015–0.034).

Based on the assumed distribution of Māui dolphin, the risk assessment suggests that potential fatalities of Māui dolphin would be reduced by extending the set-net ban into North Island west coast harbours, particularly Kaipara, Raglan, Aotea, and Kāwhia harbours. There was also overlap with set-net fisheries operating near New Plymouth, toward the south of the range of Māui dolphin. Conclusions from the current study are sensitive to assumptions about the distributions of Māui dolphin and of unlocated set-net effort within harbours. The vulnerability of Māui dolphin was assumed to be the same as for Hector's dolphin (they were treated as the same species in the model). The estimated capture rate largely depended on observations made on the South Island east coast of Hector's dolphin. Since 2012–13, there has been observer coverage of set-net fisheries in the Taranaki area, focused on the warehou set-net fishery that operates near New Plymouth. Between 2012–13 and 2014–15, observer coverage of the minor species set-net fishery (which includes warehou targets) in the Taranaki region has varied between 38% and 73%. There were no observed captures of Māui dolphin.

6.4.5 ONGOING AND SPECIES SPECIFIC RISK ASSESSMENT

The MMRA was based on the SEFRA method described in Chapter 3, but included some modifications with particular relevance for Hector's and Māui dolphins. For example the model in Abraham et al. (2017) included both a 'species group' variable (in which Hector's and Māui dolphins were grouped with other small dolphins) and a species-specific variable (with Hector's and Māui dolphins recognised as a single species). Crucially, concerns raised by the expert review of Lonergan et al. (2017) about the reliance on Delphi-derived species spatial distributions were not applicable to Māui and Hector's dolphin specifically, because the MMRA instead utilised the distributions shown in Figures 6.1 and 6.6. However the 'small dolphin' species group variable also included other species such as common dolphin for which empirically derived distributions were not available. To the extent that the species group variable affects the estimated vulnerability of Hector's and Māui dolphins, captures and risk estimates for these species may still reflect Delphi-derived distributions. These estimates will be updated when new cetacean distributions are available (from PRO2014-01).

For Māui dolphins in particular, the current MMRA risk estimates remain sensitive to assumptions about to what extent the dolphin distribution extends offshore and into the mouths of North Island harbours. Research is underway under project PRO2017-12 to refine the estimated spatial distribution in Figure 6.1 using spatial modelling informed by acoustic detections and sightings data. Māui dolphin risk estimates are also sensitive to the spatial distribution of small vessel set-net fishing effort in harbours (which historically has only been reported within Statistical Areas, not with precise locations). As positional effort reporting improves, the risk associated with harbour set-net fishing can be estimated with higher precision.

The current MMRA treats risk to Hector's dolphins as a single species, but overlap with fisheries varies between subpopulations. Separately, under project SEA2016-30, MPI is developing the capacity to disaggregate and query spatial risk assessment outputs on a regional basis for Hector's and Māui dolphins, and to examine sensitivities

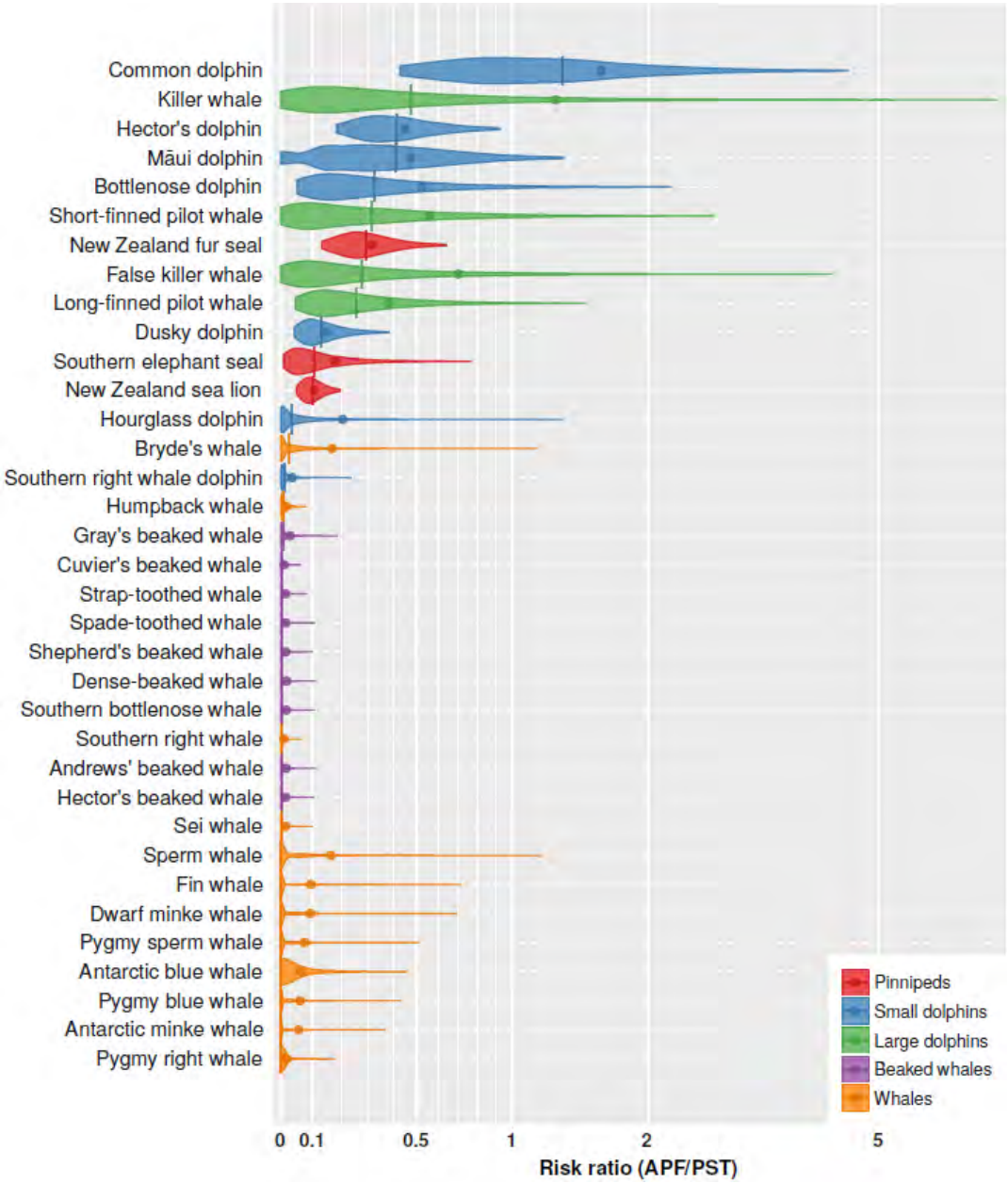


Figure 6.14: Risk ratio for New Zealand marine mammals, calculated as the ratio of fishery-related deaths (APF) to the Population Sustainability Threshold (PST). Values are displayed on a logarithmic scale, and the distribution of the risk ratios within their 95% credible interval indicated by the coloured shapes, including the median risk ratio (vertical line). Species are listed in decreasing order of the median risk ratio.

(e.g., alternative spatial distributions of dolphins or of fishing effort). Outputs of this project will allow managers to consider the consequences of alternative risk management scenarios, to inform the update of the TMP in 2018.

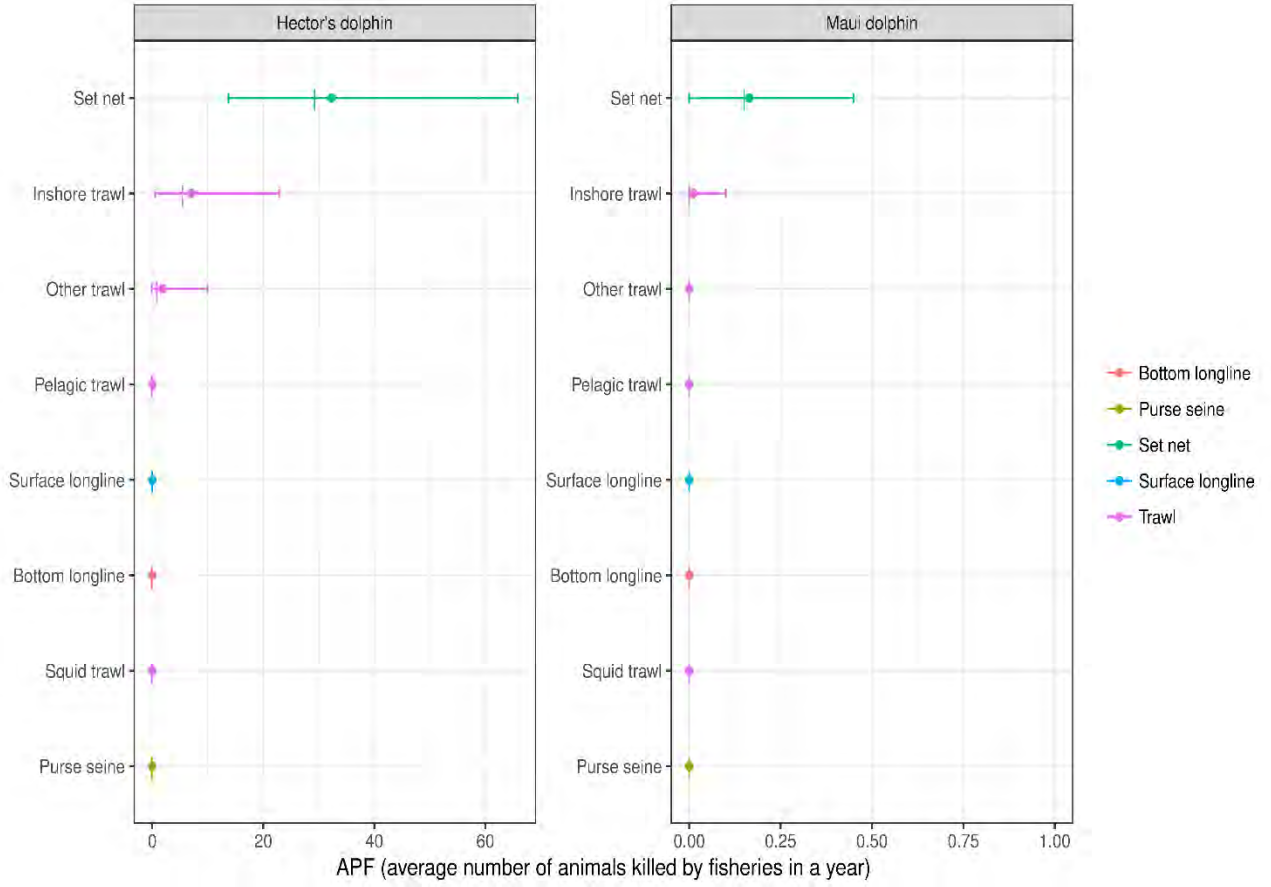


Figure 6.15: Estimated annual fishery-related deaths of Hector's dolphins and Māui dolphins by fishery group, from the multi-species New Zealand Marine Mammal Risk Assessment (Abraham et al. 2017).

6.4.6 SOURCES OF UNCERTAINTY

The uncertainties and assumptions in the modelling by Davies et al. (2008), Slooten & Dawson (2010), and Slooten & Davies (2012) were reviewed in detail by Slooten & Davies (2012). The models incorporate uncertainties in parameter distributions and hence population estimates are presented with their estimated levels of precision. The population viability analyses incorporated a distribution for population growth rate based on a wide range of values for maximum growth rate in Hector's dolphin (e.g., Slooten et al. 2000) and the Bayesian population models included a fully integrated parameter estimation of fisheries-related mortality and reproductive rate (Slooten & Davies 2012). Slooten & Dawson (2010) showed via sensitivity analysis that the probability of recovery to half the maximum population size was robust to uncertainty in the catch rate (± 0.25 times the assumed catch rate of 0.037 dolphins per set) used in the PVAs.

The AEWG discussed outstanding areas of uncertainty and concluded that the following areas represented important uncertainties in assessing the impacts of fishing on Hector's and Māui dolphins.

CAPTURE ESTIMATES AND CAPTURE RATE

Increased observer coverage, using a combination of observers and electronic monitoring, for set-net and inshore trawl fisheries is needed to reduce uncertainty in the estimates of captures and capture rate. Observer coverage should cover a sufficiently high proportion of fishing effort so as to enable the detection of rare events (particularly important for Māui dolphin), and to minimise

the risk of mistaken risk estimation arising from non-representative coverage.

CRYPTIC MORTALITY

The level of cryptic mortality associated with fisheries interactions is unknown for Hector's and Māui dolphins, but may be non-trivial if estimates for other small cetaceans are any indication (e.g., 58% of captured porpoises falling out of a net before reaching the deck; Kindt-Larsen et al. 2012). Improving our understanding of cryptic mortality is a priority for future risk assessments for Hector's and Māui dolphins.

DEMOGRAPHIC PARAMETERS

All the various risk analyses completed to date rely, at least in part, on demographic data obtained from one part of one population (i.e., Banks Peninsula). This necessitates assumptions as to how these data, and the resulting parameter estimates, were derived and how they apply outside the Banks Peninsula region. Obtaining additional demographic data from other region(s) could enable any difference between regions to be detected and reflected in future risk analyses. However, robust estimation of demographic parameters will require analysis of long-term (more than 10 years) of mark-recapture data collection to produce a time series of photographic or genetic individual identifications. A review of relevant demographic parameters and their derivation will be completed in 2018 as part of project PRO2017-12.

POPULATION ESTIMATES FOR THE SCSI POPULATION

Recent estimates of abundance are available for all populations of Hector's and Māui dolphins (Clement et al. 2011, Hamner et al. 2012b, MacKenzie & Clement 2014 and 2016). The reanalysis of the survey data for the SCSI indicated a notably smaller estimate (median = 238; 95% c.i.: 113–503) than that obtained by Clement et al. (2011) (628; 95% c.i.: 301–1311). An updated abundance estimate for the SCSI population will be obtained under project PRO2016-09.

POPULATION CONNECTIVITY AND MOVEMENT

Ongoing photo-ID research (e.g., Bräger et al. 2002, Rayment et al. 2009b) and genetic recaptures (Oremus et al. 2012, Hamner et al. 2012a, 2012b, 2014) will improve estimates of movements and dispersal (Rayment et al. 2009b, Hamner et al. 2012a, 2012b, Pichler 2002). For example, Hamner et al. (2014) suggested that failure to protect the habitat between the North and South Island will reduce the likelihood of dispersal, possibly to the detriment of Māui dolphin.

SPATIAL DISTRIBUTIONS

Application of the SEFRA method relies upon accurate estimation of species spatial distributions, including seasonal variation (if any). These are available from aerial surveys for South Island Hector's dolphin populations, but due to low population size aerial survey alone was judged to be inadequate to estimate Māui dolphin distributions. Where populations are low and even rare events may produce substantial risk there is a need to estimate risk even in the low-density 'tails' of the spatial distribution where animals occur only rarely. This work will be progressed via spatial distribution modelling under project PRO2017-12.

OTHER THREATS (NON-FISHING-RELATED, INDIRECT, SUB-LETHAL, CUMULATIVE)

Uncertainty exists over the magnitude of impacts faced by Hector's and Māui dolphins due to disease, mining and hydrocarbon extraction, tourism, vessel traffic, anthropogenic noise, pollution, aquaculture and research activities (DOC & MFish 2007, Currey et al. 2012, MPI & DOC 2012). Even if the impacts in isolation are sub-lethal, it is unknown whether the effects are cumulative, how they might affect factors such as breeding success, and whether they interact with the direct and indirect threats due to fishing (DOC & MFish 2007, Currey et al. 2012). Roe et al. (2013) identified infection with *Toxoplasma gondii* as a factor potentially contributing to the population decline of Hector's and Māui dolphins, and recommend further investigation of the source and route of entry of pathogens into the coastal environment. This work is in progress and will be summarised and considered in a spatially explicit manner under project PRO2017-12, to inform the update of the TMP.

6.4.7 POTENTIAL INDIRECT THREATS

Miller et al. (2013) note that red cod is targeted by the inshore trawl fishery and its abundance is highly variable, particularly around Banks Peninsula. Given that red cod contribute most in terms of mass to the diet of Hector's dolphins on the ECSI, Miller et al. (2013) suggest that further research is required to investigate potential trophic effects of fisheries on Hector's dolphin populations.

6.5 INDICATORS AND TRENDS

Population size	Māui dolphins: 55 (95% c.i.: 48–69) in 2010–11[18] 63 (95% c.i.: 57–75) in 2015–16[19] ECSI Hector's dolphins: Annual median estimate: 8968 (s.e.: 1377; 95% c.i.: 6649–12 096) Seasonal estimate: 9728 (CV: 17%; 95% c.i.: 7001–13 517) in summer 2012–13 and 8208 (CV 27%; 95% c.i.: 4888–13 785) in winter 2013 (out to 20 nm)[20] WCSI Hector's dolphins: Annual estimate: 5388 (CV = 21%; 95% c.i.: 3613–8034) in 2000–01 (out to 4 nm)[21] Annual median estimate: 5642 (s.e.: 936; 95% c.i.: 4085–7792) Seasonal estimate : 5490 (CV: 26%; 95% c.i.: 3319–9079) in summer and 5802 (CV: 21%; 95% c.i.: 3879–8679) in winter (out to 20 nm) [20] SCSI Hector's dolphins: Annual median estimate: 238 (s.e.: 94; 95% c.i.: 113–503) in 2011[20] Seasonal estimates: 177 (CV: 37%; 95% c.i.: 88–358) in March 2011, 299 (CV: 47%; 95% c.i.: 125–714) in August 2011[20]
Population trend	Māui dolphins: Declining over longer time period although some evidence of stabilisation from 2010–11 to 2015–16. ECSI Hector's dolphins: Probably declining. Inconsistent evidence from abundance estimates, risk analyses and demographic estimates of population growth rate. WCSI Hector's dolphins: Unknown; ECSI estimates of vulnerability and productivity applied to this area via risk analyses suggest a much lower rate of capture, due to low overlap with fisheries. SCSI Hector's dolphins: Unknown. Inconsistent evidence from abundance estimates and risk analyses.
Threat status	Māui dolphins: NZ: Nationally Critical, Criterion A(1), Conservation Dependent in 2013[22] IUCN: Critically Endangered, Criteria A4c,d and C2a(ii) in 2013[23] Hector's dolphins: NZ: Nationally Endangered, Criterion C(1/1), Conservation Dependent in 2013[23] IUCN: Endangered, Criterion A4d in 2013[23]
Number of captures[24]	Māui dolphins: <1 per annum (Davies et al. 2008), 4.97 per annum (95% c.i.: 0.28–8.04; Currey et al. 2012), 0.2 estimated annual potential fatalities (APF) (95% c.i.: 0.00–0.5)[24] ECSI Hector's dolphins: 35 to 50 per annum (Davies et al. 2008) WCSI Hector's dolphins: 70 to 100 per annum (Davies et al. 2008) SCSI Hector's dolphins: about 2 per annum (Davies et al. 2008) All Hector's dolphins: 41.3 estimated annual potential fatalities (APF) (95% c.i.: 19.1–77.7)[24]
Trends in interactions	Possible reduction from 35 to 50 per annum (Davies et al. 2008) to about 23 for ECSI (Slooten & Davies 2012) and 21.6 for all South Island (Abraham et al. 2017).

[18] Hamner et al. 2012b.
[19] Baker et al. 2016b.
[20] MacKenzie & Clement 2016.
[21] Slooten et al. 2004.
[22] Baker et al. 2016a.
[23] Reeves et al. 2013b.
[24] For more information, see: http://data.dragonfly.co.nz/psc; Abraham et al. (2017).

6.6 REFERENCES

Abraham, E R; Berkenbusch, K; Neubauer, P; Richard, Y (2017) Assessment of the risk to New Zealand marine mammals from commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 189*. 123 p.

Archipelago Marine Research Ltd (2013) Using Electronic Monitoring to document inshore set net captures of Hector's dolphins. Report prepared for Ministry for Primary Industries. Retrieved from http://www.mpi.govt.nz/document-vault/14044.

Baird, S J; Bradford, E (2000) Estimation of Hector's dolphin bycatch from inshore fisheries, 1997/98 fishing year. Department of Conservation, Wellington. 28 p.

Baker, A N (1978) The status of Hector's dolphin *Cephalorhynchus hectori* (van Beneden) in New Zealand waters. *Reports of the International Whaling Commission 28*: 331–334.

Baker, A N; Smith, A N H; Pichler, F B (2002) Geographical variation in Hector's dolphin: recognition of a new subspecies of *Cephalorhynchus hectori*. *Journal of the Royal Society of New Zealand 32*: 713–717.

Baker, C S; Chilvers, B L; Childerhouse, S; Constantine, R; Currey, R; Mattlin, R; van Helden, A; Hitchmough, R; Rolfe, J (2016a) Conservation status of New Zealand marine mammals, 2013. *New Zealand Threat Classification Series 14*. Department of Conservation, Wellington. 18 p.

Baker, C S; Hamner, R M; Cooke, J; Heimeier, D; Vant, M; Steel, D; Constantine, R (2013) Low abundance and probable decline of the critically endangered Maui's dolphin estimated by genotype capture-recapture. *Animal Conservation 16*: 224–233.

Baker, C S; Steel, D; Hamner, R M; Hickman, G; Boren, L; Arlidge, W; Constantine, R (2016b) Estimating the abundance and effective population size of Māui dolphins using microsatellite genotypes in 2015–16, with retrospective matching to 2001–16. Department of Conservation, Auckland. 74 p.

Bejder, L; Dawson, S (2001) Abundance, residency and habitat utilisation of Hector's dolphin (*Cephalorhynchus hectori*) in Porpoise Bay, New Zealand. *New Zealand Journal of Marine and Freshwater Research 35*: 277–287.

Bejder, L; Dawson, S M; Harraway, J A (1999) Responses by Hector's dolphins to boats and swimmers in Porpoise Bay, New Zealand. *Marine Mammal Science 15*: 738–750.

Blezard, R H (2002) Observations of set net and inshore trawl fishing operations in the South Canterbury Bight, 2001. *DOC Internal Science Series 85*. Department of Conservation, Wellington.

Bordino, P; Kraus, S; Albareda, D; Fazio, A; Palmerio, A; Mendez, M; Botta, S (2002) Reducing incidental mortality of franciscana dolphin *Pontoporia blainvillei* with acoustic warning devices attached to fishing nets. *Marine Mammal Science 18*: 833–842.

Bräger, S (1998) Behavioural ecology and population structure of Hector's dolphin (*Cephalorhynchus hectori*). Unpublished PhD Thesis, University of Otago, Dunedin, New Zealand. 168 p.

Bräger, S; Dawson, S M; Slooten E; Smith, S; Stone, G S; Yoshinaga, A (2002) Site fidelity and along-shore range in Hector's dolphin, an endangered marine dolphin from New Zealand. *Biological Conservation 108*: 281–287.

Bräger, S; Harraway, J; Manly, B F J (2003) Habitat selection in a coastal dolphin species (*Cephalorhynchus hectori*). *Marine Biology 143*: 233–244.

Bräger, S; Schneider, K (1998) Nearshore distribution and abundance of dolphins along the West Coast of the South Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research 32*: 105–112.

Burkhart, S M; Slooten, E (2003) Population viability analysis for Hector's dolphin (*Cephalorhynchus hectori*): a stochastic population model for local populations. *New Zealand Journal of Marine and Freshwater Research 37*: 553–566.

Cameron, C; Barker, R; Fletcher, D; Slooten, E; Dawson, S M (1999) Modelling survival of Hector's dolphins around Banks Peninsula, New Zealand. *Journal of Agricultural, Biological and Environmental Statistics 4*: 126–135.

Cawthorn, M W (1988) Recent observations of Hector's dolphin, *Cephalorhynchus hectori*, in New Zealand. *Reports of the International Whaling Commission Special Issue 9*: 303–314.

Childerhouse, S; Rayment, W; Webster, T; Scali, S; du Fresne, S D (2008) Offshore aerial survey of Maui's dolphin distribution 2008. Auckland Conservancy, Department of Conservation. 6 p.

Clement, D; Mattlin, R; Torres, L (2011) Abundance, distribution and productivity of Hector's (and Maui's) dolphins. Final Research Report for project PRO2009-01A. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Cox, T M; Lewison, R L; Zydelis, R; Crowder, L B; Safina, C; Read, A J (2007) Comparing effectiveness of experimental and implemented bycatch reduction measures: the ideal and the real. Conservation Biology 21: 1155–1164.

Cox, T M; Read, A J; Solow, A; Tregenza, N (2001) Will harbour porpoises (*Phocoena phocoena*) habituate to pingers? *Journal of Cetacean Research and Management 3*: 81–86.

Currey, R J C; Boren, L; Sharp, B; Peterson, D (2012) A risk assessment of threats to Maui's dolphins. Ministry for Primary Industries and Department of Conservation, Wellington. 51 p.

Davies, N M; Bian, R; Starr, P; Lallemand, P; Gilbert, D J; McKenzie, J R (2008) Risk analysis of Maui's dolphin and Hector's dolphin subpopulations to commercial set net fishing using a temporal-spatial age-structured model. Ministry of Fisheries, Wellington. 113 p.

Davies, N M; Gilbert, D J (2003) A risk analysis of an endangered dolphin subspecies using a temporal-spatial age-structured model.

Final Research Report for Ministry of Fisheries research project MOF2002/03D, Objectives 1, 2 and 3. 42 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Dawson, S M (1991) Incidental catch of Hector's dolphin in inshore gillnets. *Marine Mammal Science 7*: 283–295.

Dawson, S; Fletcher, D; Slooten, E (2013a) Habitat use and conservation of an endangered dolphin. *Endangered Species Research 21*: 45–54.

Dawson, S M; Lusseau, D (2005) Pseudoreplication problems in studies of dolphin and porpoise reaction to pingers. *Marine Mammal Science 21*: 175–176.

Dawson, S; Northridge, S; Waples, D; Read, A J (2013b) To ping or not to ping: the use of active acoustic devices in mitigating interactions between small cetaceans and gillnet fisheries. *Endangered Species Research 19*: 201–221.

Dawson, S M; Slooten, E (1988) Hector's Dolphin *Cephalorhynchus hectori*: Distribution and abundance. *Reports of the International Whaling Commission Special Issue 9*: 315–324.

Dawson, S M; Slooten, E (1993) Conservation of Hector's dolphins: The case and process which led to the establishment of the Banks Peninsula Marine Mammal Sanctuary. *Aquatic Conservation: Marine and Freshwater Ecosystems 3*: 207–221.

Dawson, S M; Slooten, E (2005) Management of gillnet bycatch of cetaceans in New Zealand. *Journal of Cetacean Research and Management 7 (1)*: 59–64.

Dawson, S M; Slooten, E; DuFresne, S D; Wade, P; Clement, D M (2004) Small-boat surveys for coastal dolphins: Line-transect surveys of Hector's dolphins (*Cephalorhynchus hectori*). *Fishery Bulletin 102*: 441–451.

Department of Conservation and Ministry of Fisheries (2007) Hector's and Maui's Dolphin Threat Management Plan: Draft for Public Consultation. 298 p.

DOC (2013) Consultation on a proposed variation to the West Coast North Island Marine Mammal Sanctuary to prohibit commercial and recreational set net fishing between two and seven nautical miles offshore between Pariokariwa Point and the Waiwhakaiho River, Taranaki. 44 p.

DOC (2017a) Hector's and Maui's dolphin incident database. Retrieved from http://www.doc.govt.nz/our-work/hectors-and-maui-dolphin-incident-database.

DOC (2017b) Māui dolphin sightings. Retrieved from http://www.doc.govt.nz/our-work/our-work-with-maui-dolphin/maui-dolphin-sightings.

Du Fresne, S (2004) Conservation biology of Hector's dolphin. PhD Thesis, University of Otago, Dunedin, New Zealand. 166 p.

Du Fresne, S (2010) Distribution of Maui's dolphin (*Cephalorhynchus hectori maui*) 2000–2009. *DOC Research & Development Series 322*. Department of Conservation, Wellington.

Du Fresne, S; Burns, D; Gates, E (2012) An updated, annotated bibliography for Hector's (*Cephalorhynchus hectori hectori*) and Maui's (*C. hectori maui*) dolphins. *DOC Research and Development Series 332*. Department of Conservation, Wellington. 43 p.

Du Fresne, S; Mattlin, R (2009) Distribution and Abundance of Hector's Dolphin (*Cephalorhynchus hectori*) in Clifford and Cloudy Bays. Final Report for NIWA project CBF07401, prepared for Marine Wildlife Research Ltd.

Fairfax, D (2002) Observations of inshore trawl fishing operations in Pegasus Bay and the Canterbury Bight, 2002. *DOC Internal Science Series 86*. Department of Conservation, Wellington.

Ferreira, S M; Roberts, C C (2003) Distribution and abundance of Maui's dolphins (*Cephalorhynchus hectori maui*) along the North Island west coast, New Zealand. *DOC Science Internal Series 93*. 19 p.

Geijer, C K; Read, A J (2013) Mitigation of marine mammal bycatch in U.S. fisheries since 1994. *Biological Conservation 159*: 54–60.

Gormley, A M (2009) Population modelling of Hector's dolphin. Unpublished PhD Thesis, University of Otago, Dunedin, New Zealand. 217 p.

Gormley, A M; Dawson, S M; Slooten, E; Bräger, S (2005) Capture-recapture estimates of Hector's dolphin abundance at Banks Peninsula, New Zealand. *Marine Mammal Science 21*: 204–216.

Gormley, A M; Slooten, E; Dawson, S M; Barker, R J; Rayment, W; Du Fresne, S; Bräger, S (2012) First evidence that marine protected areas can work for marine mammals. *Journal of Applied Ecology 49*: 474–480.

Green, E; Charteris, C; Rodda, J (2007) Te Waewae Bay Hector's dolphins: abundance, distribution and threats. Southland Conservancy, Department of Conservation, Invercargill. 54 p.

Hamner, R M; Constantine, R; Mattlin, R; Waples, R; Baker, C S (2013) Genotype capture-recapture estimates of abundance and effective population size of Hector's dolphins in Cloudy Bay, New Zealand. SC/65a/SM07. 65th meeting of the International Whaling Commission Scientific Committee, Jeju, South Korea.

Hamner, R M; Constantine, R; Oremus, M; Stanley, M; Brown, P; Baker, C S (2014) Long range genetic movement by Hector's dolphins provides potential genetic enhancement for critically endangered Maui's dolphin. *Marine Mammal Science 30*: 139–153.

Hamner, R M; Oremus, M; Stanley, M; Brown, P; Constantine, R; Baker, C S (2012b) Estimating the abundance and effective population size of Maui's dolphins using microsatellite genotypes in

2010–11, with retrospective matching to 2001–2007. Department of Conservation, Auckland. 48 p.

Hamner, R M; Pichler, F B; Heimeier, D; Constantine, R; Baker, C S (2012a) Genetic differentiation and limited gene flow among fragmented populations of New Zealand endemic Hector's and Maui's dolphins. *Conservation Genetics 13*: 987–1002.

Hembree, D; Harwood, M B (1987) Pelagic gillnet modification trials in northern Australian seas. *Reports of the International Whaling Commission 37*: 369–373.

Hodgson, P (2002) South Island Hector's dolphin decisions. Ministry of Fisheries, Wellington. 13 p.

International Whaling Commission (2016a) Report of the Scientific Committee. Annex L. Report of the Sub-Committee on Small Cetaceans. *Journal of Cetacean Resource Management (supp) 17*: 357–384.

International Whaling Commission (2016b) Report of the Scientific Committee. 7–19 June 2016, Bled, Slovenia. IWC/6/Rep01. 138 p.

IUCN (2013) The IUCN Red List of Threatened Species. Version 2013.1. Retrieved from http://www.iucnredlist.org.

Kindt-Larsen, L; Dalskov, J; Stage, B; Larsen, F (2012) Observing incidental harbour porpoise *Phocoena phocoena* bycatch by remote electronic monitoring. *Endangered Species Research 19*: 75–83.

Kraus, S D; Read, A J; Solow, A; Baldwin, K; Spradlin, T; Anderson, E; Williamson, J (1997) Acoustic alarms reduce porpoise mortality. *Nature 388*: 525.

Lonergan, M E; Phillips, R A; Thomson, R B; Zhou, S (2017) Independent review of New Zealand's Spatially Explicit Fisheries Risk Assessment approach – 2017. *New Zealand Fisheries Science Review 2017/2*. 36 p.

McElderry, H; McCullough, D; Schrader, J; Illingworth, J (2007) Pilot study to test the effectiveness of electronic monitoring in Canterbury fisheries. *DOC Research & Development Series 264*. Department of Conservation, Wellington. 27 p.

MacKenzie, D; Clement, D (2014) Abundance and Distribution of ECSI Hector's dolphin. *New Zealand Aquatic Environment and Biodiversity Report No. 123*. 79 p.

MacKenzie, D; Clement, D (2016) Abundance and distribution of WCSI Hector's dolphin. *New Zealand Aquatic Environment and Biodiversity Report No. 168*. 67 p.

MacKenzie, D I; Clement, D; Mattlin, R (2012) Abundance, distribution and productivity of Hector's (and Maui's) dolphins. Final Research Report for project PRO2009-01B. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Martien, K K; Taylor, B L; Slooten, E; Dawson, S M (1999) A sensitivity analysis to guide research and management for Hector's dolphin. *Biological Conservation 90*: 183–191.

Martinez, E; Orams, M; Pawley, M; Stockin, K (2012) The use of auditory stimulants during swim encounters with Hector's dolphins in Akaroa harbour, New Zealand. *Marine Mammal Science 28*: E295–E315.

Middleton, D A J; Starr, P J; Gilbert, D J (2007) Modelling the impact of fisheries bycatch on Hector's dolphin: comment on Slooten (2007). *Endangered Species Research 3*: 331–334.

Miller, E; Lalas, C; Dawson, S; Ratz, H; Slooten, E (2013) Hector's dolphin diet: the species, sizes and relative importance of prey eaten by *Cephalorhynchus hectori*, investigated using stomach content analysis. *Marine Mammal Science 29*: 606–628. doi:10.1111/j.1748-7692.2012.00594.x.

Ministry for Primary Industries (2016) Independent review of prosecution decisions. Retrieved from http://www.mpi.govt.nz/law-and-policy/legal-overviews/fisheries/independent-review-of-prosecution-decisions.

Ministry for Primary Industries and Department of Conservation (2012) Review of the Maui's dolphin threat management plan. MPI & DOC Joint Discussion Paper No. 2012/18.

Oremus, M; Hamner, R M; Stanley, M; Brown, P; Baker, C S; Constantine, R (2012) Distribution, group characteristics and movements of the Critically Endangered Maui's dolphin *Cephalorhynchus hectori maui*. *Endangered Species Research 19*: 1–10.

Paul, L (2000) New Zealand Fishes: identification, natural history and fisheries. Reed Publishing, Auckland, New Zealand.

Perrin, W F; Reilly, S B (1984) Reproductive parameters of dolphins and small whales of the family Delphinidae. *Reports of the International Whaling Commission Special Issue 6*: 97–133.

Pichler, F B (2002) Genetic assessment of population boundaries and genetic exchange in Hector's dolphin. *DoC Science Internal Series 44*. Department of Conservation, Wellington. 37 p.

Pichler, F B; Baker, C S (2000) Loss of genetic diversity in the endemic Hector's dolphin due to fisheries related mortality. *Proceedings of the Royal Society B 267*: 97–102.

Pichler, F B; Dawson, S M; Slooten, E; Baker, C S (1998) Geographic isolation of Hector's dolphin populations described by mitochondrial DNA sequences. *Conservation Biology 12*: 676–682.

Ramm, K (2010) Conservation Services Programme Observer Report: 1 July 2008 to 30 June 2009. Final Draft. Department of Conservation, Wellington. 121 p.

Ramm, K (2012a) Conservation Services Programme Observer Report: 1 July 2009 to 30 June 2010. Final Report. Department of Conservation, Wellington. 130 p.

Ramm, K (2012b) Conservation Services Programme Observer Report: 1 July 2010 to 30 June 2011. Final Report. Department of Conservation, Wellington. 126 p.

Rayment, W; Clement, D; Dawson, S; Slooten, E; Secchi, E (2011a) Distribution of Hector's dolphin (*Cephalorhynchus hectori*) off the west coast, South Island, New Zealand, with implications for the management of bycatch. *Marine Mammal Science 27*: 398–420.

Rayment, W; Dawson, S; Scali, S; Slooten, E (2011b) Listening for a needle in a haystack: passive acoustic detection of dolphins at very low densities. *Endangered Species Research 14*: 149–156.

Rayment, W; Dawson, S; Slooten, E (2009a) Use of T-PODs for acoustic monitoring of *Cephalorhynchus* dolphins: a case study with Hector's dolphins in a marine protected area. *Endangered Species Research 10*: 333–339.

Rayment, W; Dawson, S; Slooten, E (2010) Seasonal changes in distribution of Hector's dolphin at Banks Peninsula, New Zealand: implications for protected area design. *Aquatic Conservation: Marine and Freshwater Ecosystems 20*: 106–116.

Rayment, W; Dawson, S; Slooten, E; Bräger, S; Du Fresne, S; Webster, T (2009b) Kernel density estimates of alongshore home range of Hector's dolphins at Banks Peninsula, New Zealand. *Marine Mammal Science 25*: 537–556.

Rayment, W; du Fresne, S D (2007) Offshore aerial survey of Maui's dolphin distribution 2007. Auckland Conservancy, Department of Conservation. 6 p.

Rayment, W; Webster, T (2009) Observation of Hector's dolphins associating with inshore fishing trawlers at Banks Peninsula, New Zealand. *New Zealand Journal of Marine and Freshwater Research 43*: 911–916.

Read, A J (2008) The looming crisis: interactions between marine mammals and fisheries. *Journal of Mammalogy 89*: 541–548.

Read, A J; Drinker, P; Northridge, S (2006) Bycatch of marine mammals in US and global fisheries. *Conservation Biology 20*: 163–169.

Reeves, R R; Dawson, S M; Jefferson, T A; Karczmarski, L; Laidre, K; O'Corry-Crowe, G; Rojas-Bracho, L; Secchi, E R; Slooten, E; Smith, B D; Wang, J Y; Zhou, K (2013a) *Cephalorhynchus hectori* ssp. maui. *In*: IUCN Red List of Threatened Species. Version 2013.1. Retrieved from www.iucnredlist.org.

Reeves, R R; Dawson, S M; Jefferson, T A; Karczmarski, L; Laidre, K; O'Corry-Crowe, G; Rojas-Bracho, L; Secchi, E R; Slooten, E; Smith, B D; Wang, J Y; Zhou, K (2013b) *Cephalorhynchus hectori*. *In*: IUCN Red List of Threatened Species. Version 2013.1. Retrieved from www.iucnredlist.org.

Reeves, R R; Smith, B D; Crespo, E A; Notarbartolo di Sciara, G (Comp.) (2003) Dolphins, Whales and Porpoises: 2002–2010 Conservation Action Plan for the World's Cetaceans. IUCN/SSC Cetacean Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK. ix + 139 p.

Richard, Y; Abraham, E R (2013a) Application of Potential Biological Removal methods to seabird populations. *New Zealand Aquatic Environment and Biodiversity Report No. 108*. 30 p.

Richard, Y; Abraham, E R (2013b) Risk of commercial fisheries to New Zealand seabird populations, 2006–07 to 2010–11. *New Zealand Aquatic Environment and Biodiversity Report No. 109*. 58 p.

Roberts, J. O; Webber, D. N; Goetz, K. T; Edwards, C. T. T; Roe, W. D; Doonan, I. J (2019) Spatial risk assessment of threats to Hector's and Māui dolphins (*Cephaloryhnchus hectori*). *New Zealand Aquatic Environment and Biodiversity Report No. XX*. 169 p.

Roe, W D; Howe, L; Baker, E; Burrows, L; Hunter, S (2013) An atypical genotype of *Toxoplasma gondii* as a cause of mortality in Hector's dolphins. *Veterinary Parasitology 192*: 67–74.

Rowe, S J (2009) Conservations Services Programme observer report, 01 July 2004 to 30 June 2007. DOC Marine Conservation Services Series 1. Department of Conservation, Wellington. 94 p.

Rowe, S J (2010) Conservations Services Programme observer report, 01 July 2007 to 30 June 2008. DOC Marine Conservation Services Series 4. Department of Conservation, Wellington. 98 p.

Russell, K (1999) The North Island Hector's dolphin: a species in need of conservation. Unpublished MSc Thesis, University of Auckland, New Zealand.

Scali, S (2006) Use of harbours by the critically endangered species Maui's dolphin (*Cephalorhynchus hectori maui*) Auckland Conservancy, Department of Conversation. 28 p.

Slooten, E (1990) Population biology, social organisation and behaviour of Hector's dolphin. PhD Thesis, University of Canterbury, New Zealand.

Slooten, E (1991) Age, growth and reproduction in Hector's dolphins. *Canadian Journal of Zoology 69*: 1689–1700.

Slooten, E (2007a) Conservation management in the face of uncertainty: effectiveness of four options for managing Hector's dolphin bycatch. *Endangered Species Research 3*: 169–179.

Slooten, E (2007b) Criticism is unfounded: reply to Middleton et al. (2007). *Endangered Species Research 3*: 335–339.

Slooten, E (2013) Effectiveness of area-based management in reducing bycatch of the New Zealand dolphin. *Endangered Species Research 20*: 121–130.

Slooten, E; Davies, N (2012) Hector's dolphin risk assessments: old and new analyses show consistent results. *Journal of the Royal Society of New Zealand 42*: 49–60.

Slooten, E; Dawson, S M (1994) Hector's dolphin *Cephalorhynchus hectori*. *In*: Ridgway, S H; Harrison, R (Eds.), Handbook of Marine Mammals. Volume V (Delphinidae and Phocoenidae). pp 311–333. Academic Press, New York.

Slooten, E; Dawson, S (2010) Assessing the effectiveness of conservation management decisions: likely effects of new protection measures for Hector's dolphin. *Aquatic Conservation: Marine and Freshwater Ecosystems 20*: 334–347.

Slooten, E; Dawson, S; Du Fresne, S (2001) Report on interactions between Hector's dolphins (*Cephalorhynchus hectori*) and a Golden Bay mussel farm. Unpublished report for Environment Canterbury, No. 11. 8 p.

Slooten, E; Dawson, S M; Lad, F (1992) Survival rates of photographically identified Hector's dolphins from 1984 to 1988. *Marine Mammal Science 8*: 327–343.

Slooten, E; Dawson, S M; Rayment, W J (2004) Aerial surveys for coastal dolphins: abundance of Hector's dolphins off the South Island West Coast, New Zealand. *Marine Mammal Science 20*: 477–490.

Slooten, E; Dawson, S M; Rayment, W J; Childerhouse, S J (2005) Distribution of Maui's dolphin, *Cepahalorhynchus hectori maui*. *New Zealand Fisheries Assessment Report 2005/28*. 22 p.

Slooten, E; Dawson, S M; Rayment, W; Childerhouse, S (2006) A new abundance estimate for Maui's dolphin: What does it mean for managing this critically endangered species? *Biological Conservation 128*: 576–581.

Slooten, E; Dawson, S M; Whitehead, H (1993) Associations among photographically identified Hector's dolphins. *Canadian Journal of Zoology 71*: 2311–2318.

Slooten, E; Fletcher, D; Taylor, B (2000) Accounting for uncertainty in risk assessment: case study of Hector's dolphin mortality due to gillnet entanglement. *Conservation Biology 14*: 1264–1270.

Slooten, E; Lad, F (1991) Population biology and conservation of Hector's dolphins. *Canadian Journal of Zoology 69*: 1701–1707.

Southeast Finfish Management Company (2000) Voluntary code of practice – commercial net setters. SE Finfish Management Ltd, Lyttelton. 22 p.

Stanley, M (2009) Maui's winter offshore aerial survey June/July 2009. Auckland Conservancy, Department of Conservation. 10 p.

Starr, P; Langley, A (2000) Inshore Fishery Observer Programme for Hector's dolphins in Pegasus Bay, Canterbury Bight, 1997/1998. Published client report on contract 3020, funded by Conservation Services Levy. Department of Conservation, Wellington. 28 p.

Stockin, K A; Law, R; Roe, W; Meynier, L; Martinez, E; Duignan, P; Bridgen, P; Jones, B (2010) PCBs and organochlorine pesticides in Hector's and Maui's dolphins. *Marine Pollution Bulletin 60*: 834–842.

Stone, G; Hutt, A; Duignan, P; Teilmann, J; Cooper, R; Geschke, K; Yoshinaga, A; Russell, K; Baker, A; Suisted, R; Baker, S; Brown, J; Jones, G; Higgins, D (2005) Hector's Dolphin (*Cephalorhynchus hectori hectori*) Satellite Tagging, Health and Genetic Assessment. Final Report. New England Aquarium, Boston, USA.

Stone, G; Kraus, S; Hutt, A; Martin, S; Yoshinaga, A; Joy, L (1997) Reducing by-catch: can acoustic pingers keep Hector's dolphins out of fishing nets. *Marine Technological Society Journal 31*: 3–7.

Stone, G S; Yoshinaga, A (2000) Hector's dolphin (*Cephalorhynchus hectori*) calf mortalities may indicate new risks from boat traffic and habituation. *Pacific Conservation Biology 6*: 162–170.

Suisted, R; Neale, D (2004) Department of Conservation Marine Mammal Action Plan for 2005–2010. Department of Conservation, Wellington. 89 p.

Taylor, P R (1992) Incidental catch of non-fish species in set nets in New Zealand waters. *New Zealand Fisheries Assessment Research document 92/21*. (Unpublished document held at NIWA library, Wellington.)

Thompson, F N; Richard, Y (2012) Maui's dolphin distribution and fishing effort. *In*: Currey, R J C; Boren, L J; Sharp, B R; Peterson, D A (Eds.) Risk assessment of threats to Maui's dolphins. pp 33–49. Ministry for Primary Industries and Department of Conservation, Wellington. 51 p.

Townsend, A J; de Lange, P J; Duffy, C A J; Miskelly, C M; Molloy, J; Norton, D A (2008) New Zealand Threat Classification System manual. Department of Conservation, Wellington. 35 p.

Trippel, E A; Strong, M B; Terhune, J M; Conway, J D (1999) Mitigation of harbour porpoise (*Phocoena phocoena*) by-catch in the gillnet fishery in the lower Bay of Fundy. *Canadian Journal of Fisheries and Aquatic Sciences 56*: 113–123.

Turek, J; Slooten, E; Dawson, S; Rayment, W; Turek, D (2013) Distribution and abundance of Hector's dolphins off Otago, New Zealand. *New Zealand Journal of Marine and Freshwater Research 47*: 181–191.

Voller, R (1992) Entanglements of Hector's dolphins in set nets between Motunau and Timaru. Ministry of Agriculture and Fisheries. 21 p.

Wade, P R (1998) Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Marine Mammal Science 14*: 1–37.

Wade, P R; Hamner, R M; Constantine, R; Baker, C S (2012) The Potential Biological Removal (PBR) and probability of decline for Maui's dolphin. *In*: Currey, R J C; Boren, L J; Sharp, B R; Peterson, D A (Eds.), Risk assessment of threats to Maui's dolphins. pp 28–32. Ministry for Primary Industries and Department of Conservation, Wellington. 51 p.

Webster, T A; Dawson, S M; Slooten, E (2009) Evidence of sex segregation in Hector's dolphin. *Aquatic Mammals 35*: 212–219.

Webster, T A; Dawson, S M; Slooten, E (2010) A simple laser photogrammetry technique for measuring Hector's dolphins in the field. *Marine Mammal Science 26*: 296–308.

7 COMMON DOLPHIN (*DELPHINUS DELPHIS DELPHIS*)

Status of chapter	This chapter has been partially updated for AEBAR 2018. Updated figures and tables have been identified in their captions.
Scope of chapter	This chapter briefly describes: the biology of short-beaked common dolphins (*Delphinus delphis delphis*); the nature and extent of potential interactions with fisheries; management of fisheries interactions; means of estimating fisheries impacts and population level risk; and remaining sources of uncertainty, to guide future work.
Area	All of the New Zealand EEZ and Territorial Sea.
Focal localities	Areas where significant fisheries interactions are known to have occurred include waters off the west coast of the North Island (including Taranaki Bight) and to a lesser extent Cook Strait.
Key issues	Improved means of estimating incidental captures and risk in poorly observed inshore fisheries; improved understanding of population size and structure; improved understanding of common dolphin spatio-temporal distributions affecting interaction rates with fishing effort.
Emerging issues	Improved ability to assess risk and apply risk management solutions on a regional subpopulation basis, or at finer spatial and temporal scales
MPI research (current)	PRO2013-01 *Estimation of Seabird and Marine Mammal Captures*; PRO2014-01 *Improving information on the distribution of seabirds and marine mammals*; PRO2017-08A *Research into the demographic parameters for at-risk marine mammals as identified by the risk assessment (common dolphins).*
NZ government research (current)	DOC Marine Conservation Services Programme (CSP): INT2015-01 *To understand the nature and extent of protected species interactions with New Zealand commercial fishing activities*; INT2015-03 *To determine which marine mammal, turtle and protected fish species are captured in fisheries and their mode of capture*.
Other research	Massey University: Skull morphometrics, growth and reproductive biology, diet and nutritional ecology, fine-scale distribution and abundance, and mother-offspring dynamics of common dolphins in New Zealand. Auckland University: Impacts of tourism on dolphin behaviour examining and the effectiveness of permit changes to the dolphins' responses to swimmers and boats.
Related chapters/issues	Chapter 3: Spatially Explicit Fisheries Risk Assessment (SEFRA); See also the JMA chapter, page 557, of the Fisheries Assessment Plenary Volume 2 (MPI 2017)

7.1 CONTEXT

Short-beaked common dolphins (*Delphinus delphis delphis*) were first described by Linnaeus in 1758 and have a worldwide distribution. In New Zealand waters, this species is protected under the Marine Mammal Protection Act (MMPA) of 1978 and the Fisheries Act (FA) of 1996. All marine mammals are protected under the s.2 (1) of the FA. The ministers for the Department of Conservation (DOC) and the Ministry for Primary Industries (MPI) can jointly approve a population management plan (PMP) for one or more species under s.14F of the Wildlife Act or s.3E of the MMPA. This PMP can include a maximum allowable level of fishing-related mortality of the species in New Zealand waters and recommendations to the Minister of Fisheries on 1) measures to mitigate fishing-related mortality and 2) the standard of information to be collected on fishing-related mortality. Currently, a PMP does not exist for common dolphins.

MPI manages fishing-related mortalities of common dolphins under s.15 (2) of the FA '*to avoid, remedy, or mitigate the effect of fishing-related mortality of any protected species and such measures may include setting a limit on fishing-related mortality.*' The 2005 Conservation General Policy administered by DOC specifies that *'protected marine species should be managed for their long-term viability and recovery throughout their natural range'*. The management of fisheries interactions with common dolphins aligns with the 2030 objective 6 to *'manage impacts of fishing and aquaculture'* and Strategic Action 6.2

to *'set and monitor environmental standards, including for threatened and protected species and seabed impacts'*.

Under the National Deepwater Plan, Objective 2.5 is most relevant to the management of common dolphins in New Zealand waters: *'manage deepwater and middle-depth fisheries to avoid or minimise adverse effects on the long-term viability of endangered, threatened, and protected species'* (Ministry for Primary Industries 2012). The National Deepwater Plan contains information for fisheries to assess and manage marine mammal interactions with the deepwater fishing activity including a Marine Mammal Operating Procedure (MMOP), which outlines specific mitigation practices and proper handling of incidental marine mammal captures (Ministry for Primary Industries 2012).

Management Objective 7 of the National Fisheries Plan for Highly Migratory Species (HMS) is to 'implement an ecosystem approach to fisheries management, taking into account associated and dependent species' (Ministry of Fisheries 2010). The goals under this objective are as follows:

1. Avoid, remedy, or mitigate the adverse effects of fishing on associated and dependent species, including through maintaining food chain relationships.
2. Minimise unwanted bycatch and maximise survival of incidental catches of protected species in HMS fisheries using a risk management approach.
3. Increase the level and quality of information available on the capture of protected species.

The Draft National Fisheries Plan for Inshore Finfish states that the objectives of all groups is *'to minimise the adverse impact of fishing actives on the aquatic environment, including on biological diversity'* (Ministry of Fisheries 2011).

7.2 BIOLOGY

7.2.1 TAXONOMY

Within the Delphindae family, common dolphin are a member of the subfamily Delphininae (Perrin 1989). Based on genetic and morphological differences, there are two currently recognised species of common dolphins, the short-beaked (*Delphinus delphis*) and the long-beaked (*D. capensis*) (Rosel et al. 1994, Heyning & Perrin 1994). There are two subspecies of the short-beaked common dolphin (*D. d. Delphis* and *D. d. ponticus*), which is found only in the black sea) and two subspecies of long-beaked common dolphin (*D. c. capensis* and a nominal subspecies recognized as *D. c. tropicalis*; Jefferson & Waerebeek 2002). Genetic and morphometric differences between common dolphin populations in the South Pacific and those from other parts of the world have cast uncertainty as to the taxonomic identity of the New Zealand population of common dolphins (Bell et al. 2002, Stockin 2008, Stockin & Visser 2005). Skull morphometry values from Australia and New Zealand common dolphins fall between those reported for short- and long-beaked common dolphins. However, initial evidence suggests that the species in New Zealand waters is a larger form of the short-beaked common dolphin found elsewhere (Jordan et al. 2015, Jordan 2012, Bell et al. 2002). For the remainder of this chapter, 'common dolphin' will refer to the short-beaked species – *D. d. delphis*.

7.2.2 DISTRIBUTION

Common dolphins are found worldwide in tropical, subtropical, and temperate waters of the Pacific and Atlantic oceans (Hammond et al. 2008, Evans 1994) (Figure 7.1). This species also occurs in confined seas such as the Sea of Okhotsk and Sea of Japan as well as in small subpopulations in places such as the Mediterranean and Black Seas (Hammond et al. 2008). New Zealand waters represent the southern-most limit of common dolphins. Common dolphins are found around both the North and South Island (Brager & Schnieder 1998, Gaskin 1968, Berkenbusch et al. 2013, Constantine & Baker 1997) (Figures 1.2 and 1.3). However, Gaskin (1968) suggests that the distribution of common dolphins in New Zealand waters is constrained to warmer waters (greater than ca. 14°C) and is limited by the subtropical East Cape Current in the north and the subtropical convergence in the south.

Common dolphins are frequently observed along the northern and eastern coast of the North Island in the Bay of Islands, Hauraki Gulf, Mercury Bay, and in small groups, outside Wellington Harbour (Gaskin 1968, Constantine & Baker 1997, Neumann & Orams 2005, O'Callaghan & Baker 2002). Similar to other populations, common dolphins in New Zealand waters exhibit inshore and offshore daily and seasonal movements (Meynier et al. 2008, Neumann 2001c, Stockin 2008). The seasonal distribution of common dolphins is largely determined by the behaviour of their prey. Common dolphins are known to forage on small schooling fish that are strongly linked to sea surface

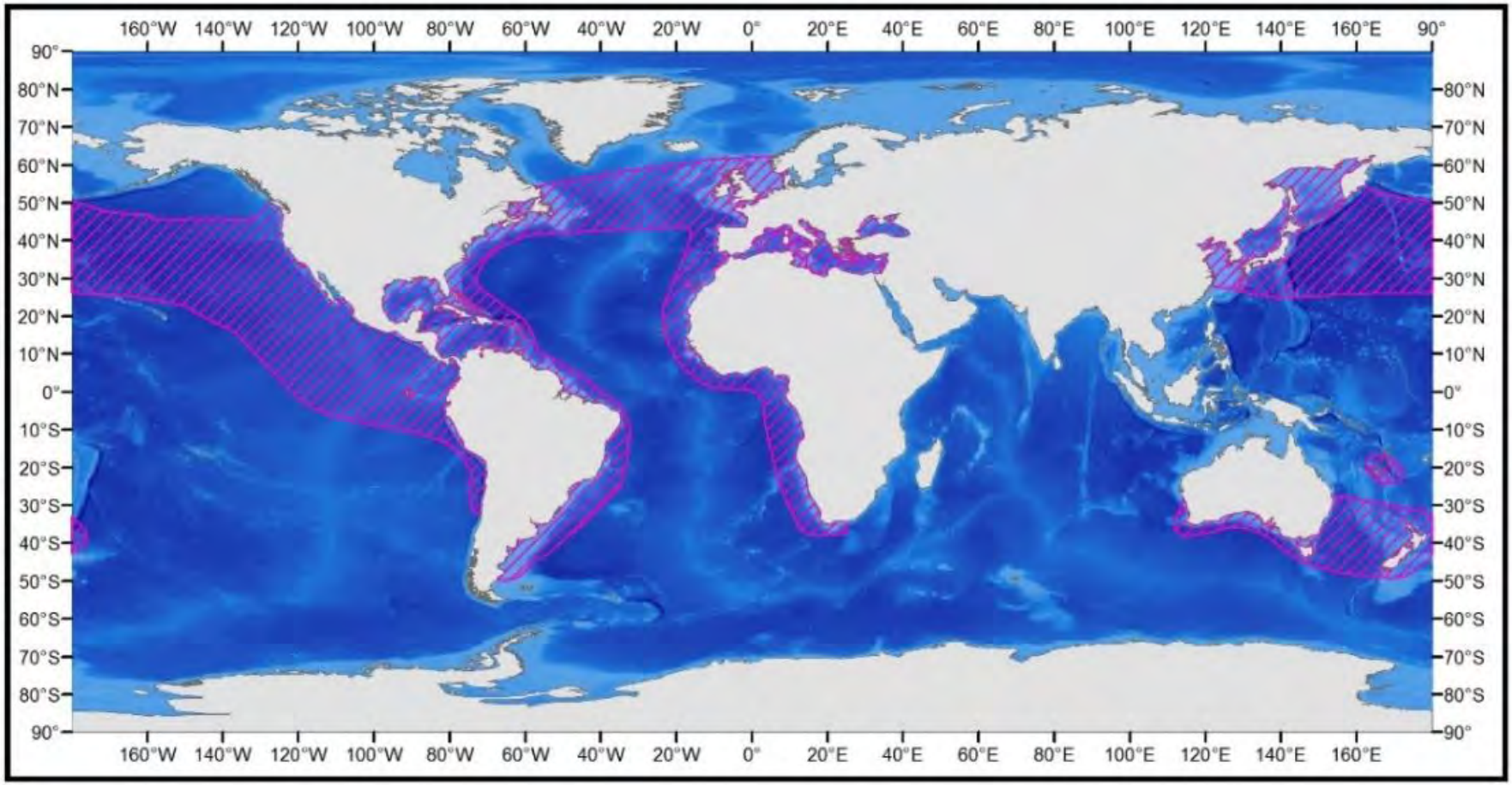


Figure 7.1: Worldwide distribution of short-beaked common dolphins (*Delphinus delphis delphis*) provided by the International Union for the Conservation of Nature (IUCN) (Hammond et al. 2008). Magenta hatched areas indicate range.

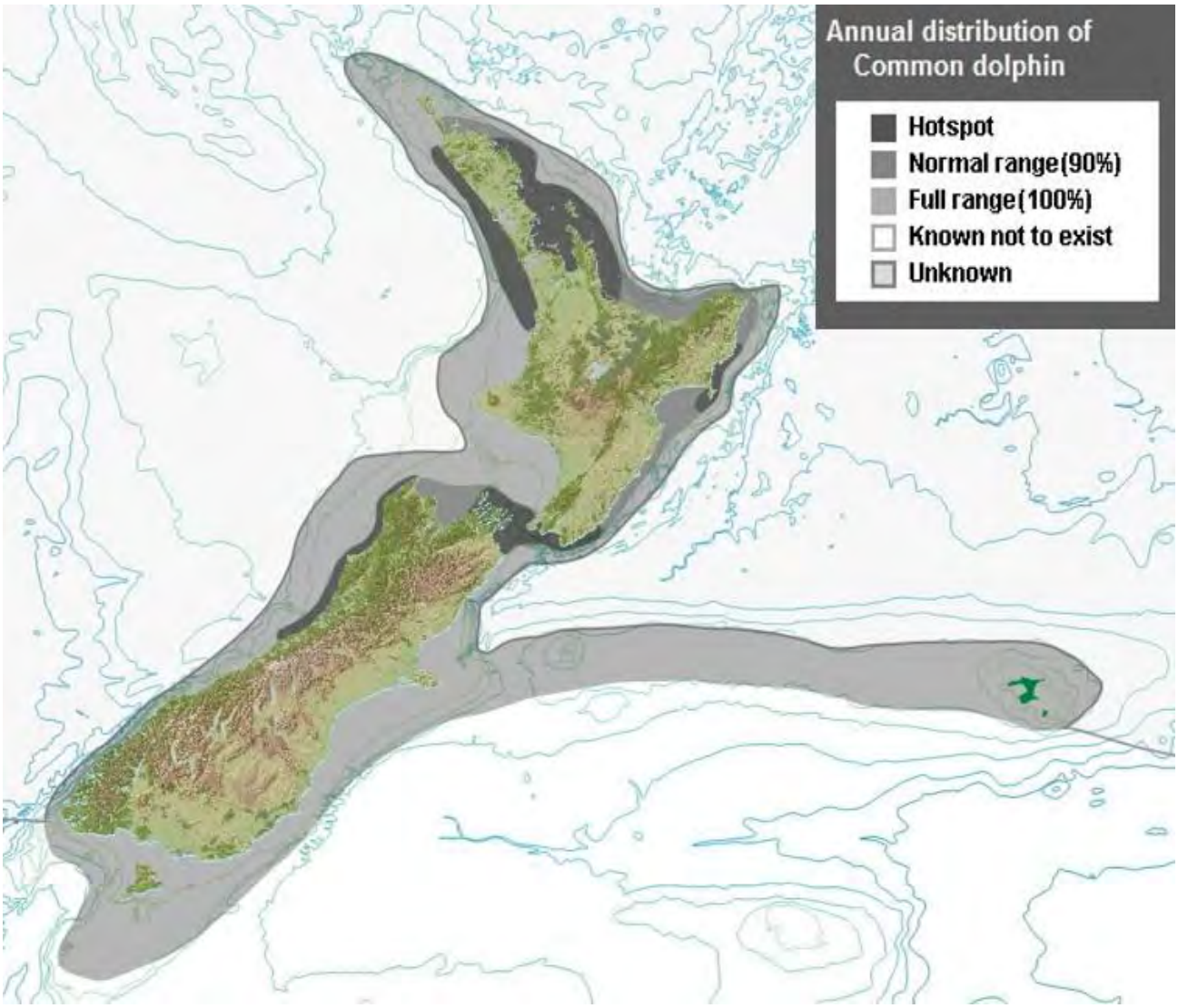


Figure 7.2: Distribution of short-beaked common dolphins (*Delphinus delphis delphis*) in New Zealand waters (from www.nabis.govt.nz).

temperature (SST). As a result, both common dolphins and their prey are found close to shore in the spring and summer when SST is high and further offshore in the autumn when SST drops (Neumann 2001, Stockin 2008, Neumann 2001). This species is also known to adjust their seasonal movements to take advantage of warmer water during a La Niña event (Neumann 2001).

Common dolphins are encountered in single and large multi-species groups with both seabirds and other marine mammals (hundreds to thousands) and found in waters both nearshore and thousands of kilometres offshore, in pelagic waters (Evans 1994). In New Zealand waters, they are known to form large aggregations with approximately 10 seabird and seven cetacean species. Of the seabird species, common dolphins are most often associated with the Australasian gannet (*Morus serrator*). Associations with other cetaceans include: bottlenose dolphins (*Tursiops truncatus*), striped dolphins (*Stenella coeruleoaba*), Hector's dolphin (*Cephalorhynchus hectori hectori*), Dusky dolphin (*Lagenorhynchus obscurus*), Minke whale (*Balaenoptera acutorostrata*), Sei whale (*Balaenoptera borealis*), and Bryde's whales (*Balaenoptera brydei*) (Stockin 2009).

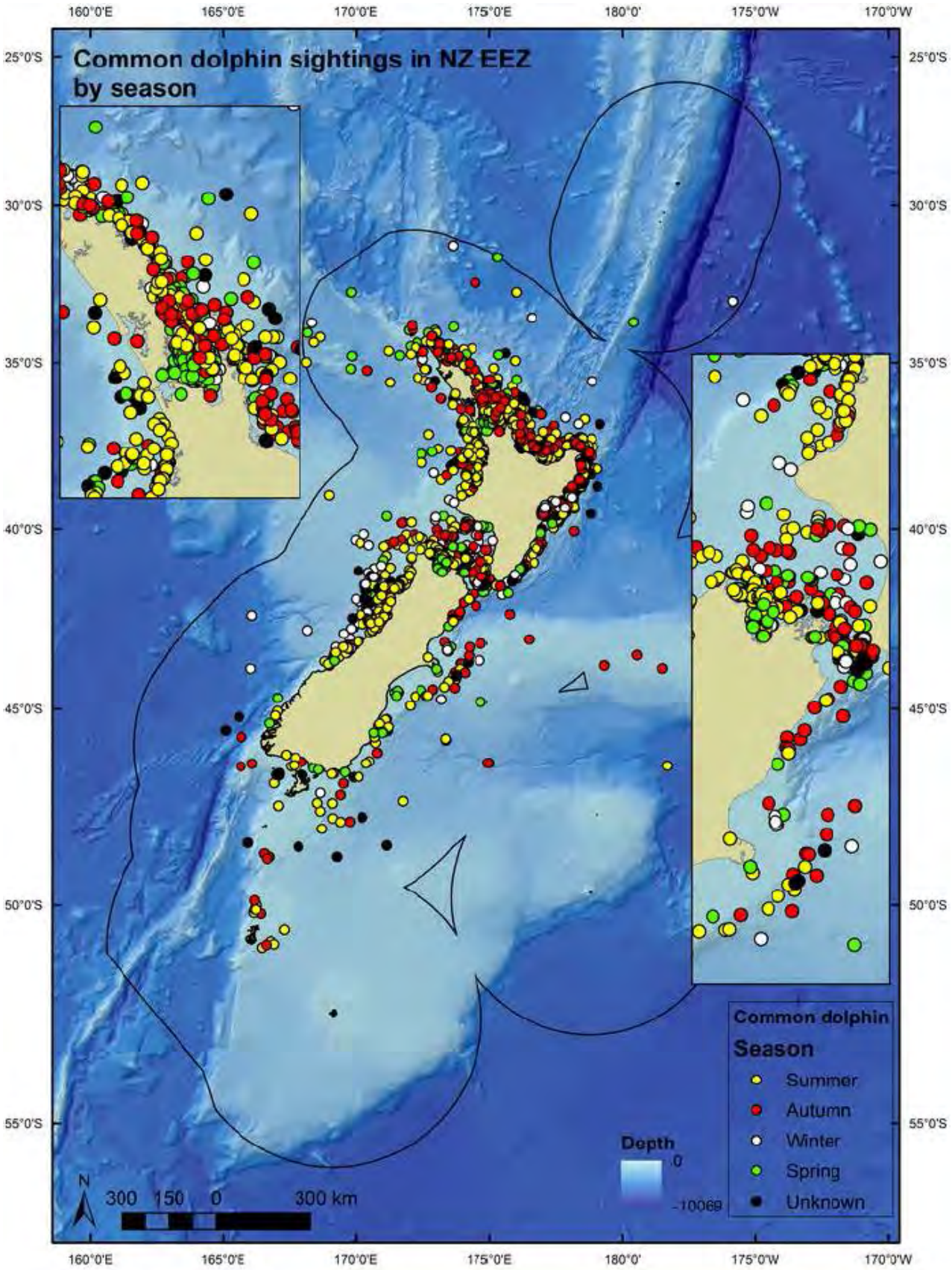


Figure 7.3: Systematic and opportunistic sightings of short-beaked common dolphins (*Delphinus delphis delphis*) in New Zealand waters between 1970 and 2013. Data sources include Department of Conservation (DOC), Cawthorn (2009), opportunistic at-sea sightings (NIWA), and the Centralised Observer Database (COD). (Sightings are indicative of the distribution only). Figure from Berkenbusch et al. (2013).

7.2.3 FORAGING ECOLOGY

The diet of common dolphins has primarily been assessed from the stomach contents of stranded and incidentally captured animals. Studies on common dolphins worldwide have documented the primary prey items as small schooling epipelagic and mesopelagic fish such as mackerel, sardines, and anchovies, as well as squid (Hammond et al. 2008, Young & Cockcroft 1994, Silva 1999, Bearzi et al. 2003, Pusineri et al. 2007, Overholtz 1991, Morizur et al. 1999). While there is abundant information on the diet of common dolphins for many populations, there is relatively little information for common dolphins in New Zealand waters.

Although research has specifically identified the Hauraki Gulf as an area extensively used for feeding, common dolphins forage in waters all around New Zealand (Stockin et al. 2009a). In one study, common dolphins off the east coast of the North Island were observed foraging on schools of jack mackerel (*Trachurus novaezelandiae*), schools of juvenile kahawai (*Arripis trutta),* yellow-eyed mullet (*Aldrichetta forsteri*), flying fish (*Cypselurus lineatus*), and, on one occasion, a school of parore (*Girella tricuspidata*), and garfish (*Hyporamphus ihi*) (Neumann & Orams 2003). The prevalent prey species from the stomach contents of animals stranded around the New Zealand coastline (n=27) and animals incidentally captured in the jack mackerel fishery off the west coast of the North Island (n=10) included arrow squid (*Nototodarus* sp.), anchovy (*Engraulis australis*), jack mackerel (*Trachurus* spp.) (Meynier et al. 2008). In another study, pilchard (*Sardinops neopilchardus*), and garfish (*Hyporhamphus ihi*) were the predominant prey items found in the stomachs of nine New Zealand common dolphin carcasses (n=9) classified as 'entanglement' (Stockin et al. 2009b).

The similarity in prey items found in the stomachs of coastal and offshore animals provides further support that common dolphins in New Zealand make daily excursions between nearshore and offshore environments (Meynier et al. 2008). In addition, many of the prey species (e.g., squid) found in the stomachs of common dolphins are found in the deep scattering layer, which migrates towards the surface at night (Hammond et al. 2008, Neumann & Orams 2003). Neumann & Orams (2003) cite personal communication with S. Morrison in which common dolphin were sighted by crew members of squid boats during nocturnal fishing in Mercury Bay suggesting that time of day may provide important foraging opportunities for this species. The ability of common dolphins to feed on small schooling fish in shallow coastal waters during the day and on prey in the deep scattering layer in pelagic waters at night may indicate foraging plasticity (Neumann 2001). Acoustic research in New Zealand waters showed that during the day the mesopelagic layer occupied waters deeper than 200 m, then rapidly ascended to close to the surface after sunset; throughout the night, this layer dispersed downwards but remained in depths of less than 200 m until dawn when the it descended to day depths (McClatchie & Dunford 2003, O'Driscoll et al. 2009). O'Driscoll et al. (2013) found that schools of jack mackerel ascended and dispersed at night and were seen in depths of 10–30 m before dawn.

To exploit a large range of prey species, common dolphins exhibit a variety of foraging strategies. In New Zealand waters, both individual and coordinated feeding strategies have been documented (Neumann & Orams 2003). Individual foraging strategies include four types of behaviour: high-speed pursuit (traveling at high velocity in a zig-zag erratic fashion), fish-whacking (fish whacked with tail-fluke) and kerplunking (rapid tail-fluke movement in shallow water) (Neumann & Orams 2003, Constantine & Baker 1997). Furthermore, coordinated foraging strategies include: wall formation (driving fish into shallower water), carouseling (herding fish against the water surface), and bubble-blowing (startling herded fish).

Common dolphins are often observed foraging in association with other species (Neumann & Orams 2003). Rather than initiating feeding as a multi-species group, research indicates that birds and cetaceans may alert one another to prey by their presence and behaviour (Neumann & Orams 2003).

7.2.4 REPRODUCTIVE BIOLOGY

Despite their global distribution, relatively little information exists on the reproductive biology of common dolphins. Most of the existing information comes from studies on common dolphin populations in the North Pacific, Eastern Tropical Pacific, or North Atlantic and may or may not be applicable to the population of animals in New Zealand waters. Male and

female age of sexual maturity for common dolphins in the North Pacific is 10.5 years for males and 8.0 years for females with lengths ranging 179–182 cm and 170.7–172.8 cm, respectively (Ferrero & Walker 1995). In the North Atlantic population, males reach sexual maturity at 9.5 years and 213 cm and females at 8.3 years and 200 cm (Westgate & Read 2007). A later age of sexual maturity for males may be the result of delayed breeding until the testes are large enough to compete with other males (Westgate & Read 2007).

Testes weight of sexually mature males ranges from 273.2 to 1190 g (Ferrero & Walker 1995). Male common dolphins in the North Atlantic exhibit seasonal changes in testes size with largest testes occurring in mid-July and smallest in October (Westgate & Read 2007). The peak in testes size corresponds with the timing of ovulation, conception and parturition and changes five-fold between maximum expansion and retraction. In mature males, testes comprised 2.2–3% of their total body mass (Westgate & Read 2007). Results from Westgate & Read (2007) suggest common dolphins in the North Atlantic engage in sperm competition as evidences by the seasonal change in testes size. The slight sexual dimorphism between sexes, in addition to seasonal changes in testes size, indicate that males compete for access to oestrous females and that females likely mate with many males (Westgate & Read 2007). Given that many common dolphins in temperate environments exhibit reproductive seasonality, it is likely that the New Zealand population of animals also exhibits a peak in reproduction that may correspond to seasonally abundant prey or optimal water temperatures.

Although gestation time for common dolphins in New Zealand waters is unknown, the length of gestation for this species is about 11 months for the North Pacific population, 11–12 months for the North Atlantic population, and 11 months for the Black Sea population (Westgate & Read 2007, Ferrero & Walker 1995, Gaskin 1972). Like all odontecetes, common dolphins give birth to a single calf, though one occurrence of a twin birth was reported off the coast of Spain (Gonzalez et al. 1999). At parturition, Westgate & Read (2007) estimated the length of neonate common dolphins in the North Atlantic at 93.2 cm. Neonates nurse for approximately six months and begin foraging at three to six months of age (Brophy et al. 2009). Common dolphins in the North Atlantic were found to have a minimum inter-birth interval of two years (Westgate & Read 2007).

In New Zealand waters, calves are seen year-round in the Hauraki Gulf, however, peak numbers are recorded in late spring and early summer months of December and January (Stockin et al. 2008). Common dolphins are considered a social species, showing non-random associations with other individuals. Sexual segregation in which animals divide into 'bachelor' (adult males), and 'nursery' (adult females and calves) groups has been observed in common dolphins in New Zealand waters (Neumann 2001, Neumann et al. 2002, Viricel et al. 2008). Mixed-sex groups also occur though they are usually associated with mating activities. The lack of stability in group composition is known as a 'fission-fusion' society in which group composition changes almost daily (Connor et al. 2000, Neumann 2001).

7.2.5 POPULATION BIOLOGY

The abundance of common dolphins is estimated at 4 000 000 worldwide with population estimates existing for many regions: 370 000 in the western US; 3 000 000 in the Eastern Tropical Pacific; 30 000 off the eastern US; 96 000 in the Black Sea; 60 000 on the eastern Atlantic continental shelf; 14 700 in the Alboran Sea; 75 000 in the Celtic Sea Shelf; and 19 400 in the western Mediterranean Sea (Jefferson et al. 2011, Hammond et al. 2008).

Although there is currently no abundance estimate for common dolphins in New Zealand waters, they are considered the most abundant and widespread cetacean recorded in the Hauraki Gulf, an important foraging and nursery area, in the summer (O'Callaghan & Baker 2002). Unlike common dolphins in other areas of New Zealand waters, in the Hauraki Gulf, this species exhibits high site fidelity (Stockin et al. 2008, 2014).

The maximum age of short-beaked common dolphins in western North Atlantic teeth was estimated at over 30 years using teeth samples from 204 bycaught and stranded animals (Westgate & Read 2007). Similarly, growth layers of teeth collected from 206 common dolphins in New Zealand waters that were stranded or bycaught in the midwater trawl fishery for jack mackerel (*T. novaezelandiae*) estimated maximum age at over 20 years and 29 years for males and females, respectively (Stockin et al. 2011, Murphy et al. 2014). Seven common dolphins incidentally caught by New Zealand fisheries and returned for autopsy were aged between 4 and 11

years (based on dentinel growth layers) (Duignan et al. 2003, 2004, Duignan & Jones 2005).

Microsatellite analyses of nearshore and offshore New Zealand common dolphins suggest that these animals have recently diverged (Stockin et al. 2014). In addition, the presence of high genetic variation at the southern limit of their distribution suggests that the overall population in New Zealand waters may be expanding and that there are fine-scale population level differences (Stockin et al. 2014).

Common dolphin populations are subject to many natural and anthropogenic threats that include but are not limited to: stranding, disease, predation, toxins, habitat loss, vessel-strike, recreational and commercial fishing and tourism-based activities. The cumulative impact of these threats on common dolphin populations has not been assessed. Drivers of common dolphin mortality include seasonal environmental variation, commercial fisheries interactions, habitat degradation, high-intensity acoustic disturbance, and disease (Murphy et al. 2013).

The Mediterranean Sea population of common dolphins was greatly reduced due to five main factors: 1) habitat loss, 2) prey depletion, 3) incidental captures by fisheries, and 4) immuno-suppression caused by chemical contamination, and 5) environmental fluctuations (Bearzi et al. 2003). In addition, at least 840 000 animals were removed from the Black Sea by hunters between 1946 and 1983, after which the population further declined due to disease and overfishing of prey species (Hammond et al. 2008).

In the absence of a population estimate for common dolphins in New Zealand waters, the impact of natural and human-induced effects cannot be accurately determined. Two of the main known threats to common dolphins in New Zealand waters are incidental capture by fisheries and tourism-related impacts (Thompson et al. 2013, Neumann & Orams 2005, Meissner et al. 2015, Constantine & Baker 1997, Stockin 2009). Fisheries-related threats are discussed in detail in the Sections 7.3 and 7.4.

7.2.6 CONSERVATION BIOLOGY AND THREAT CLASSIFICATION

Common dolphins are currently listed as a species of least concern under the International Union for the Conservation of Nature (IUCN) Red List of Threatened species with the exception of the Mediterranean subpopulation, which is listed as 'endangered' (Hammond et al. 2008).

In 2010, the conservation status of New Zealand marine mammals was reassessed using the 2008 version of the New Zealand Threat Classification system (Baker 2010). Based on several levels of criteria, common dolphins were classified as 'not threatened' with the qualifiers that the information was considered 'data poor', that the species was 'secure overseas', and that some subpopulations were 'threatened overseas'.

7.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Interactions between cetaceans and fisheries occur worldwide. Cetaceans have been incidentally captured by numerous types of fishing gear including trawl nets, purse seine nets, and static nets such as driftnets or gillnets (Reeves et al. 2005). Hall et al. (2000) state that cetaceans are at greater risk of capture by midwater trawls, which are towed faster than bottom trawls and usually target fish and squid. As a result, cetaceans may be captured when foraging in areas where fisheries using such gear also operate.

Due to their high global abundance, interactions between common dolphins and fisheries are not unusual. The highest rates of interactions are associated with fisheries that use trawl, purse seine, and drift nets. Outside New Zealand, perhaps the most well-known interaction occurs in the Eastern Tropical Pacific where common dolphins are found in association with yellowfin tuna (*Thunnus albacares*). In the 1960s, about 350 000 common dolphins were estimated to have been taken by this purse seine fishery (Joseph 1994). However, due to mitigation measures introduced in the 1970s, the rate of dolphin captures has been greatly reduced and is no longer a conservation concern (Reeves 2003).

In addition to the Eastern Tropical Pacific, interactions between common dolphins and fisheries

are known to occur in the north and south Atlantic and Pacific oceans. Common dolphins were the most commonly caught cetacean in the US shark and swordfish gillnet fishery with an estimated mortality of 861 dolphins between 1996 and 2002 (Carretta et al. 2005). In the UK and the French pelagic trawl fishery for bass, ca. 800 common dolphins were taken annually (Hammond et al. 2008). In addition, the pelagic pair-trawl fishery off southwest England captured approximately 200 common dolphins per annum, with most animals being captured at night (de Boer et al. 2012). Male dolphins were at a greater risk of capture in pair-trawls offshore whereas females and calves were more vulnerable to gillnets close to shore (de Boer et al. 2012). Other areas where interactions between common dolphins and fisheries are known to occur include:

- The North Sea, predominantly in gillnets (Reijnders & Lankester 1990).
- Off the coast of Africa, predominantly in gillnet and purse seine fisheries (Maigret 1994, Jefferson et al. 1997).
- Off the south coast of Australia, mostly in gillnets or anti-shark netting (Kemper et al. 2005).
- Off the coast of Portugal, where 59% of 124 bycaught common dolphins were bycaught in primarily gill and seine nets between 1975 and 1998 and where fisheries interactions were responsible for up to 44% of strandings (Silva & Sequeira 2003).
- The Mediterranean Sea, where dolphins have a moderate (6–30% of sightings) or strong (35–50% of sightings) association with foreign purse seine tuna fishing, dolphin fish fishing activities, and illegal drift nets for swordfish offshore (Vella 2005, Tudela et al. 2005, Bearzi et al. 2008).
- The Black Sea, in pelagic trawl nets (Hammond et al. 2008, Reeves & Notarbartolo di Sciara 2006).

The Mediterranean Sea subpopulation of common dolphins has been declining since the 1960s and has been subjected to the effects of illegal drift-netting and other anthropogenic impacts (Reeves 2003, Forcada & Hammond 1998, Piroddi et al. 2011). It is believed that overfishing in the Mediterranean Sea has outcompeted common dolphins for prey (Bearzi et al. 2003). Bearzi et al. (2008) found that 10 active purse seine vessels were responsible for removing 33% of the biomass and suggested that they had the largest impact on dolphin prey species.

To reduce mortality from incidental captures, many countries have put implemented monitoring programmes to mitigate direct fisheries impacts to common dolphins. For example, after the creation of the US Marine Mammal Protection Act in 1972, observer coverage in the purse seine fishery was increased to 100% to ensure compliance. The European Union has also introduced legislation to establish observer programmes for most fisheries (Hammond et al. 2008). Other measures to reduce unwanted bycatch include: modification of fishing gear and methods (acoustic deterrents), input and output controls (limiting fishing effort or capacity), compensatory mitigation (investing in conservation projects), establishment of Marine Protected Areas (MPAs), fleet communication (reporting real-time observation of unpredictable bycatch hotspots), industry self-policing (peer pressure from within the industry), handling and release practices (backing down and hand rescue procedures to release dolphins), and changing gear (using alternative fishing methods that results in lower bycatch) (Gilman & Lundin 2009).

7.4 STATE OF KNOWLEDGE IN NEW ZEALAND

Common dolphins and fisheries in New Zealand waters often target the same fish species in the same areas. Early reports to the International Whaling Commission suggested that during June 1979 and April 1992, common dolphins were captured in trawl nets, crayfish pots, and purse seine nets (see Berkenbusch et al. 2013). Scientific observer data show that the primary fishery in New Zealand waters that is responsible for common dolphin mortality has been the midwater trawl fishery for jack mackerel species. Evidence from the early 1990s, after the establishment of the government observer programme, indicated that single and multiple captures of common dolphins occurred in the trawl nets of foreign-chartered trawlers targeting jack mackerel species off the west coast of New Zealand, in Quota Management Areas 7, 8 and 9 (61 animals between 1989–90 and 1992–93; see Baird 1994). This fishery operated offshore in the north and south Taranaki Bight waters, mainly in the summer months of November to

April. During these years, observers reported a change in this fleet from the use of bottom trawls with headline heights of 5.2–9.8 m to midwater trawls with headline heights of 20–45 m (MPI unpublished observer data). The midwater trawls could be towed near the bottom during the day and in the water column at night and thus follow the movement of the jack mackerel schools. Alternatively, both gear types were used, alternating according to time of day.

Midwater nets were towed for 4–6 hrs and nets hauled between 2330 and 0615 h were responsible for almost all the dolphin captures, particularly in south Taranaki Bight in 70–130 m depths (Baird 1994). These mortalities resulted in the development of voluntary Codes of Practice (COPs) by the company operating the vessels, which aims to outline best practices to remedy, mitigate, or avoid incidental captures (Rowe 2007) (see Baird 1994, Appendix 9). The COPs addressed several aspects of the fishing operation thought to increase the likelihood of capture, mainly: the practice of undertaking a U-turn with the trawl doors up but the net in the water near the surface; the timing of setting; and the vessel lighting during night fishing activities. In addition, the codes may include recommendation for gear modifications and voluntary area closures (Rowe 2007). The government response led to increased observer coverage and provision for the necropsy of captured animals. MPI observer data shows that 10 common dolphins have been autopsied since 1994 (see also Duignan et al. 2003, 2004, Duignan & Jones 2005). However, capture incidents continued to occur until this fleet of vessels ceased fishing in New Zealand waters in the mid–late 1990s (Baird 1996).

Subsequently, midwater trawling for jack mackerels has remained the main method and target fishery responsible for common dolphin captures (based on observer data) (see Abraham & Thompson 2011). However, since the late 1990s, the observed common dolphin captures have been almost entirely from a different fleet of large foreign-charted trawlers operating mainly off the west coast of the North Island during summer months (Thompson et al. 2013a).

These vessels use midwater nets with headline heights of 30–60 m in depths of less than 200 m. The largest capture event in this fishery caught nine dolphins in one tow (Thompson et al. 2013a). Observer coverage between 1995–96 and 2010–11 was at least 20% for most fishing years but fluctuated considerably between 7 and 70% (Thompson et al. 2013a). The vessels are required to follow Operational Procedures for mitigating incidental captures of marine mammals as agreed by quota owners (see Section 7.4.2 for a fuller explanation).

Headline depth of trawl nets (distance from the headline to the surface) was found to be an important factor in explaining common dolphin captures in this fishery (Thompson et al. 2013). The majority of dolphin captures occurred when headline depth was between 10 and 40 m; however, 50% of observed capture events and 54% of common dolphins captured in large vessel mackerel fishery occurred on the 10% of the observed trawls that had a headline depth shallower than 30 m (Figure 7.4) (Thompson et al. 2010). Thompson et al. (2013, 2010) estimated that an increase of 21 m in headline depth may reduce the number of common dolphin captures by half. Longer tows caught more dolphins, as did tows in darkness, and tows conducted in the waters off the north Taranaki Bight. Of all shallow trawl tows (headline depths shallower than 40 m), 69% occurred at night when the fish migrate to the surface (Thompson et al. 2013). Common dolphins are known to follow diel migrating prey, which likely explains higher captures rates in shallow waters at night. Table 7.1 shows common dolphin captures in the jack mackerel fishery from 1989–90 to 1994–95. Most common dolphin captures occurred when conducting midwater trawls at night. The number of captures between 1995–96 and 2001–02 fishing years ranged between zero and 31 animals (Thompson et al. 2013).

Captures have also been reported occasionally from observed trawl fisheries that targeted other middle depth species such as barracouta, hoki and arrow squid, as well as trawl nets targeting inshore species such as trevally and tarakihi (MPI unpublished data). The distributions of the fishing effort and observed captures for 2002–03 to 2015–16 are shown for all trawl fisheries (Figure 7.5) and for jack mackerel fisheries (Figure 7.6). During this time period there were 150 observed captures of common dolphins in trawl fisheries, 134 of which occurred in the jack mackerel fishery (see Section 7.4.1, Tables 7.2 and Table 7.3).

There were no observed common dolphin captures by the following New Zealand fisheries between 2002–03 and 2015–16: trawl (all except jack mackerel, hoki, middle depth and inshore); surface longline (southern

bluefin, albacore and swordfish); bottom longline (ling, snapper); set net (flatfish and mullet); and purse seine (mackerel and skipjack tuna). There was a single common dolphin observed caught in the bigeye surface-longline fishery, in 2014–15. It should be noted that the proportion of the commercial effort covered by observers is highest in deepwater trawl fisheries, with relatively small amounts of effort observed for inshore trawl fisheries and fisheries using other types of fishing gear (see Abraham & Thompson 2011). Between 1995–96 and 2011–12 fishing years, observer effort in the middle-depth, inshore, and flatfish trawl fisheries was 3.4%, 0.5% and 0.3%, respectively (Berkenbusch et al. 2013).

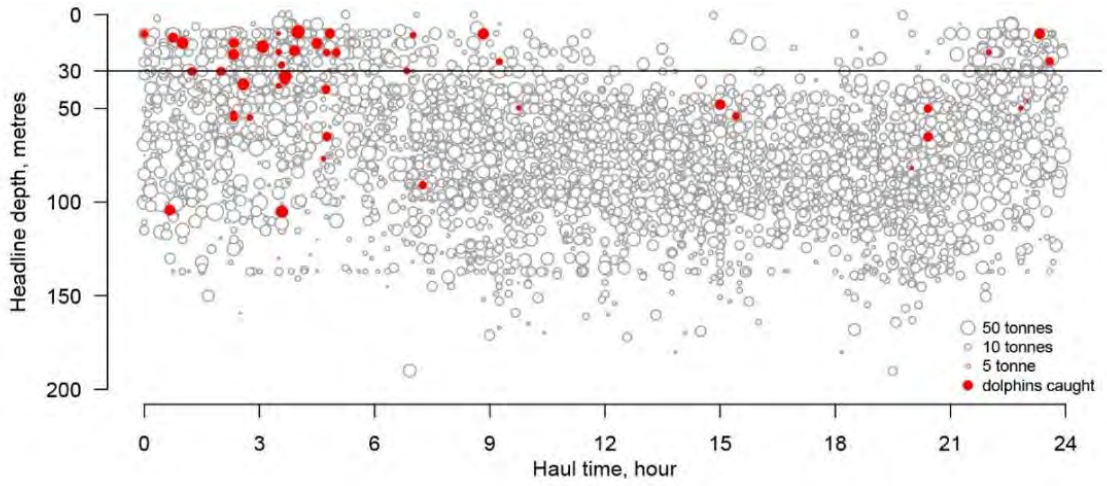


Figure 7.4: Headline depth versus the haul time for observed trawls in the large-vessel jack mackerel fishery. The catch weight is indicated by the size of the circles. Tows where an observed common dolphin capture event occurred are filled (from Thompson et al. 2010).

Table 7.1: Total and observed numbers of tows, observed number of dolphin mortalities and the number of events (tows) that incidentally caught dolphins in the jack mackerel fishery around the North (NT) and South (ST) Taranaki Bights by gear type (MW: midwater and BT: Bottom Tow), and time of day (D: Day and N: Night) for fishing years 1989–90 to 1994–95. Red bold numbers indicate that the species was confirmed as common dolphin (*Delphinus delphis delphis*). Table reproduced from Baird (1994, 1996). [Continued on next page]

Fishing year	Region	Gear	Time of day	Effort		Observed captures	
				Fishing tows	% observed	Mortality	Events
1989–90	NT	BT	D	1191	48	0	0
	NT	MW	D	41	0	0	0
	NT	BT	N	173	6	0	0
	NT	MW	N	28	1	0	0
	ST	BT	D	1418	139	0	0
	ST	MW	D	15	6	0	0
	ST	BT	N	186	6	0	0
	ST	MW	N	105	90	**23**	10
1990–91	NT	BT	D	603	2	0	0
	NT	MW	D	53	0	0	0
	NT	MT	N	72	0	0	0
	NT	MW	N	63	0	0	0
	ST	BT	D	676	47	0	0
	ST	MW	D	147	110	0	0
	ST	BT	N	84	12	0	0
	ST	MW	N	146	73	0	0
1991–92	NT	BT	D	1523	101	0	0
	NT	MW	D	361	4	0	0
	NT	BT	N	279	36	**2**	2
	NT	MW	N	500	3	**5**	3

	ST	BT	D	618	74	1	1
	ST	MW	D	151	3	0	0
	ST	BT	N	95	7	5	1
	ST	MW	N	146	15	16	5
1992–93	NT	BT	D	1759	135	0	0
	NT	MW	D	21	3	0	0
	NT	BT	N	438	22	0	0
	NT	MW	N	156	16	0	0
	ST	BT	D	588	112	0	0
	ST	MW	D	51	0	0	0
1992–93	ST	BT	N	48	6	0	0
	ST	MW	N	305	28	9	3
1993–94	NT	BT	D	1494	78	0	0
	NT	BT	D	219	19	0	0
	NT	MT	N	309	13	0	0
	NT	MW	N	300	28	0	0
	ST	BT	D	645	155	0	0
	ST	MW	D	120	20	0	0
	ST	BT	N	35	14	0	0
	ST	MW	N	279	71	8	5
1994–95	NT	BT	D	391	17	0	0
	NT	MW	D	399	80	0	0
	NT	BT	N	93	9	0	0
	NT	MW	N	258	74	0	0
	ST	BT	D	198	41	0	0
	ST	MW	D	228	73	6	3
	ST	BT	N	27	13	0	0
	ST	MW	N	147	74	15	3

Table 7.2: Fishing and observed effort (number of tows), the number and rate of observed captures, and estimated mean from statistical models of common dolphin (*Delphinus delphis delphis*) captures by all trawl fisheries by fishing year in the New Zealand EEZ (see MPI data analysis at https://data.dragonfly.co.nz/psc, data version 2018001). For each fishing year, the table gives the total number of fishing tows, the percentage of tows that were observed; the number of observed captures (both dead and alive); the capture rate (captures per hundred tows); and the mean number of estimated total captures (with 95% confidence interval). For more information on the methods used to prepare the data, see Thompson et al. 2010 and 2013). [Updated for AEBAR 2018].

Fishing year	Effort		Observed captures		Estimated captures	
	Fishing tows	% observed	Number	Rate	Mean	95% c.i.
2002–03	130 195	5.3	21	0.31	258	135-428
2003–04	120 850	5.4	17	0.26	245	129-413
2004–05	120 468	6.4	22	0.29	218	120-360
2005–06	109 933	6.0	4	0.06	123	52-239
2006–07	103 323	7.7	11	0.14	172	84-310
2007–08	89 533	10.1	20	0.22	141	71-250
2008–09	87 552	11.2	20	0.20	139	68-255
2009–10	92 892	9.7	4	0.04	144	61-274
2010–11	86 085	8.7	9	0.12	165	84-280
2011–12	84 418	11.1	5	0.06	108	42-210
2012–13	83 838	14.8	17	0.14	116	52-218
2013–14	85 110	15.5	30	0.23	118	62-208
2014–15	78 767	17.2	21	0.15	104	50-189
2015–16	78 030	16.6	8	0.06		
2016–17	78 172	17.6	1	0.01		

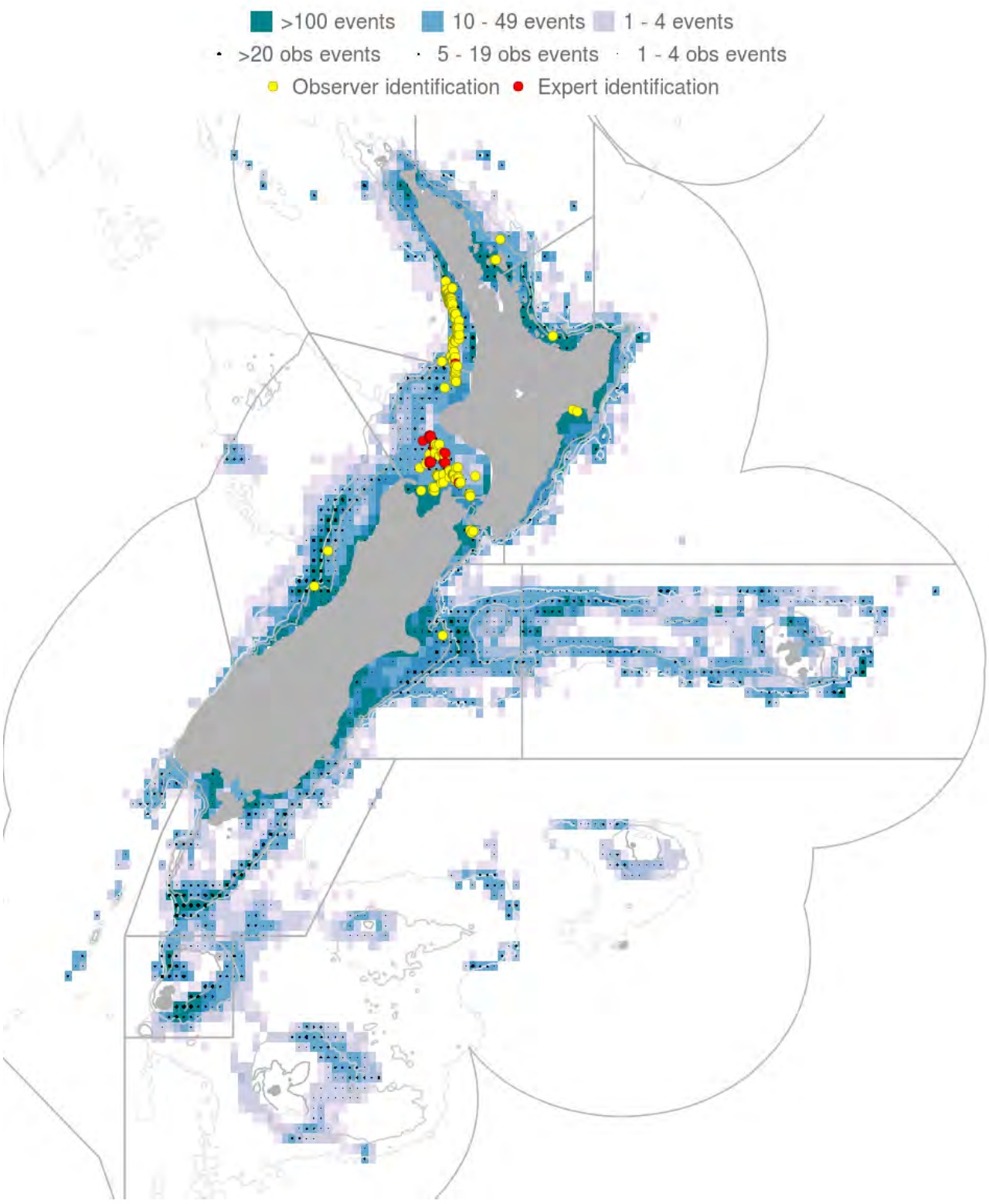


Figure 7.5: Distribution of all trawl fishing effort and observed common dolphin (*Delphinus delphis delphis*) captures, 2002–03 to 2016–17 (for more information see MPI data analysis at https://data.dragonfly.co.nz/psc, data version v2018001). Fishing effort is mapped into 0.2-degree cells, coloured to represent the number of tows. Observed fishing events are indicated by black dots, and observed capture events are indicated by red dots. Fishing effort is shown for all tows with latitude and longitude data, where three or more vessels fished within a cell. [Updated for AEBAR 2018].

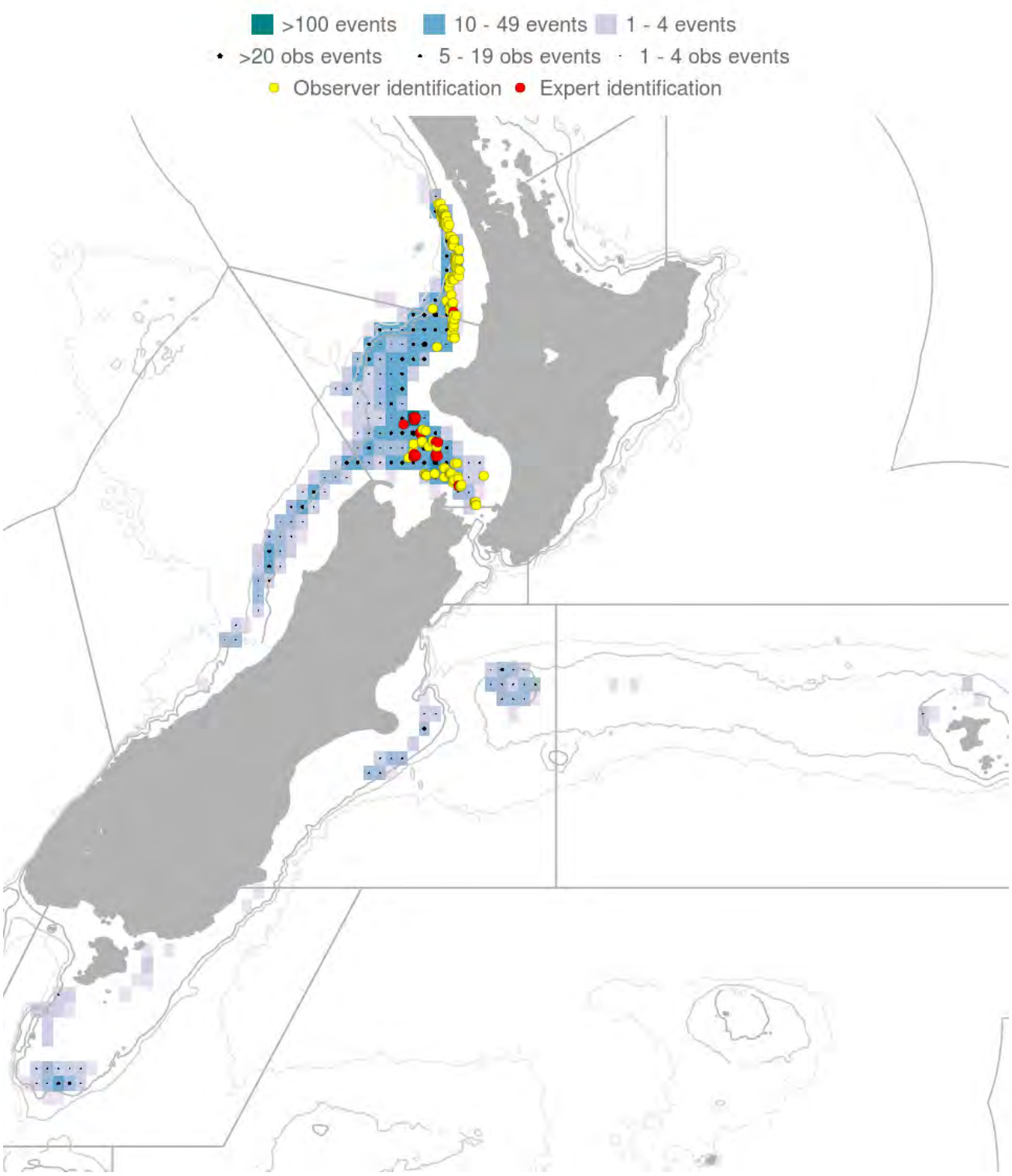


Figure 7.6: Distribution of trawl fishing effort for jack mackerel and observed common dolphin (*Delphinus delphis delphis*) captures, 2002–03 to 2016–17 (for more information see MPI data analysis at https://data.dragonfly.co.nz/psc, data version v2018001). Fishing effort is mapped into 0.2-degree cells, coloured to represent the number of tows. Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. Fishing effort is shown for all tows with latitude and longitude data, where three or more vessels fished within a cell. [Updated for AEBAR 2018].

Table 7.3: Fishing and observed effort (number of tows) and the number, rate, and estimated mean for common dolphin (*Delphinus delphis delphis*) captures by jack mackerel fisheries by fishing year in the New Zealand EEZ (see MPI data analysis at https://data.dragonfly.co.nz/psc, data version 2018001). For each fishing year, the table gives the total number of trawl tows, the number of tows observed and the percentage of tows that were observed; the number of observed captures (both dead and alive); the capture rate (captures per hundred tows); and the mean number of estimated total captures (with 95% confidence interval). For more information on the methods used to prepare the data, see Thompson et al. 2010 and 2013. [Updated for AEBAR 2018].

Fishing year	Effort		Observed captures		Estimated captures	
	Fishing tows	% Observed	Number	Rate	Mean	95% c.i.
2002–03	3 067	11.3	21	6.07	128	54-243
2003–04	2 383	6.4	17	11.18	105	46-196
2004–05	2 510	22.2	21	3.76	82	43-135
2005–06	2 808	25.2	2	0.28	10	2-29
2006–07	2 711	29.6	11	1.37	50	20-94
2007–08	2 653	30.8	20	2.44	41	23-68
2008–09	2 169	37.5	11	1.35	26	13-49
2009–10	2 406	32.7	4	0.51	23	6-55
2010–11	1 881	31.5	7	1.18	63	24-120
2011–12	2 031	76.3	5	0.32	7	5-14
2012–13	2 215	87.6	15	0.77	16	15-20
2013–14	2 453	89.4	28	1.28	30	28-36
2014–15	1 752	86.5	19	1.25	21	19-28
2015–16	1 544	89.5	2	0.14		
2016–17	1 405	72.7	0	0.00		

7.4.1 QUANTIFYING FISHERIES INTERACTIONS

Bayesian models have been applied to fishing effort and observer data collected from trawl fisheries to estimate the number of common dolphin captures within New Zealand's EEZ (Abraham & Thompson 2011) (Figure 7.7). Note that while there were a small number of live captures, most capture events resulted in dolphin mortality. A separate two-step Bayesian hurdle model was developed by Thompson et al. (2010) to estimate the number of captures by the jack mackerel trawl fishery off the west coast of the North Island (Figure 7.8). The first part of the model estimated the presence of a capture event and the second part estimated how many capture events occurred if a capture event was estimated to have been present. Because no captures were recorded from smaller vessels, this analysis only included data from vessels over 90 m in length (Thompson et al. 2010). However, observer coverage of these vessels was limited to 0–0.5% for the years analysed (Thompson et al. 2010). Model-based capture estimates have been created for fishing years since 1995–96 (Thompson et al. 2013) and updated estimates to 2015–16 are presented in Table 7.2.

During the 2002–03 and 2015–16 fishing seasons, less than 3% of the total trawl effort (number of tows) occurred in the jack mackerel fishery, yet 90% of the 206 common dolphin captures recorded by observers occurred in this fishery (Tables 7.2 and 7.3).

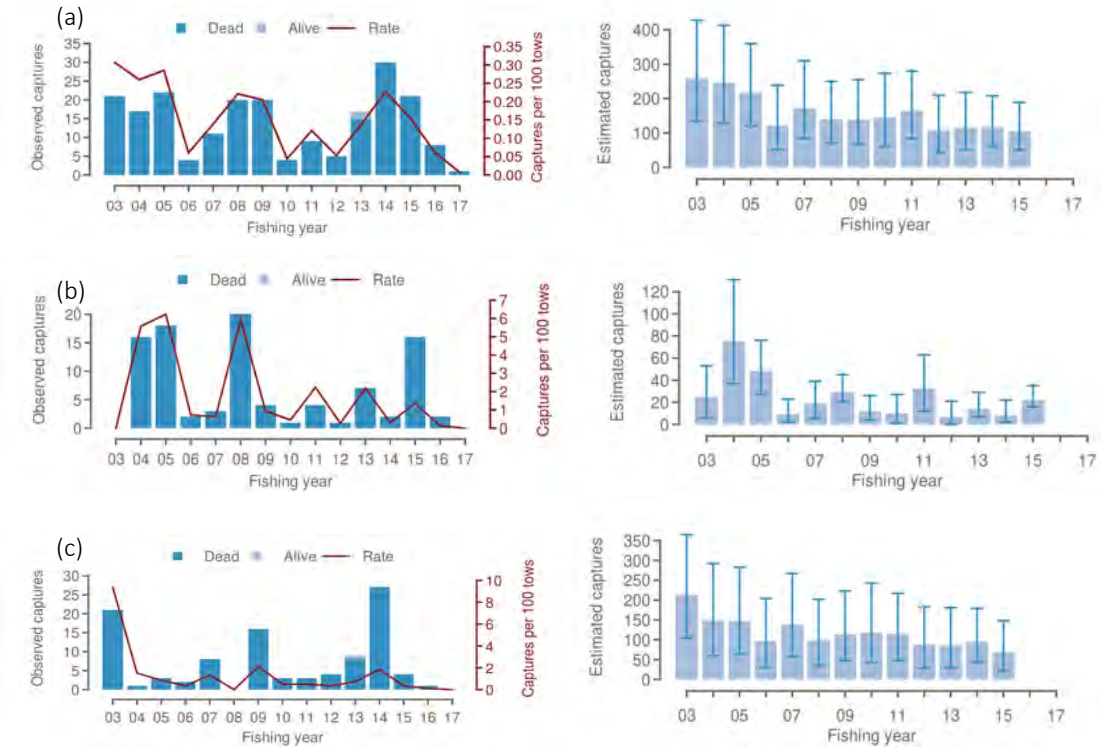


Figure 7.7: Observed captures of common dolphins (*Delphinus delphis delphis*) (dead and alive) in all trawl fisheries, the capture rate (captures per hundred tows) and the mean number of estimated total captures (with 95% confidence interval) by fishing years from 2002–03 to 2016–17, inclusive of three regions: (a) New Zealand's EEZ; (b) West coast of North Island; and (c) the Taranaki region (MPI data analysis at https://data.dragonfly.co.nz/psc, data version v2018001). Percentage effort included in the estimation is shown when it was less than 100%. For more information on the methods used to prepare the data, see Thompson el al. 2010 and 2013. [Updated for AEBAR 2018].

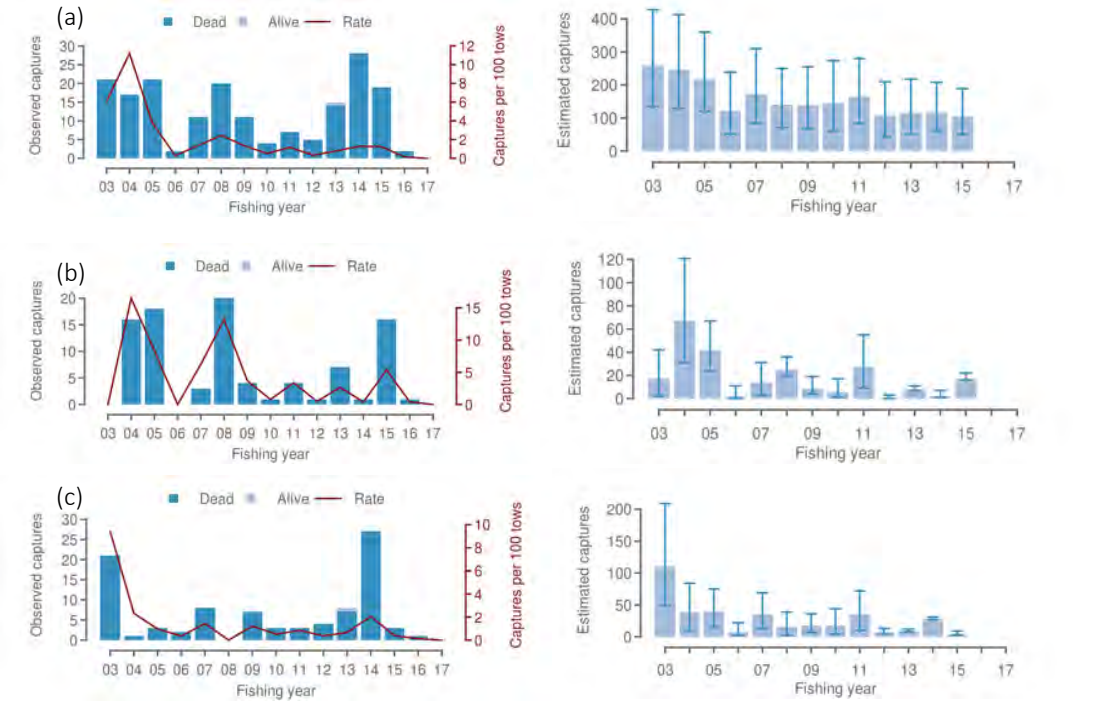


Figure 7.8: Observed captures of common dolphins (*Delphinus delphis delphis*) (dead and alive) in the jack mackerel trawl fisheries, the capture rate (captures per hundred tows) and the mean number of estimated total captures (with 95% confidence interval) by fishing years from 2002–03 to 2016–17 for three regions: (a) New Zealand's EEZ; (b) West coast of North Island; and (c) the Taranaki region (MPI data analysis at https://data.dragonfly.co.nz/psc, data version v2018001). Percentage effort included in the estimation is shown when it was less than 100%. For more information on the methods used to prepare the data, see Thompson el al. 2010 and 2013. [Updated for AEBAR 2018].

7.4.2 MANAGING FISHERIES INTERACTIONS

Because little is known about the population of common dolphins in New Zealand, the level of fisheries impact and population level risk cannot be estimated with certainty. Given the large numbers of common dolphins worldwide, it is unlikely that the interaction between common dolphins and fisheries will have an adverse effect at the scale of the global population. However, there is still debate regarding the taxonomy of common dolphins found in New Zealand waters and whether a unique subpopulation inhabits New Zealand's EEZ. New research is currently underway to investigate population size and structure of common dolphins, to enable assessment of fisheries impacts and risk at the scale of regional subpopulations (if any).

MPI monitors interactions between fishing vessels and marine mammals primarily via the observer programme. In addition, MPI and the deepwater quota owners and trawl operators have developed a Marine Mammal Operating Procedure (MMOP) that specifies how skippers of trawlers greater than 28 m in length are expected to provide reports to the government of all marine mammal interactions, and specifies what fishers should do reduce capture rates and fisheries risk. Observer reviews provide information that contributes to managing interaction of the deepwater fleet. Specific risk management actions are identified for implementation in all JMA trawl fisheries, and there are additional requirements north of latitude 40° 30' S where most interactions occur.

Vessel practices required under the MMOP include: refraining from deploying fishing gear when dolphins are present; assigning an officer on watch and deck to report all sightings; ensuring trawl gear is closed during turns, by keeping doors at or above surface; using acoustic dissuasive devices attached to net on night-time tows for jack mackerel species; and (in the northern area) refraining from deploying trawl gear between 0230 and 0430 h. Additionally, under the MMOP all vessel officers are briefed annually on the risk factors regarding common dolphin captures especially area, depth and temporal factors. The full requirements can be seen at http://deepwatergroup.org/wp-content/uploads/2016/11/Marine-Mammals-Operational-Procedures-2016-17.pdf.

Vessels are required to report any captures to all vessels in the vicinity (by VHF radio) and must also notify the DeepWater Group (DWG) within a 24 hour period, and record captures in the ship's log, any time a common dolphin is caught (see Annual Review Report for Deepwater Fisheries, http://www.mpi.govt.nz/document-vault/4090, for more information).

7.4.3 MODELLING POPULATION-LEVEL IMPACTS OF FISHERIES INTERACTIONS

Because common dolphins are abundant and widespread, fisheries interactions are not considered a threat to the population at a global scale. However, small subpopulations of common dolphins such as the Mediterranean Sea population have been significantly impacted by fishing.

The number of common dolphins captured in deepwater trawl fisheries is known with high certainty, due to high levels of observer coverage. Capture rates in poorly observed inshore fisheries are far less certain. Regardless of captures, the level of fisheries risk to common dolphins is estimated very poorly, in large part due to unknown population structure, such that there is no clear understanding of what size population these impacts should be considered against. New MPI research (PRO2017-08A) is underway applying genetic analyses to better understand common dolphin population structure and population size for potentially impacted populations, to improve estimates of fisheries risk. Other research is also in progress to estimate spatial distributions for New Zealand cetacean species (PRO2014-01), including common dolphins. Outputs from this work will inform spatially explicit estimates of encounter rate and capture rate in fisheries, which can then be applied to estimate population level risk at any spatial scale, applying the SEFRA method (Chapter 3, and below).

Total estimated captures per year varied between 0.15 (95% c.i.: 0.00–1.74) and 6.27 (95% c.i.: 2.49–12.27) captures per 100 tows over this 16-year period between the 1995–96 and 2011–12 fishing years (Thompson et al. 2013, Berkenbusch et al. 2013, Abraham & Berkenbusch 2017). The majority of observed common dolphin bycatch events in New Zealand waters have been in trawl fisheries targeting jack mackerel (*Trachurus declivis, T. murphyi* and *T. novaezelandiae*) on the west coast of the North Island.

7.4.4 MULTI-SPECIES MARINE MAMMAL RISK ASSESSMENT

In 2017, a New Zealand Marine Mammal Risk Assessment (MMRA) was completed (Abraham et al. 2017) applying a

modification of the SEFRA method described in Chapter 3. Outputs of the MMRA suggest that common dolphins are the species potentially most at risk from New Zealand commercial fisheries. Fisheries risk to common dolphins is attributed primarily to pelagic trawl fisheries, for which historically observed captures are sufficient to estimate vulnerability and risk with some confidence, and also to inshore trawl and set-net fisheries, for which species vulnerability (hence total captures) is very poorly estimated (due to very low levels of historical observer coverage). Furthermore, as previously noted, estimates of biological population size are highly uncertain due to unknown population structure. As a consequence, cumulative fisheries risk for common dolphins remains highly uncertain, with an estimated risk score that may be less than half the Population Sustainability Threshold (PST) or may exceed the PST by a factor of two (Figure 7.9). (Note that the particular definition of PST used in the multi-species MMRA represents a number of anthropogenic deaths that would allow population recovery to, or stabilisation at, 50% of K with 90% certainty. Other species-specific risk assessments may adopt other population reference outcomes in the definition of PST, reflecting policy choices.)

Estimated fishery-related deaths for common dolphins in each fishery group, as estimated in the MMRA, are shown in Figure 7.10.

In 2017 an independent expert review of the SEFRA method and its implementations, including the (at that time unpublished) MMRA, made recommendations to improve this and future implementations of the MMRA (Lonergan et al. 2017). Of particular relevance to common dolphins, the review cautioned against uncritical use of Delphi-derived spatial species distribution layers as inputs. Research is currently in progress to estimate common dolphin distributions empirically on a finer spatial scale, using habitat suitability models informed by sightings data (PRO2014-01). When outputs of this work is available, it is expected that these will be combined with improved population estimates (from PRO2017-08A) in an updated marine mammal risk assessment.

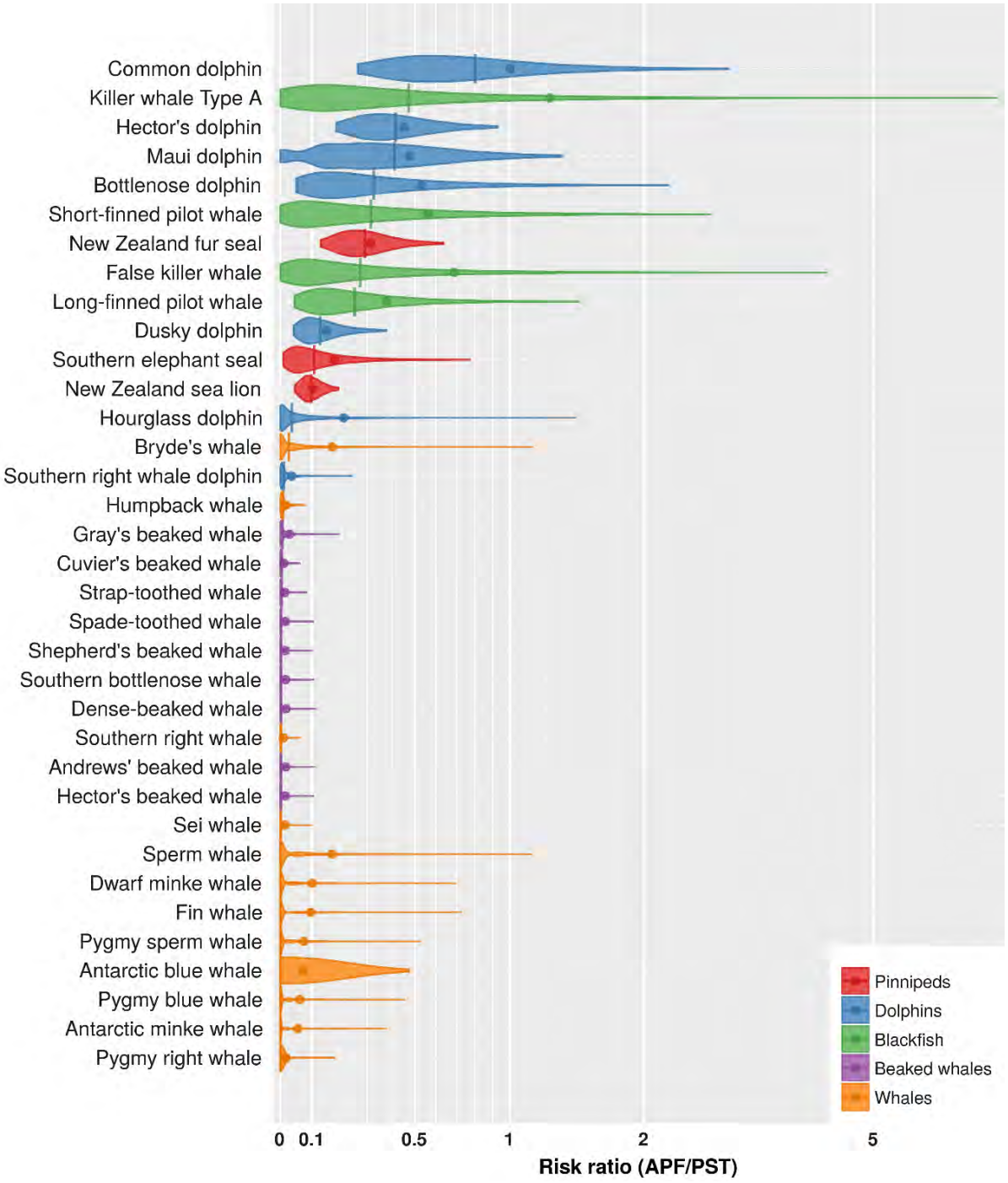


Figure 7.9: Cumulative fishery risk across all fishery groups as estimated by the 2016 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017). Species groups are colour coded.

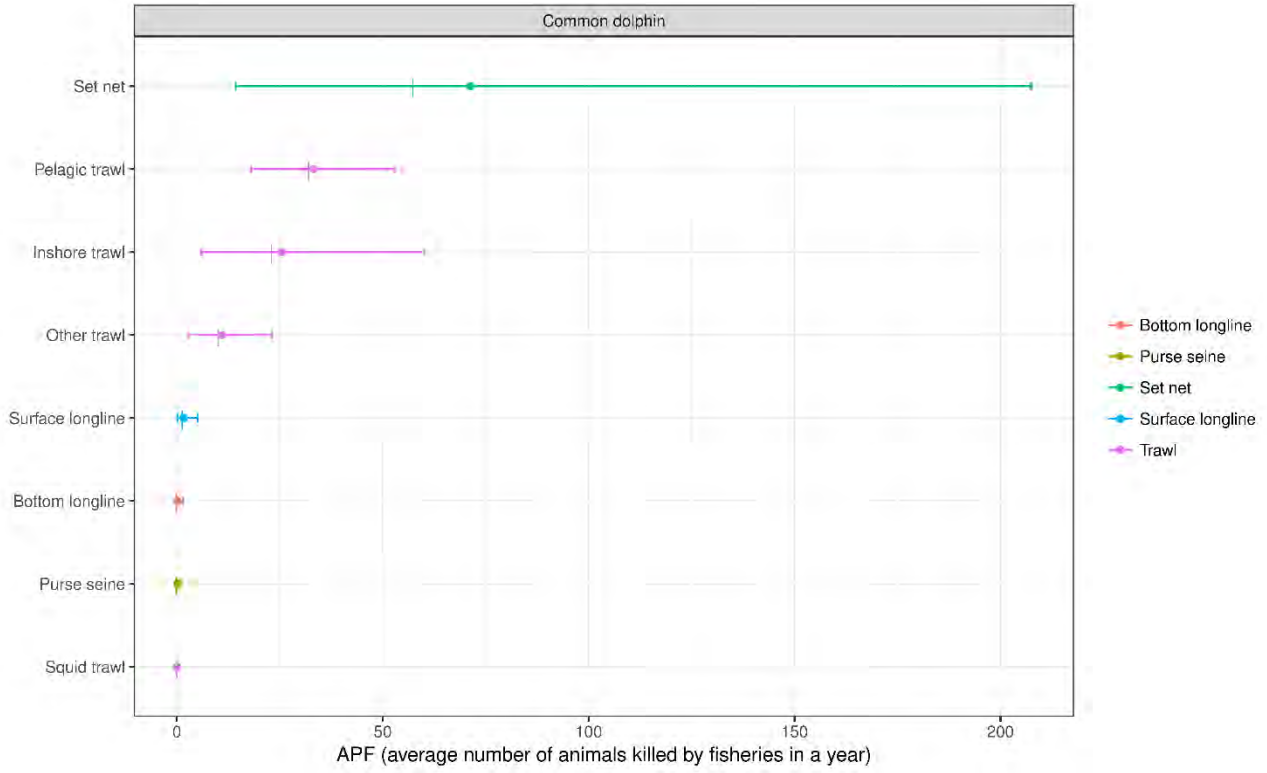


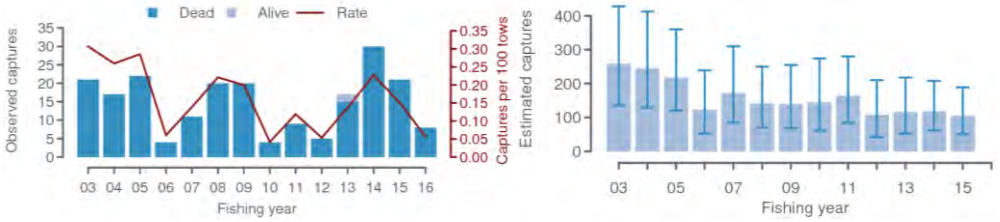
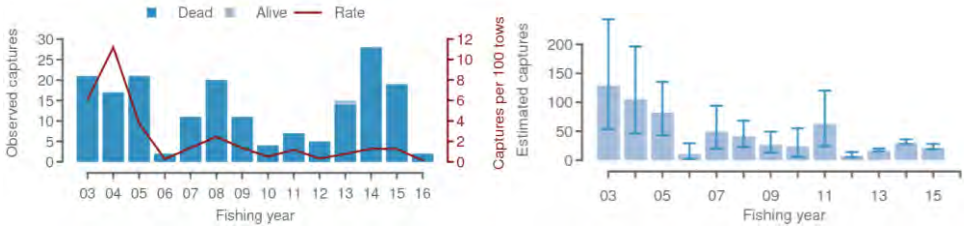
Figure 7.10: Annual fishery-related deaths of common dolphins in each fishery group, as estimated by the 2017 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017).

7.4.5 SOURCES OF UNCERTAINTY

While there is an abundance of knowledge on common dolphins worldwide, relatively little is known about this species in New Zealand waters. The latest research suggests that common dolphins in New Zealand waters are a larger form of the short-beaked common dolphin found elsewhere; however further work is needed to verify this conclusion, which is based on a study with small sample size (Jordan 2012). As identified above, there is considerable uncertainty regarding population size and/or subpopulation structure of common dolphins around New Zealand. MPI project PRO2017-08A will address this uncertainty.

Due to historically low levels of observer coverage incidental captures of common dolphins by inshore fisheries are only poorly estimated. Improved observer coverage or monitoring by other means may help to address this uncertainty. Where captures are observed, improved understanding of factors affecting capture rates in different parts of the fishing event (i.e., setting, towing, or hauling) may be useful to inform management strategies or mitigation options to reduce captures.

7.5 INDICATORS AND TRENDS

Population size	Unknown in New Zealand EEZ, but approximately 4 000 000 worldwide.[1]
Population trend	Unknown.
Threat status	New Zealand: Not Threatened; Data Poor, and Secure Overseas in 2013.[2] IUCN: Least Concern, in 2008.[3]
Number of interactions[4]	46 estimated captures (95% c.i.: 32–67) in modelled trawl fisheries in 2014–15[4] 8 observed captures in trawl fisheries in 2015–16[4] 23 estimated captures (95% c.i.: 19–30) in the jack mackerel trawl fisheries in 2014–15[4] 19 observed captures in the jack mackerel trawl fisheries in 2014–15[4] 2 observed captures in the jack mackerel trawl fisheries in 2015–16[4] 142.7 estimated annual potential fatalities (APF) (95% c.i.: 70.7–285.1)[5]
Trends in interactions[4]	Trawl fisheries:  Jack mackerel trawl fishery: 

7.6 REFERENCES

Abraham, E R; Berkenbusch, K (2017) Estimated captures of New Zealand fur seal, New Zealand sea lion, common dolphin, and turtles in New Zealand commercial fisheries, 1995–96 to 2014–15. *New Zealand Aquatic Environment and Biodiversity Report No. 188*. 66 p.

Abraham, E R; Berkenbusch, K; Neubauer, P; Richard, Y (2017) Assessment of the risk to New Zealand marine mammals from commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 189*. 123 p.

Abraham, E R; Thompson, F N (2011) Summary of the capture of seabirds, marine mammals, and turtles in New Zealand commercial fisheries, 1998-99 to 2008-09. Final research report prepared for Ministry of Fisheries project PRO2007/01. 170 p.

Baird, S (1994) Nonfish species and fisheries interactions working group report 94/1. 27 p.

Baird, S (1996) Nonfish species and fisheries interactions working group report 96/1. 27 p.

[1] Hammond et al. (2008).
[2] Baker et al. (2016).
[3] Hammond et al. (2008).
[4] For more information, see: http://data.dragonfly.co.nz/psc
[5] Abraham & Berkenbusch 2017.

Baker, C S; Chilvers, B L; Childerhouse, S; Constantine, R; Currey, R; Mattlin, R; van Helden, A; Hitchmough, R; Rolfe, J (2016) Conservation status of New Zealand marine mammals, 2013. *New Zealand Threat Classification Series 14*. Department of Conservation, Wellington. 18 p.

Bearzi, G; Agazzi, S; Gonzalvo, J; Costa, M; Bonizzoni, S; Politi, E; Piroddi, C; Reeves, R R (2008) Overfishing and the disappearance of short-beaked common dolphins from western Greece. *Endangered Species Research 5 (1)*: 1–12.

Bearzi, G; Reeves, R R; Notarbartolo-di Sciara, G; Politi, E; Canadas, A; Frantzis, A; Mussi, B (2003) Ecology, status and conservation of short-beaked common dolphins *Delphinus delphis* in the Mediterranean Sea. *Mammal Review 33 (3–4)*: 224–252.

Bell, C; Kemper, C; Conran , J (2002) Common dolphins (*Delphinus delphis*) in southern Australia: A morphometric study. *Australian Mammalogy 24 (1)*: 1–10.

Berkenbusch, K; Abraham, E R; Torres, L (2013) New Zealand marine mammals and commercial fisheries. *New Zealand Aquatic Environment and Biodiveristy Report No. 119*. 110 p.

Brager, S; Schneider, K (1998) Near-shore distribution and abundance of dolphins along the West Coast of the South Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research 32 (1)*: 105–112.

Brophy, J; Murphy, S; Rogan, E (2009) The diet and feeding ecology of the short-beaked common dolphin (*Delphinus delphis*) in the northeast Atlantic. International Whaling Commission Scientific Committee paper SC/61/SM 14.

Carretta, J; Price, T; Peterson, D; Rear, R (2005) Estimates of marine mammals, sea turtle, and seabird mortality in the California drift gillnet fishery for swordfish and thresher shark, 1996-2002. *Marine Fisheries Review 66*: 21–30.

Cawthorn, M (2009) Incidental cetacean sighting records by transiting ships between New Zealand and overseas ports. Cawthorn & Associates, Plimmerton, New Zealand.

Connor, R C; Wells, R S; Mann, J; Read, A J (2000) The bottlenose dolphin: social relationships in a fission-fusion society. *In*: Mann, J; Connor, R C; Tyack, P; Whitehead, H, Cetacean Societies: field studies of dolphins and whales. pp 91–125. University of Chicago Press, Chicago.

Constantine, R; Baker, C S (1997) Monitoring the commercial swim-with-dolphin operations in the Bay of Islands. Department of Conservation, New Zealand. 59 p.

de Boer, M N; Saulino, J T; Leopold, M F; Reijnders, P J; Simmonds, M P (2012) Interactions Between Short-Beaked Common Dolphin (*Delphinus delphis*) and the Winter Pelagic Pair-Trawl Fishery off Southwest England (UK). *International Journal of Biodiversity and Conservation 4 (13)*: 481–499.

Deepwater Group (2016) 2014/15 Marine Mammal Operating Procedures, updated September 2016. Retrieved from http://deepwatergroup.org/wp-content/uploads/2016/11/Marine-Mammals-Operational-Procedures-2016-17.pdf.

Duignan, P J; Gibbs, N J; Jones, G W (2003) Autopsy of cetaceans incidentally caught in fishing operations 1997/98, 1999/2000, and 2000/01. *DOC Science Internal Series No. 119*. 66 p.

Duignan, P J; Gibbs, N J; Jones, G W (2004) Autopsy of cetaceans incidentally caught in commercial fisheries, and all beachcast specimens of Hector's dolphins, 2001/02. *DOC Science Internal Series No. 176*. 28 p.

Duignan, P J; Jones, G W (2005) Autopsy of cetaceans including those incidentally caught in commercial fisheries, 2002/03. *DOC Science Internal Series No. 195*. 22 p.

Evans, W E (1994) Common dolphin, white-bellied porpoise *Delphinus delphis* Linnaeus, 1758. *Handbook of Marine Mammals.*

Ferrero, R; Walker, W (1995) Growth and reproduction of the common dolphin, *Delphinus delphis* Linnaeus, in the offshore waters of the North Pacific Ocean. *Fishery Bulletin 93 (3)*: 483–494.

Forcada, J; Hammond, P (1998) Geographical variation in abundance of striped and common dolphins of the western Mediterranean. *Journal of Sea Research 39 (3)*: 313–325.

Gaskin, D E (1968) Distribution of Delphinidae (Cetacea) in relation to sea surface temperatures off eastern and southern New Zealand. *New Zealand Journal of Marine and Freshwater Research 2 (3)*: 527–534.

Gaskin, D E (1972) Whales, dolphins, and seals: with special reference to the New Zealand region. Heiemann Educational Books. 200 p.

Gilman, E L; Lundin, C G (2009) Minimizing bycatch of sensitive species groups in marine capture fisheries: lessons from tuna fisheries. *In*: Grafton, Q; Hillborn, R; Squires, D; Tait, M; Williams, M (Eds.), Handbook of Marine Fisheries Conservation and Management. pp 150. Oxford University Press, Oxford.

Gonzalez, A F; López, A; Benavente, P (1999) A multiple gestation in a *Delphinus delphis* stranded on the north-western Spanish coast. *Journal of the Marine Biological Association of the UK 79 (06)*: 1147–1148.

Hall, M A; Alverson, D L; Metuzals, K I (2000) By-catch: problems and solutions. *Marine Pollution Bulletin 41 (1)*: 204–219.

Hammond, P S; Bearzi, G; Bjørge, A; Forney, K; Karczmarski, L; Kasuya, T; Perrin, W F; Scott, M D; Wang, J Y; Wells, R S; Wilson, B (2008) *Delphinus delphis*. The IUCN Red List of Threatened Species. Retrieved from http://www.iucnredlist.org.

Heyning, J E; Perrin, W F (1994) Evidence for two species of common dolphins (genus *Delphinus*) from the eastern North Pacific. *Contributions in Science, Serial Publications of the Natural History Museum of Los Angeles County 442*: 1–35.

Jefferson, T; Curry, B; Leatherwood, S; Powell, J (1997) Dolphins and porpoises of West Africa: a review of records (Cetacea: Delphinidae, Phocoenidae). *Mammalia 61 (1)*: 87–108.

Jefferson, T A; Waerebeek, K V (2002) The taxonomic status of the nominal dolphin species *Delphinus tropicalis* van Bree, 1971. *Marine Mammal Science 18 (4)*: 787–818.

Jefferson, T A; Webber, M A; Pitman, R L (2011) Marine Mammals of the World: A Comprehensive Guide to Their Identification. Academic Press. 573 p.

Jordan, F (2012) Skull morphometry of the common dolphin, *Delphinus* sp., from New Zealand waters. Master's Thesis, Massey University, Auckland, New Zealand. 242 p.

Jordan, F F; Murphy, S; Martinez, E; Amiot, C; Helden, A; Stockin, K A (2015) Criteria for assessing maturity of skulls in the common dolphin, *Delphinus* sp., from New Zealand waters. *Marine Mammal Science 31 (3)*: 1077–1097.

Joseph, J (1994) The tuna-dolphin controversy in the eastern pacific ocean: Biological, economic, and political impacts. *Ocean Development & International Law 25 (1)*: 1–30.

Kemper, C; Flaherty, A; Gibbs, S; Hill, M; Long, M; Byard, R (2005) Cetacean captures, strandings and mortalities in South Australia 1881-2000, with special reference to human interactions. *Australian Mammalogy 27 (1)*: 37–47.

Lonergan, M E; Phillips, R A; Thomson, R B; Zhou, S (2017) Independent review of New Zealand's Spatially Explicit Fisheries Risk Assessment approach – 2017. *New Zealand Fisheries Science Review 2017/2*. 36 p.

McClatchie, S; Dunford, A (2003) Estimated biomass of vertically migrating mesopelagic fish off New Zealand. *Deep-Sea Research I 50*: 1263–1281.

Maigret, J (1994) Marine mammals and fisheries along the West African coast. *Rep. Int. Whal. Comm. Spec (15)*: 307–316.

Meissner, A M; Christiansen, F; Martinez, E; Pawley, M; Orams, M B; Stockin, K A (2015) Behavioural Effects of Tourism on Oceanic Common Dolphins, *Delphinus* sp. in New Zealand: The Effects of Markov Analysis Variations and Current Tour Operator Compliance with Regulations. *Plos One 10 (1)*: e0116962.

Meynier, L; Stockin, K; Bando, M; Duignan, P (2008) Stomach contents of common dolphin (*Deiphinus* sp.) from New Zealand waters. *New Zealand Journal of Marine and Freshwater Research 42 (2)*: 257–268.

Ministry of Fisheries (2010) National Fisheries Plan for Highly Migratory Species. 43 p.

Ministry of Fisheries (2011) Draft National Fisheries Plan for Inshore Finfish. 47 p.

Morizur, Y; Berrow, S; Tregenza, N; Couperus, A; Pouvreau, S (1999) Incidental catches of marine-mammals in pelagic trawl fisheries of the northeast Atlantic. *Fisheries Research 41 (3)*: 297–307.

Ministry for Primary Industries (2012) National Operational Plan for Deepwater Fisheries for 2012/13. 71 p.

Ministry for Primary Industries (2015) Fisheries Assessment Plenary, May 2015: stock assessments and stock status. Compiled by the Fisheries Science Group, Ministry for Primary Industries, Wellington, New Zealand.

Murphy, S; Perrott, M; McVee, J; Read, F L; Stockin, K A (2014) Deposition of growth layer groups in dentine tissue of captive common dolphins *Delphinus delphis*. *NAMMCO Scientific Publications 10*.

Murphy, S; Pinn, E H; Jepson, P D (2013) The short-beaked common dolphin (*Delphinus delphis*) in the North-east Atlantic: Distribution, ecology, management, and conservation status. *Oceanography and Marine Biology: An Annual Review 51*: 193–280.

Neumann, D R (2001a) Activity budget of free-ranging common dolphins (*Delphinus delphis*) in the northwestern Bay of Plenty, New Zealand. *Aquatic Mammals 27 (2)*: 121–136.

Neumann, D R (2001b) The behaviour and ecology of short-beaked common dolphins (*Delphinus delphis*) along the east coast of Coromandel Peninsula, North Island, New Zealand: with a note on their interactions with humans. Massey Universtity. 344 p.

Neumann, D R (2001c) Seasonal movements of short-beaked common dolphins (*Delphinus delphis*) in the north-western Bay of Plenty, New Zealand: Influence of sea surface temperature and El Niño/La Niña. *New Zealand Journal of Marine and Freshwater Research 35 (2)*: 371–374.

Neumann, D; Orams, M B (2003) Feeding behaviours of short-beaked common dolphins, *Delphinus delphis*, in New Zealand. *Aquatic Mammals 29 (1)*: 137–149.

Neumann, D R; Orams, M (2005) Behaviour and ecology of common dolphins (*Delphinus delphis*) and the impact of tourism in Mercury Bay, North Island, New Zealand. Science & Technical Publishing, Department of Conservation. 40 p.

Neumann, D R; Russell, K; Orams, M B; Baker, C S; Duignan, P (2002) Identifying sexually mature, male short-beaked common dolphins (*Delphinus delphis*) at sea, based on the presence of a postanal hump. *Aquatic Mammals 28 (2)*: 181–187.

O'Callaghan, T M; Baker, C S (2002) Summer cetacean community, with particular reference to Bryde's whales, in the Hauraki Gulf, New Zealand. *DOC Science Series.* Department of Conservation, Wellington. 18 p.

O'Driscoll, R L; Gauthier, S; Devine, J A (2009) Acoustic estimates of mesopelagic fish: as clear as day and night? *ICES Journal of Marine Science 66*: 1310–1317.

O'Driscoll, R L; Oeffner, J; Ross, O; Dunford, A J; McMillan, P J (2013) Pilot acoustic survey for jack mackerel on the west coast New Zealand (JMA7).. *New Zealand Fisheries Assessment Report 2013/1*. 53 p.

Overholtz, W J (1991) *Delphlnus delphis* in the Mid-Atlantlc Blght during Sprlng 1989. *Fishery Bulletin 89*: 723–728.

Perrin, W (1989) Dolphins, porpoises, and whales. An action plan for the conservation of biological diversity 1988–1992. IUCN, Gland, Switzerland. 27 p.

Piroddi, C; Bearzi, G; Gonzalvo, J; Christensen, V (2011) From common to rare: the case of the Mediterranean common dolphin. *Biological Conservation 144 (10)*: 2490–2498.

Pusineri, C; Magnin, V; Meynier, L; Spitz, J; Hassani, S; Ridoux, V (2007) Food and feeding ecology of the common dolphin (*Delphinus delphis*) in the oceanic Northeast Atlantic and comparison with its diet in neritic areas. *Marine Mammal Science 23 (1)*: 30–47.

Reeves, R R (2003) Dolphins, whales and porpoises: 2002–2010 conservation action plan for the world's cetaceans. IUCN/SSC Cetacean Specialist Group. 139 p.

Reeves, R R; Berggren, P; Crespo, E A; Gales, N; Northridge, S P; di Sciara, G N; Perrin, W F; Read, A J; Rogan, E; Smith, B D (2005) Global priorities for reduction of cetacean bycatch. Report to the World Wildlife Fund for Nature. 28 p.

Reeves, R R; Notarbartolo di Sciara, G (2006) The status and distribution of cetaceans in the Black Sea and Mediterranean Sea. IUCN Centre for Mediterranean Cooperation, Malaga, Spain. 137 p.

Reijnders, P; Lankester, K (1990) Status of marine mammals in the North Sea. *Netherlands Journal of Sea Research 26 (2)*: 427–435.

Rosel, P; Dizon, A; Heyning, J (1994) Genetic analysis of sympatric morphotypes of common dolphins (genus *Delphinus*). *Marine Biology 119 (2)*: 159–167.

Rowe, S J (2007) A review of methodologies for mitigating incidental catch of protected marine mammals. *DOC Research and Development Series No. 283*. 48 p.

Silva, M (1999) Diet of common dolphins, *Delphinus delphis*, off the Portuguese continental coast. *Journal of the Marine Biological Association of the UK 79 (03)*: 531–540.

Silva, M A; Sequeira, M (2003) Patterns in the mortality of common dolphins (*Delphinus delphis*) on the Portuguese coast, using stranding records, 1975-1998. *Aquatic Mammals 29 (1)*: 88–98.

Stockin, K A (2008) The New Zealand common dolphin (*Delphinus* sp.). Massey University. 235 p.

Stockin, K A; Amaral, A R; Latimer, J; Lambert, D M; Natoli, A (2014) Population genetic structure and taxonomy of the common dolphin (*Delphinus* sp.) at its southernmost range limit: New Zealand waters. *Marine Mammal Science 30 (1)*: 44–63.

Stockin, K A; Binedell, V; Wiseman, N; Brunton, D H; Orams, M B (2009a) Behavior of free-ranging common dolphins (*Delphinus* sp.) in the Hauraki Gulf, New Zealand. *Marine Mammal Science 25 (2)*: 283–301.

Stockin, K A; Duignan, P J; Roe, W D; Meynier, L; Alley, M; Fettermann, T (2009b) Causes of mortality in stranded common dolphin (*Delphinus* sp.) from New Zealand waters between 1998 and 2008. *Pacific Conservation Biology 15 (3)*: 217–227.

Stockin, K A; Murphy, S N; Duignan, P J; Perrott, M M; Jones, G W; Roe, W D (2011) Reproductive biology of New Zealand common dolphins (*Delphinus* sp). *In*: 19th Biennial Conference on the Biology of Marine Mammals. pp 283. Tampa, Florida, 27 November-2 December 2011.

Stockin, K; Orams, M (2009) The status of common dolphins (*Delphinus delphis*) within New Zealand waters. *Journal of Cetacean Research Management SC/61/SM20.*

Stockin, K A; Pierce, G J; Binedell, V; Wiseman, N; Orams, M B (2008) Factors affecting the occurrence and demographics of common dolphins (*Delphinus* sp.) in the Hauraki Gulf, New Zealand. *Aquatic Mammals 34 (2)*: 200–211.

Stockin, K A; Visser, I N (2005) Anomalously Pigmented Common Dolphins (*Delphinus* Dolphins sp.) off Northern New Zealand. *Aquatic Mammals 31 (1)*: 43–51.

Thompson, F N; Abraham, E R; Berkenbusch, K (2010) Common dolphin (*Delphinus Delphis*) bycatch in New Zealand mackerel trawl fisheries, 1995-96 to 2008-09. *New Zealand Aquatic Environment and Biodiversity Report No. 63.* 20 p.

Thompson, F N; Abraham, E R; Berkenbusch, K (2013a) Common dolphin (*Delphinus delphis*) bycatch in New Zealand commercial trawl fisheries. *Plos One 8 (5)*: e64438.

Thompson, F N; Abraham, E R; Berkenbusch, K (2013b) Marine Mammal Bycatch in New Zealand Trawl Fisheries, 1995-96 to 2010-11. *New Zealand Aquatic Environment and Biodiversity Report No. 105.* 73 p.

Tudela, S; Kai, A K; Maynou, F; El Andalossi, M; Guglielmi, P (2005) Driftnet fishing and biodiversity conservation: the case study of the large-scale Moroccan driftnet fleet operating in the Alboran Sea (SW Mediterranean). *Biological Conservation 121 (1)*: 65–78.

Vella, A (2005) Proceedings of the workshop on Common dolphins: current research, threats and issues. *European Cetacean Society Newsletter Special Issue No. 45.* 39 p.

Viricel, A; Strand, A E; Rosel, P E; Ridoux, V; Garcia, P (2008) Insights on common dolphin (*Delphinus delphis*) social organization from genetic analysis of a mass-stranded pod. *Behavioral Ecology and Sociobiology 63 (2)*: 173–185.

Westgate, A J; Read, A J (2007) Reproduction in short-beaked common dolphins (*Delphinus delphis*) from the western North Atlantic. *Marine Biology 150 (5)*: 1011–1024.

Young, D D; Cockcroft, V (1994) Diet of common dolphins (*Delphinus delphis*) off the south-east coast of southern Africa: opportunism or specialization? *Journal of Zoology 234 (1)*: 41–53.

AEBAR 2018: *Snapshot* of Chapter 8 - Seabirds

SEABIRD BYCATCH



Albatrosses, petrels, shearwaters

1. THE ISSUE

Unfortunately, fishing operations are a threat to seabirds. The birds are attracted to fishing activities and can get entangled in wires or nets, suffer damage by flying into trawl warps, get hooked by longlines or drowned by the gear.

New Zealand is the seabird capital of the world with 85 species of albatross, petrel, penguin and shags breeding here. Many species are classified as endangered or threatened. MPI, industry and DOC work together to reduce mortality and bycatch.

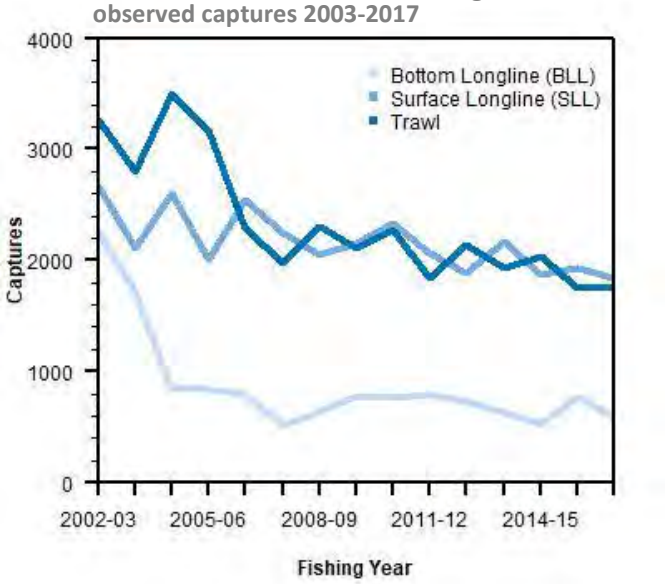
2. Total seabird fatalities are estimated to be 14,400 birds a year.

Trawling by inshore vessels and deepwater fisheries such as hoki, squid and scampi, as well as bottom long line vessels fishing for ling and snapper are the most problematic. Surface longliners targeting southern bluefin tuna, bigeye tuna and swordfish also capture seabirds. Bycatch levels appear to be declining (see 3 ⟹).

Estimates of seabird deaths are modelled from observer data and include cryptic mortality (unseen deaths) (see 4 ⇓).

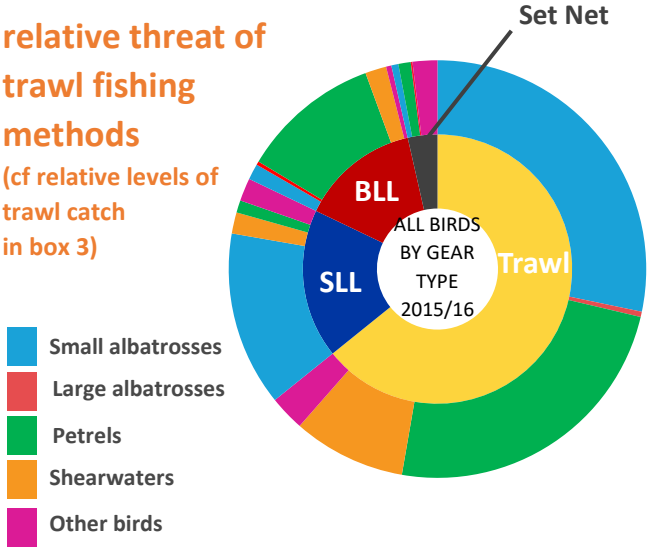
3. Annual seabird bycatch estimates*

*based on statistical modelling of observed captures 2003-2017



4. Adding cryptic mortality to the model increases the relative threat of trawl fishing methods

(cf relative levels of trawl catch in box 3)



5. New Zealand's National Plan of Action for seabirds (NPOA)

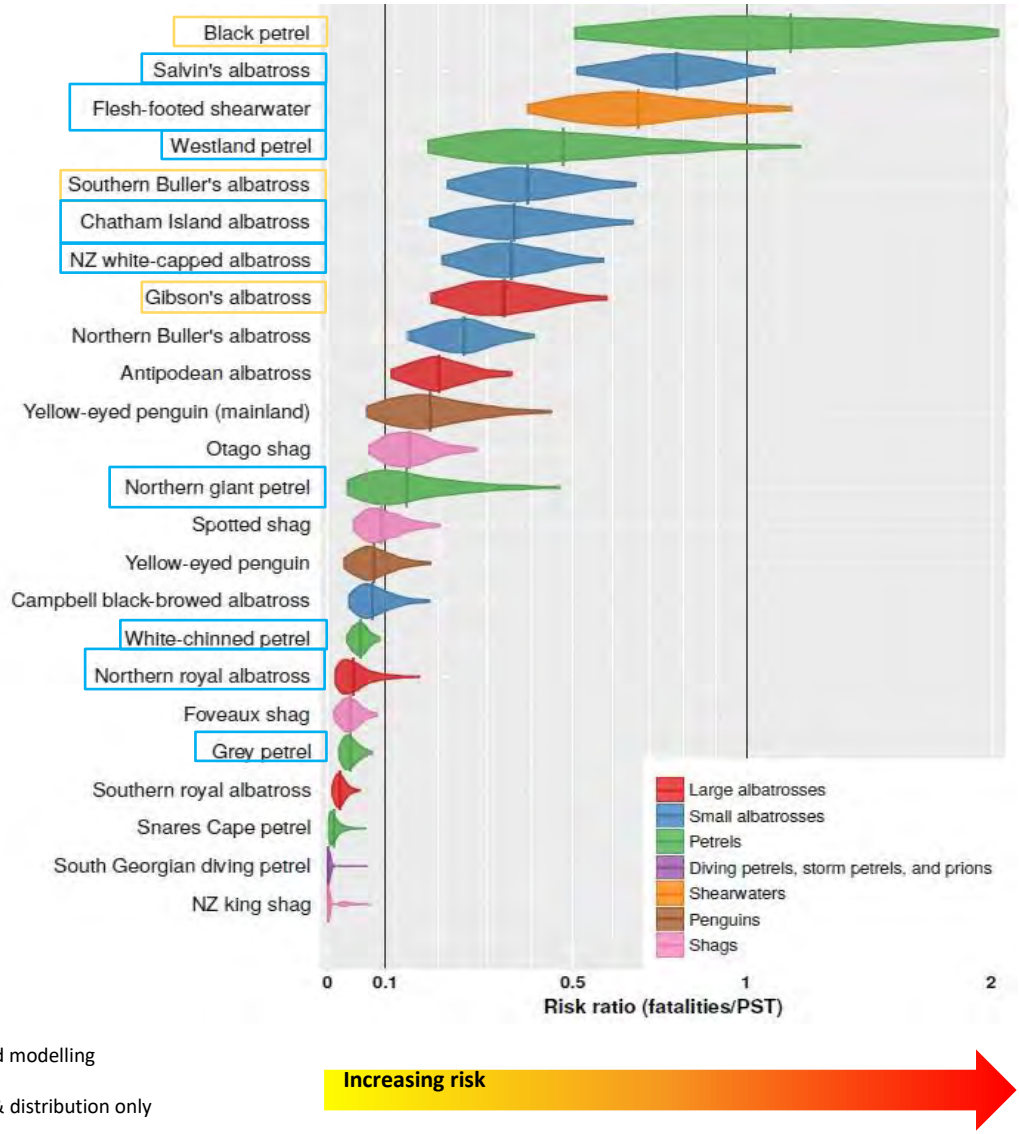
The NPOA describes fisheries objectives for the prevention, monitoring and management of incidental seabird capture in New Zealand and identifies nine action points outlined in this chapter.

The NPOA is administered by DOC and MPI and it is currently under review. The NPOA is implemented through national fisheries planning processes.

6. New Zealand uses a semi-quantitative Risk Assessment to identify species at greatest risk and prioritise research

- The assessment measures the risk to multiple seabird species from multiple fisheries.
- The risk ratio is an estimate of annual potential seabird fatalities relative to the population sustainability threshold (PST). It includes cryptic mortality across trawl and long line fisheries. Risk ratios > 1 means that the population is declining.
- Black petrel is the only species at VERY high risk, and there are 7 species at high risk.
- Inclusion of fisheries outside EEZ and recreational captures of seabirds will improve future models of risk.

7. Output from the risk assessment



Fully quantified modelling

Demographic & distribution only

Increasing risk

8. Nationally critical case studies and overall progress:

20 years ago, we knew very little about the scale of seabird captures, how to mitigate it or how to address the problem. *Now, scientifically robust demographic and distribution studies* have been completed for 10 of the 24 species in the risk assessment above. *Fully quantitative modelling* has been completed for black petrel and southern Buller's albatross and Gibson's wandering albatross; mitigation trials to decrease bird mortality and injury at sea have been implemented.

Population trends remain difficult to interpret for all species except southern Buller's albatross and Gibson's wandering albatross. The role of fishing-related mortality in driving population trends is also unclear and determining the sources of uncertainty in risk assessments remain critical.

8. NEW ZEALAND SEABIRDS

Status of chapter	This chapter has been fully updated for AEBAR 2018.
Scope of chapter	This chapter focuses on estimates of captures and risk assessments conducted for seabirds that breed in New Zealand waters. Also included are descriptions of the nature of fishing interactions, the management context and approach, trends in key indicators and major sources of uncertainty. It includes details only on species that have been the focus of MPI research; demographic studies on individual seabird species (10 taxa), and 5 taxa for which quantitative population modelling has been conducted.
Area	New Zealand EEZ and Territorial Sea (noting that many seabirds are highly migratory and spend prolonged periods outside the New Zealand EEZ; on the high seas these effects are considered by CCSBT, WCPFC, CCAMLR, SPRFMO, etc. and New Zealand capture estimates are reported to those bodies).
Focal localities	Interactions with fisheries occur in many parts of the EEZ and TS as well as on the high seas and in the EEZs of other nations.
Key issues	Quantitative and semi-quantitative risk assessments can be improved through better estimates of: incidental captures in fisheries that are poorly or unobserved; species identity, especially of birds released alive; cryptic mortality rates; survival of birds released alive; improved understanding of seabird distributions; and the ability of seabird populations to sustain given levels of fisheries mortality, especially given fisheries interactions and captures outside the New Zealand EEZ and in non-commercial fisheries. Assessing total fisheries impacts (i.e., including non-commercial and out-of-zone) and fisheries impacts in the context of other factors influencing seabird survival and reproduction, including other anthropogenic effects. Consolidating population modelling and risk assessment results are key challenges. Black petrel remains a key focus of risk assessment research, with Antipodean albatross also a key species in terms of understanding the drivers of population trajectory and of demographic parameters.
Emerging issues	Potential new fishery monitoring techniques. Yellow-eyed penguin is an emerging species in relation to fisheries interactions. Global observer programs.
MPI research (current)	PRO2017-01A *Demographic parameters of black petrels;* PRO2017-01B *Demographic parameters of Southern Buller's*; PRO2017-05A *Population modelling of black petrels*; PRO2017-05B *Population modelling of Chatham Island Albatross*; PRO2017-06 *Characterisation of yellow-eyed penguin/fishery interactions*; PRO2017-15 *Innovative tag technology to examine foraging patterns of seabirds and association with fishing vessels*; PRO2017-19 *Capture rate of black petrels and flesh-footed shearwaters*; SEA2017-08 *A synthesis of the 'population' work in PRO2006-01*; SEA2017-10 *Black petrel electronic monitoring audit and analysis*; DAE2015-01 *Characterisation of seabird capture data*; PRO2015-01 *Improving estimates of cryptic mortality for seabird risk assessments*; PRO2014-06 *Update level-2 seabird risk assessment*.
NZ government research (current)	DOC Conservation Services Programme (CSP) projects: INT2017-02 *Observing commercial fisheries*, INT2016-02 *Identification of seabirds captured in New Zealand fisheries,* INT2017-02 *Supporting the utility of electronic monitoring to identify protected species interacting with commercial fisheries,* INT2015-04 *Black petrel and flesh-footed shearwater foraging behaviour around fishing vessels,* POP2015-02 *Flesh-footed shearwater population*, POP2016-05 *Yellow-eyed penguin foraging and indirect effects*, POP2017-01 *Seabird population research: Chatham Island 2017-18,* POP2017-03 *Salvin's albatross: Bounties Islands population*, POP2017-04 *Seabird population research: Auckland Islands 2017-18,* POP2017-*06 Indirect effects on seabird in north-east North Island region,* MIT2016-01 *Protected species bycatch newsletter*, MIT2017-01 *Protected species liaison,* MIT2017-02 *Characterisation and development of offal management for small vessels*, MIT2017-03 *Characterisation and mitigation of protected species interactions in the inshore trawl fishery.*
Related chapters/issues	National Plan of Action (2013) to Reduce the Incidental Catch of Seabirds in New Zealand Fisheries (MPI 2013)

8.1 CONTEXT

Seabird names and taxonomy in this document generally follow that adopted by the Ornithological Society of New Zealand (OSNZ 2010) except where a different classification has been agreed by the parties to the Agreement for the Conservation of Albatrosses and Petrels (ACAP) or the New Zealand Threat Classification Scheme (NZTCS) has classified multiple taxa within a single OSNZ species. There are probably more than 10 000 bird species worldwide, but fewer than 400 are classified as seabirds (being specialised marine foragers). All but seven seabird taxa in New Zealand are absolutely protected under s3 of the Wildlife Act 1953, meaning that it is an offence to hunt or kill them. Southern black-backed gull, *Larus dominicanus dominicanus*, is the only species that is not protected. Black shag, *Phalacrocorax carbo novaehollandiae*, and subantarctic skua, *Catharacta antarctica lonnbergi*, are partially protected, and sooty shearwater, *Puffinus griseus*, grey-faced petrel, *Pterodroma macroptera gouldi*, little shag, *Phalacrocorax melanoleucos brevirostris*, and pied shag, *Phalacrocorax varius varius*, may be hunted or killed subject to Minister's notification. Of the 99 seabird taxa that breed in New Zealand, 28 are considered threatened. For albatrosses and petrels, a key threat is injury or death in fishing operations, although the Wildlife Act provides defences if the accidental or incidental death or injury took place in the course of fishing pursuant to a permit, licence, authority, or approval issued, granted, or given under the Fisheries Act 1996, as long as the interaction is reported. Commercial fishers are required to complete a Non-Fish and Protected Species Catch Return (NFPSCR, s11E of the Fisheries (Reporting) Regulations 2001).

The Minister of Conservation may approve a Population Management Plan (PMP) for one or more species under s14F of the Wildlife Act and a PMP can include a maximum allowable level of fishing-related mortality for a species (MALFiRM). Such a limit would apply to New Zealand fisheries waters and would be for the purpose of enabling a threatened species to achieve a non-threatened status as soon as reasonably practicable, and in any event within a period not exceeding 20 years, or, in the case of non-threatened species, neither cause a net reduction in the size of the population nor seriously threaten the reproductive capacity of the species (s14G). No PMPs are in place for seabirds but, in the absence of a PMP, the Minister for Primary Industries may, after consultation with the Minister of Conservation, take such measures as they consider necessary to avoid, remedy, or mitigate the effect of fishing-related mortality on any protected species (s15(2) of the Fisheries Act 1996).

Relevant, high-level guidance from the 2005 statement of General Policy under the Conservation Act 1987 and Wildlife Act 1953 includes the following stated policies:

> 4.4 (f) Marine protected species should be managed for their long-term viability and recovery throughout their natural range.
>
> 4.4 (g) Where unprotected marine species are identified as threatened, consideration will be given to amending the Wildlife Act 1953 schedules to declare such species absolutely protected.
>
> 4.4 (j) Human interactions with marine mammals and other marine protected species should be managed to avoid or minimise adverse effects on populations and individuals.
>
> 4.4 (l) The Department should work with other agencies and interests to protect marine species.

New Zealand is a signatory to a number of international conventions and agreements to provide for the management of threats to seabirds, including:

- the United Nations Convention on the Law of the Sea (UNCLOS);
- the United Nations Fish Stocks Agreement (insofar as it relates to the conservation of non-target, associated and dependent species);
- the Convention on Biological Diversity (CBD);
- the Convention on Migratory Species (CMS);
- the Food and Agriculture Organisation's (FAO) International Plan of Action for Reducing the Incidental Catch of Seabirds in Longline Fisheries (IPOA);
- the FAO Code of Conduct for Responsible Fisheries and the interpretive Best Practice Technical Guidelines;
- the Agreement on the Conservation of Albatrosses and Petrels (ACAP)
- Western & Central Pacific Fisheries Commission (WCPFC)
- Convention on the Conservation and Management of High Seas Fishery Resources in the South Pacific Ocean (SPRFMO).

- Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR).

The ACAP agreement requires that parties achieve and maintain a favourable conservation status for selected albatross and petrel taxa. Under the IPOA-seabirds, New Zealand developed a National Plan of Action (NPOA) to reduce the incidental catch of seabirds in New Zealand fisheries in 2004 (MFish & DOC 2004) and recently revised NPOA-seabirds (MPI 2013) (http://www.fish.govt.nz/en-nz/Environmental/Seabirds/default.htm). The scopes of the 2004 and 2013 NPOA are broader than the original IPOA to facilitate a coordinated and long-term approach to reducing the impact of fishing activity on seabirds. The 2013 NPOA covers all New Zealand fisheries and has a long-term objective that *'New Zealand seabirds thrive without pressure from fishing related mortalities, New Zealand fishers avoid or mitigate against seabird captures and New Zealand fisheries are globally recognised as seabird friendly.'* There are high-level subsidiary objectives related to practical aspects, biological risk, research and development, and international issues. Implementation is largely through MPI fisheries plans (see below). More detail is included in Section 8.4.1**Error! Reference source not found.**, Quantifying *fisheries interactions*.

Strictly speaking, birds considered 'captured' in this context are those that have been brought on board a fishing vessel having been impaled on a hook, entangled by a line, trapped in a net, snagged on a trawl warp or otherwise retrieved by some part of the fishing gear.

Management of fishing-related mortality of seabirds is consistent with Fisheries 2030 Objective 6: Manage impacts of fishing and aquaculture. Further, the management actions follow Strategic Action 6.2: *Set and monitor environmental standards, including for threatened and protected species and seabed impacts.*

All National Fisheries Plans except that for freshwater fisheries are relevant to the management of fishing-related mortality of seabirds.

Under the National Fisheries Plan for Deepwater and Middle-depth Fisheries, the objective most relevant for management of seabirds is Management Objective 2.5: *Manage deepwater and middle-depth fisheries to avoid or minimise adverse effects on the long-term viability of endangered, threatened and protected species.*

Management Objective 7 of the National Fisheries Plan for Highly Migratory Species (HMS) is to '*Implement an ecosystem approach to fisheries management, taking into account associated and dependent species'*.

The Environment Objective is the same for all groups of fisheries in the draft National Fisheries Plan for Inshore Finfish and the draft National Fisheries Plan for Inshore Shellfish, to *'Minimise adverse effects of fishing on the aquatic environment, including on biological diversity'*. The draft National Fisheries Plan for Freshwater has the same objective but is unlikely to be relevant to management of fishing-related mortality of seabirds

Table 8.1: List of New Zealand seabird taxa, excluding occasional visitors and vagrants, according to the Ornithological Society of New Zealand (OSNZ 2010) unless otherwise indicated (all taxa under the New Zealand Threat Classification System are listed, ACAP taxonomy generally takes precedence). International Union for the Conservation of Nature (IUCN) and New Zealand (DOC) classifications are shown (http://www.iucnredlist.org and Robertson et al. 2017 at http://www.doc.govt.nz/documents/science-and-technical/nztcs19entire.pdf). [Continued on following pages]

Common name	Scientific name	DOC category	IUCN category
Albatrosses			
Antipodean albatross	*Diomedea antipodensis antipodensis*	Threatened: Nationally Critical	#Endangered
Gibson's albatross	*Diomedea antipodensis gibsonii*	Threatened: Nationally Critical	#Endangered
Southern royal albatross	*Diomedea epomophora*	At Risk: Naturally Uncommon	Vulnerable
Wandering albatross	*Diomedea exulans*	Non-Resident Native: Migrant	Vulnerable
Northern royal albatross	*Diomedea sanfordi*	At Risk: Naturally Uncommon	Endangered

Common name	Scientific name	DOC category	IUCN category
Light mantled sooty albatross	*Phoebetria palpebrata*	At Risk: Declining	Near Threatened
Southern Buller's albatross	*Thalassarche bulleri bulleri*	At Risk: Naturally Uncommon	#Near Threatened
Northern Buller's albatross	*Thalassarche bulleri platei.*	At Risk: Naturally Uncommon	#Near Threatened
Eastern yellow-nosed albatross	*Thalassarche carteri*	Non-Resident Native: Coloniser	Endangered
Grey-headed albatross	*Thalassarche chrysostoma*	Threatened: Nationally Vulnerable	Endangered
Chatham Island albatross	*Thalassarche eremita*	At Risk: Naturally Uncommon	Vulnerable
Campbell Island albatross	*Thalassarche impavida*	Threatened: Nationally Vulnerable	Vulnerable
Black-browed albatross	*Thalassarche melanophris*	Non-Resident Native: Coloniser	Least Concern
Salvin's albatross	*Thalassarche salvini*	Threatened: Nationally Critical	Vulnerable
White-capped albatross*	*Thalassarche steadi**	At Risk: Declining	Near Threatened
Shearwaters			
North Island little shearwater	*Puffinus assimilis haurakiensis*	At Risk: Recovering	#Least Concern
Kermadec little shearwater	*Puffinus assimilis kermadecensis*	At Risk: Relict	#Least Concern
Buller's shearwater	*Puffinus bulleri*	At Risk: Naturally Uncommon	Vulnerable
Flesh-footed shearwater	*Puffinus carneipes*	Threatened: Nationally Vulnerable	Near Threatened
Subantarctic little shearwater	*Puffinus elegans*	At Risk: Naturally Uncommon	Least Concern
Fluttering shearwater	*Puffinus gavia*	At Risk: Relict	Least Concern
Sooty shearwater	*Puffinus griseus*	At Risk: Declining	Near Threatened
Hutton's shearwater	*Puffinus huttoni*	Threatened: Nationally Vulnerable	Endangered
Wedge-tailed shearwater	*Puffinus pacificus*	At Risk: Relict	Least Concern
Short-tailed shearwater	*Puffinus tenuirostris*	Non-Resident Native: Migrant	Least Concern
Petrels and prions			
Southern Cape petrel	*Daption capense capense*	Non-Resident Native: Migrant	#Least Concern
Snares Cape petrel	*Daption capense australe*	At Risk: Naturally Uncommon	#Least Concern
White-bellied storm petrel	*Fregetta grallaria grallaria*	Threatened: Nationally Endangered	#Least Concern
New Zealand storm petrel	*Fregetta maoriana*	Threatened: Nationally Vulnerable	Critically Endangered
Black-bellied storm petrel	*Fregetta tropica*	Not Threatened	Least Concern
Antarctic fulmar	*Fulmarus glacialoides*	Non-Resident Native: Migrant	Least Concern
Grey-backed storm petrel	*Garrodia nereis*	At Risk: Relict	Least Concern
Blue petrel	*Halobaena caerulea*	Non-Resident Native: Migrant	Least Concern

Common name	Scientific name	DOC category	IUCN category
Kerguelen petrel	*Lugensa brevirostris*	Non-Resident Native: Migrant	Least Concern
Southern giant petrel	*Macronectes giganteus*	Non-Resident Native: Migrant	Least Concern
Northern giant petrel	*Macronectes halli*	At Risk: Recovering	Least Concern
Wilson's storm petrel	*Oceanites oceanicus*	Non-Resident Native: Migrant	Least Concern
Fulmar prion	*Pachyptila crassirostris crassirostris*	At Risk: Naturally Uncommon	#Least Concern
Lesser fulmar prion	*Pachyptila crassirostris flemingi*	At Risk: Naturally Uncommon	#Least Concern
Chatham fulmar prion	*Pachyptila crassirostris pyramidalis*	At Risk: Naturally Uncommon	#Least Concern
Thin-billed prion	*Pachyptila belcheri*	Non-Resident Native: Migrant	Least Concern
Antarctic prion	*Pachyptila desolata*	At Risk: Naturally Uncommon	Least Concern
Salvin's prion	*Pachyptila salvini*	Non-Resident Native: Migrant	Least Concern
Fairy prion	*Pachyptila turtur*	At Risk: Relict	Least Concern
Broad-billed prion	*Pachyptila vittata*	At Risk: Relict	Least Concern
South Georgian diving petrel	*Pelecanoides georgicus* †	Threatened: Nationally Critical	Least Concern
Southern diving petrel	*Pelecanoides urinatrix chathamensis*	At Risk: Relict	#Least Concern
Subantarctic diving petrel	*Pelecanoides urinatrix exsul*	Not Threatened	#Least Concern
Northern diving petrel	*Pelecanoides urinatrix urinatrix*	At Risk: Relict	#Least Concern
Kermadec storm petrel	*Pelagodroma albiclunis*	Threatened: Nationally Critical	–
New Zealand white-faced storm petrel	*Pelagodroma marina maoriana*	At Risk: Relict	#Least Concern
White-chinned petrel	*Procellaria aequinoctialis*	Not Threatened	Vulnerable
Grey petrel	*Procellaria cinerea*	At Risk: Naturally Uncommon	Near Threatened
Black petrel	*Procellaria parkinsoni*	Threatened: Nationally	Vulnerable
Westland petrel	*Procellaria westlandica*	At Risk: Naturally Uncommon	Endangered
Chatham petrel	*Pterodroma axillaris*	Threatened: Nationally Vulnerable	Vulnerable
White-naped petrel	*Pterodroma cervicalis*	At Risk: Relict	Vulnerable
Cook's petrel	*Pterodroma cookii*	At Risk: Relict	Vulnerable
Mottled petrel	*Pterodroma inexpectata*	At Risk: Relict	Near Threatened
White-headed petrel	*Pterodroma lessonii*	Not Threatened	Least Concern
Grey-faced petrel	*Pterodroma macroptera gouldi*	Not Threatened	Least Concern
Chatham Island taiko	*Pterodroma magentae*	Threatened: Nationally Critical	Critically Endangered
Soft-plumaged petrel	*Pterodroma mollis*	At Risk: Naturally Uncommon	Least Concern
Kermadec petrel	*Pterodroma neglecta "summer"* †	Threatened: Nationally Endangered	#Least Concern
Kermadec petrel	*Pterodroma neglecta "winter"* †	At Risk: Naturally Uncommon	#Least Concern

Common name	Scientific name	DOC category	IUCN category
Black-winged petrel	*Pterodroma nigripennis*	Not Threatened	Least Concern
Pycroft's petrel	*Pterodroma pycrofti*	At Risk: Recovering	Vulnerable
Penguins			
Eastern rockhopper penguin	*Eudyptes filholi*	Threatened: Nationally Vulnerable	#Vulnerable
Fiordland crested penguin	*Eudyptes pachyrhynchus*	Threatened: Nationally Vulnerable	Vulnerable
Snares crested penguin	*Eudyptes robustus*	At Risk: Naturally Uncommon	Vulnerable
Erect-crested penguin	*Eudyptes sclateri*	At Risk: Declining	Endangered
White-flippered blue penguin**	*Eudyptula minor albosignata***	At Risk: Declining	#Least Concern
Chatham Island blue penguin**	*Eudyptula minor chathamensis***	At Risk: Naturally Uncommon	#Least Concern
Northern blue penguin**	*Eudyptula minor iredalei***	At Risk: Declining	#Least Concern
Southern blue penguin**	*Eudyptula minor minor***	At Risk: Declining	#Least Concern
Australian little penguin	*Eudyptula novaehollandiae*	At Risk: Recovering	-
Yellow-eyed penguin	*Megadyptes antipodes*	Threatened: Nationally Endangered	Endangered
Shags			
Campbell Island shag	*Leococarbo campbelli*	At Risk: Naturally Uncommon	Vulnerable
King shag	*Leucocarbo carunculatus*	Threatened: Nationally Endangered	Vulnerable
Otago shag	*Leucocarbo chalconotus*	At Risk: Recovering	Vulnerable
Auckland Island shag	*Leucocarbo colensoi*	Threatened: Nationally Vulnerable	Vulnerable
Chatham Island shag	*Leucocarbo onslowi*	Threatened: Nationally Critical	Critically Endangered
Bounty Island shag	*Leucocarbo ranfurlyi*	At Risk: Naturally Uncommon	Vulnerable
Foveaux shag	*Leucocarbo stewarti*	Threatened: Nationally Vulnerable	-
Black shag	*Phalacrocorax carbo novaehollandiae*	At Risk: Naturally Uncommon	#Least Concern
Little shag	*Phalacrocorax melanoleucos brevirostris*	Not Threatened	#Least Concern
Little black shag	*Phalacrocorax sulcirostris*	At Risk: Naturally Uncommon	Least Concern
Pied shag	*Phalacrocorax varius varius*	At Risk: Recovering	#Least Concern
Pitt Island shag	*Stictocarbo featherstoni*	Threatened: Nationally Critical	Endangered
Blue shag	*Stictocarbo punctatus oliveri*	At Risk: Naturally Uncommon	#Least Concern
Spotted shag	*Stictocarbo punctatus punctatus*	Not Threatened	#Least Concern
Gulls and terns			

Common name	Scientific name	DOC category	IUCN category
White-capped noddy	*Anous minutus minutus*	At Risk: Naturally Uncommon	#Least Concern
Common noddy	*Anous stolidus pileatus*	Non-Resident Native: Coloniser	#Least Concern
Black-fronted tern	*Childonias albostriatus*	Threatened: Nationally Endangered	Endangered
White-winged black tern	*Childonias leucopterus*	Non-Resident Native: Migrant	Least Concern
White tern	*Gygis alba candida*	Threatened: Nationally Critical	#Least Concern
Caspian tern	*Hydroprogne caspia*	Threatened: Nationally Vulnerable	Least Concern
Black-billed gull	*Larus bulleri*	Threatened: Nationally Critical	Endangered
Southern black-backed gull	*Larus dominicanus dominicanus*	Not Threatened	#Least Concern
Red-billed gull	*Larus novaehollandiae scopulinus*	At Risk: Declining	Least Concern
Sooty tern	*Onychoprion fuscatus serratus*	At Risk: Recovering	#Least Concern
Grey ternlet	*Procelsterna cerulea albivittata*	At Risk: Naturally Uncommon	#Least Concern
New Zealand fairy tern	*Sternula nereis davisae*	Threatened: Nationally Critical	#Vulnerable
Arctic tern	*Sterna paradisaea*	Non-Resident Native: Migrant	Least Concern
Southern white-fronted tern***	*Sterna striata aucklandorna****	Threatened: Nationally Vulnerable	#Least Concern
White-fronted tern***	*Sterna striata striata****	At Risk: Declining	#Least Concern
New Zealand Antarctic tern	*Sterna vittata bethunei*	At Risk: Recovering	#Least Concern
Eastern little tern	*Sternula albifrons sinensis*	Non-Resident Native: Migrant	#Least Concern
Skuas			
Brown skua	*Catharacta antarctica lonnbergi*	At Risk: Naturally Uncommon	#Least Concern
South Polar skua	*Catharacta maccormicki*	Non-Resident Native: Migrant	Least Concern
Pomarine skua	*Coprotheres pomarinus*	Non-Resident Native: Migrant	Least Concern
Long-tailed skua	*Stercorarius longicaudus*	Non-Resident Native: Migrant	Least Concern
Arctic skua	*Stercorarius parasiticus*	Non-Resident Native: Migrant	Least Concern
Gannets and tropicbird			
Australasian gannet	*Morus serrator*	Not Threatened	Least Concern
Red-tailed tropicbird	*Phaethon rubricauda*	At Risk: Recovering	Least Concern
Masked booby	*Sula dactylatra tasmani*	Threatened: Nationally Endangered	#Least Concern

- * OSNZ (2010) classify New Zealand white-capped albatross as a subspecies *Thalassarche cauta steadi*. Full species status is used here following ACAP.
- ** OSNZ (2010) classify a single species, little penguin *Eudyptula minor*. Multiple taxa are included here to reflect classification in the New Zealand Threat Classification Scheme.

- *** OSNZ (2010) classify a single species, white-fronted tern *Sterna striata*. Multiple taxa are included here to reflect classification in the New Zealand Threat Classification Scheme.
- # indicates that the International Union for the Conservation of Nature (IUCN) classification is based on a broader definition of the species than listed in this table.
- † Taxonomically Indeterminate in the New Zealand Threat Classification Scheme

8.2 BIOLOGY

Taylor (2000) provided an excellent summary of the characteristics, ecology, and life history traits of seabirds, defined for the purpose of this document by the list in (Table 8.1) which is further summarised here. Table 8.1 includes all seabird taxa listed by Robertson et al. (2017) – the most recent iteration of the NZTCS as applied to birds - with the exception of those seabird taxa classified as 'vagrant'. For both the New Zealand Threat Classification System, and the IUCN 'Red List' system, seabird taxa with relatively small populations, and/or which have declining populations, and/or which occupy small spatial extents tend to be categorised with a higher conservation status. The New Zealand system includes additional categories not found in the Red List classifications. Specifically, the 'threatened' category is sub-divided into 'nationally critical, 'nationally endangered' and 'nationally vulnerable' categories (analogous to the Red List's 'critically endangered' 'endangered' and 'vulnerable' categories). Additionally, the New Zealand system has four 'at risk' categories: 'declining', 'recovering', 'relict' and 'naturally uncommon'. The New Zealand system also recognises 'migrant', 'vagrant' and 'coloniser' categories, in addition to a 'not threatened' category, which is similar to the Red List's 'near threatened' category.

All seabirds spend part of their lifecycle feeding over the open sea. They have webbed feet, water-resistant feathering to enable them to fully immerse in salt water, and powerful wings or flippers. All have bills with sharp hooks, points, or filters, which enable them to catch fish, cephalopods, crustaceans and plankton. Seabirds can drink saltwater and have physiological adaptations to remove excess salt.

Most seabird taxa are relatively long-lived; most live to 20 years and 30–40 years is typical for the oldest individuals. A few groups, notably albatrosses, can live for 50–60 years. Most taxa have relatively late sexual maturity. Red-billed gull *Larus novaehollandiae scopulinus* and blue penguin *Eudyptula minor* have been recorded nesting as yearlings and diving petrels *Pelecanoides urinatrix* and yellow-eyed penguin *Megadyptes antipodes* can begin as 2-year-olds, but most seabirds start nesting only at age 3–6 years, and some albatross and petrel taxa delay nesting until 8–15 years old. In these late developers, individuals first return to colonies at 2–6 years old. Richard et al. (2011) listed values for several demographic parameters that they used for a comprehensive seabird risk assessment. Most seabirds, and especially albatrosses and some petrels, usually return to the breeding colony where they were reared, or nest close-by. Seabirds also have a tendency to mate for long periods with the same partner, and albatross pairs almost always remain together unless one partner dies.

The number of eggs laid varies among families. Albatrosses and petrels lay only one egg per year (sometimes nesting every other year) and do not lay again that year if it is lost. Other taxa such as gannets lay one egg but can replace it if the egg is lost. Most penguins lay two eggs but some raise only one chick and eject the second egg; replacement laying is uncommon. Blue penguins, gulls and terns lay 1–3 eggs and can lay up to three clutches in a year if eggs are damaged or lost. Shags lay 2–5 eggs, can replace clutches, and have several breeding seasons in a year. Incubation in albatrosses and petrels lasts 40–75 days and chick rearing 50–280 days. In gulls and terns, incubation is completed in 20–25 days and chicks fledge in 20–40 days. In general, the lower the potential reproductive output of a taxon, the higher the adult survival rates and longevity.

Some seabirds such as shags, blue penguin and yellow-eyed penguin live their lives and forage relatively close to where they breed, but many, including most albatrosses and petrels, spend large parts of their lives in international waters or in the waters of other nations far from their breeding locations. They can travel great distances across oceans during foraging flights and migratory journeys.

8.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Fishing-related mortality of seabirds has been recognised as a serious, worldwide issue for only about 20 years (Bartle 1991, Brothers 1991, Brothers et al. 1999, Croxall 2008) and the Food and Agriculture Organization of the United Nations (FAO) released its International Plan of Action for reducing incidental catch of seabirds in longline fisheries (IPOA-seabirds) in 1999 (FAO 1999). The IPOA-seabirds called on countries with (longline) fisheries that interact with seabirds to assess their fisheries to determine if a problem exists and, if so, to develop national plans (NPOA-seabirds) to reduce the incidental seabird catch in their fisheries. Lewison et al. (2004) noted that, in spite of the recognition of the problem, few comprehensive assessments of the effects of fishing-related mortality had been conducted in the decade or so after the problem was recognised. They reasoned that: many vulnerable species live in pelagic habitats, making surveys logistically complex and expensive; capture data are sparse; and understanding of the potential for affected populations to sustain additional mortality is poor. Soykan et al. (2008) identified similar questions in a Theme Section published in *Endangered Species Research*, including: Where is bycatch most prevalent? Which species are taken as bycatch? Which fisheries and gear types result in the highest bycatch of marine megafauna? What are the population-level effects on bycatch species? How can bycatch be reduced?

There has been substantial progress on these questions since 2004. Croxall et al. (2012) reviewed the threats to 346 seabird taxa and concluded that: seabirds are more threatened than other comparable groups of birds; their status has deteriorated faster over recent decades; and fishing-related mortality is the most pervasive and immediate threat to many albatrosses and petrels. They listed the principal threats while at sea as being posed by commercial fisheries (through competition for food and mortality associated with fishing gear) and pollution, and those on land as being alien predators, habitat degradation and human disturbance. Direct exploitation, impacts of aquaculture, energy generation operations, and climate change were listed as threats for some taxa or areas where understanding was particularly poor.

Croxall et al. (2012) categorised responses to the issue of fishing-related mortality as:

- using long-term demographic studies of relevant seabird species, linked to observational and recovery data to identify the cause of population declines (e.g., Croxall et al. 1998, Tuck et al. 2004, Poncet et al. 2006);
- risk assessments, based on spatio-temporal overlap between seabird species susceptible to bycatch and effort data for fisheries likely to catch them (e.g., Waugh et al. 2008b, Filippi et al. 2010, Tuck et al. 2011);
- working with multinational and international bodies (e.g., FAO and RFMOs) to develop and implement appropriate regulations for the use of best-practice techniques to reduce or eliminate seabird bycatch and;
- working with fishers (and national fishery organisations) to assist cost-effective implementation of these mitigation techniques.

Seabirds are ranked by the IUCN as the world's most threatened bird grouping (Croxall et al. 2012). Globally they face a number of threats to their long-term viability, both at their breeding sites and while foraging at sea. Work at the global level on reducing threats at breeding sites is a major focus of ACAP, for which DOC is the lead New Zealand agency. However, a key threat to seabirds at sea, especially albatrosses and petrels, is incidental capture and death in fisheries (Croxall et al. 2012), which in New Zealand is managed by Fisheries New Zealand.

Some seabirds do not range far from their breeding or roosting sites and incidental captures of these taxa can be managed by a single jurisdiction. Conversely, conservation of highly migratory taxa such as albatrosses and petrels cannot be achieved by one country acting independently of other nations that share the same populations. Because of this, in recent years countries that share populations of threatened seabirds have sought to take action on an international level (e.g., ACAP) to complement policy and actions taken within their own jurisdictions.

The ICES Working Group on Seabird Ecology agreed (WGSE 2011) that the three most important indirect effects of fisheries on seabird populations were: the harvesting of seabird food; discards as food subsidies; and modification of marine habitats by dredges and trawls. Many seabird

prey species are fished commercially (e.g., Furness 2003) or can be impacted indirectly by fishing of larger predators. These relationships are complex and poorly understood but WGSE (2011) agreed that impacts on populations of seabirds were inevitable. Fishery discards and offal have the potential to benefit seabird species, especially those that ordinarily scavenge (Furness et al. 1992, Wagner & Boersma 2011). However, discarding can also modify the way in which birds forage for food (e.g., Bartumeus et al. 2010, Louzao et al. 2011), sometimes with farther-reaching behavioural consequences with negative as well as positive Boersma 2011). Dredging and bottom trawling both affect benthic habitat and fauna (see Rice 2006 and the benthic effects chapter in this document) and WGSE (2011) agreed that this probably affects some seabird populations, although little work has been done in this area.

8.4 STATE OF KNOWLEDGE IN NEW ZEALAND

Before the arrival of humans, the absence of terrestrial mammalian predators in New Zealand made it a relatively safe breeding place for seabirds and large numbers of a wide variety of taxa bred here, including substantial numbers on the main North and South islands. Today, New

Some seabirds use New Zealand waters but do not breed here. Some visit here occasionally to feed (e.g., wandering albatross *Diomedea exulans* and southern giant petrel *Macronectes giganteus*), whereas others are frequent visitors (e.g., short-tailed shearwater *Puffinus tenuirostris* and Wilson's storm petrel *Oceanites oceanicus*), sometimes for extended durations (e.g., Arctic skua *Stercorarius parasiticus*).

Taylor (2000) listed a wide range of threats to New Zealand seabird taxa including introduced mammals, avian predators (e.g., weka), disease, fire, weeds, loss of nesting habitat, competition for nest sites, coastal development, human disturbance, commercial and cultural harvesting, volcanic eruptions, pollution, plastics and marine debris, oil spills and exploration, heavy metals or chemical contaminants, global sea temperature changes, marine biotoxins, and fisheries interactions. Seabirds are caught in commercial trawl, longline, set-net, and, occasionally, other fisheries (e.g., annual assessments by SJ Baird from 1994 to 2005, Baird & Smith 2008, Waugh et al. 2008a, 2008b, Abraham et al. 2010b, 2016) as well as in non-commercial fisheries (Abraham et al. 2010a). New Zealand released its

effects (including the 'junk food hypothesis', e.g., Romano et al. 2006, Grémillet et al. 2008). Louzao et al. (2011) stated that discards can affect movement patterns (Arcos & Oro 1996), improve reproductive performance (Oro et al. 1997, 1999) and increase survival (Oro & Furness 2002, Oro et al. 2004). Benefits for scavengers and kleptoparasitic taxa (those that obtain food by stealing from other animals) feeding on discards can also have consequent negative impacts on other species, especially diving species, that share breeding sites or are subject to displacement (Wagner &

Zealand's extensive coastline, numerous inshore and offshore islands (many of them predator free) and surrounding seas and oceans continue to make it an important foraging and breeding ground for about 168 seabird taxa, second only to the USA (GA Taylor, Department of Conservation, personal communication). Roughly 99 of these taxa breed in New Zealand (Figure 8.1 and Figure 8.2, Table 8.2), including the greatest number of albatrosses (14), petrels (32), shags (13) and penguins (9) of any area in the world (Miskelly et al. 2008). More than a third are endemic (i.e., breed nowhere else in the world), giving New Zealand by far the largest number of endemic seabird taxa in the world.

first National Plan of Action to reduce the incidental catch of seabirds (NPOA-seabirds) in 2004 and this was revised in 2013. This stated that there was, at that time, limited information about the level of incidental catch and population characteristics of different seabird taxa, and that this made quantifying the overall impact of fishing difficult. This situation had improved somewhat by the time 2013 NPOA-seabirds was published but, nevertheless, that document seeks to ensure, among other things, that the development of new mitigation measures, new observation and monitoring methods, and relevant research are encouraged and resourced. Seabird taxa caught in New Zealand fisheries range in IUCN threat ranking from critically endangered (e.g., Chatham Island shag *Leucocarbo onslowi*), to least concern (e.g., common diving petrel) (e.g., Vié et al. 2009).

Different taxa and populations face different threats from fishing operations depending on their biological characteristics and foraging behaviours. Biological traits such as diving ability, agility, size, sense of smell, eyesight and diet, foraging factors such as the season and areas they forage, their aggressiveness, the boldness (or shyness) they

display in their attraction to fishing activity can all affect their susceptibility to capture, injury, or death from fishing operations. Some fishing methods pose particular threats to some guilds or types of seabirds. For example, penguins are particularly vulnerable to set-net operations and large albatrosses appear to be vulnerable to most forms of longlining. The nature and extent of interactions differs Buller's albatross *Thalassarche bulleri bulleri* (115), sooty shearwater (63), Salvin's albatross *Thalassarche salvini* (41), flesh-footed shearwater *Puffinus carneipes* (21), black petrel *Procellaria parkinsoni* (20), Westland petrel *Procellaria westlandica* (15), broad-billed prion *Pachyptila vittata* (10), little penguin (8), grey petrel *Procellaria cinerea* (5) and Antipodean albatross *Diomedea antipodensis* spatially, temporally, seasonally and diurnally between sectors, fisheries and between fleets and vessels within fisheries. In 2015–16 the taxa most frequently observed caught in New Zealand commercial fisheries in descending order were common diving petrel (288), white-chinned petrel *Procellaria aequinoctialis* (252), New Zealand white-capped albatross *Thalassarche steadi* (156), southern *antipodensis* (4). It should be noted that some of these totals included birds not technically 'captured' (see definition in section 8.1) but which were deck strikes (notably common diving petrel) and which were released alive: for example, all but two of the 288 common diving petrels were released alive.

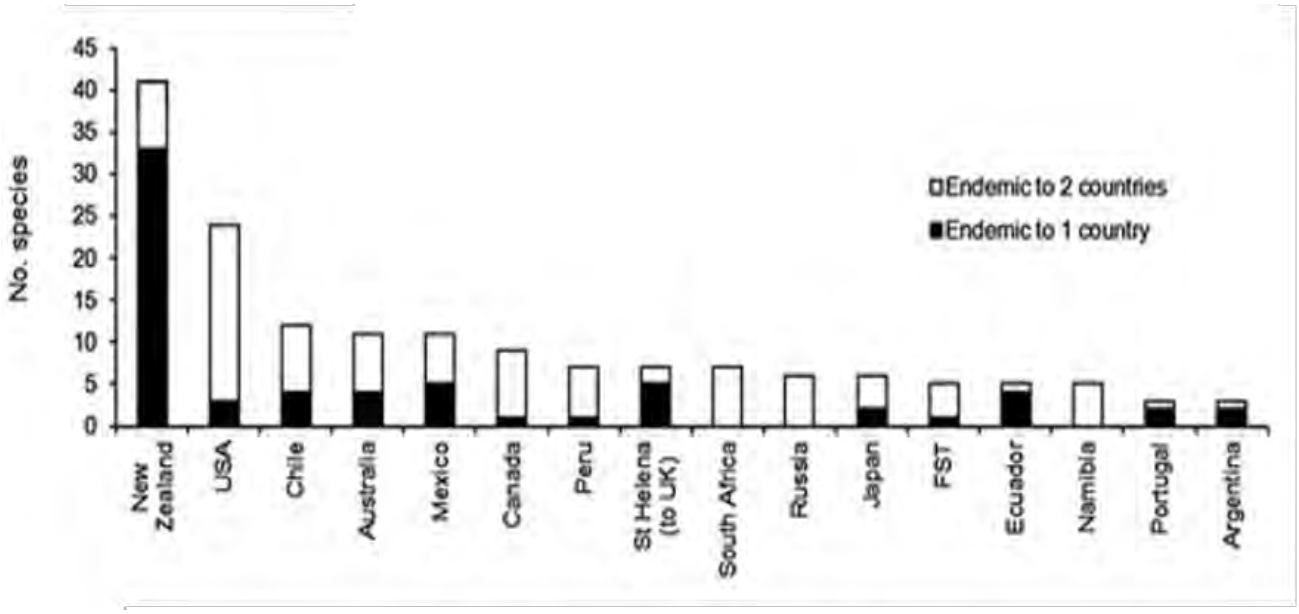


Figure 8.1: (from Croxall et al. 2012). Number of endemic breeding seabird taxa by country.

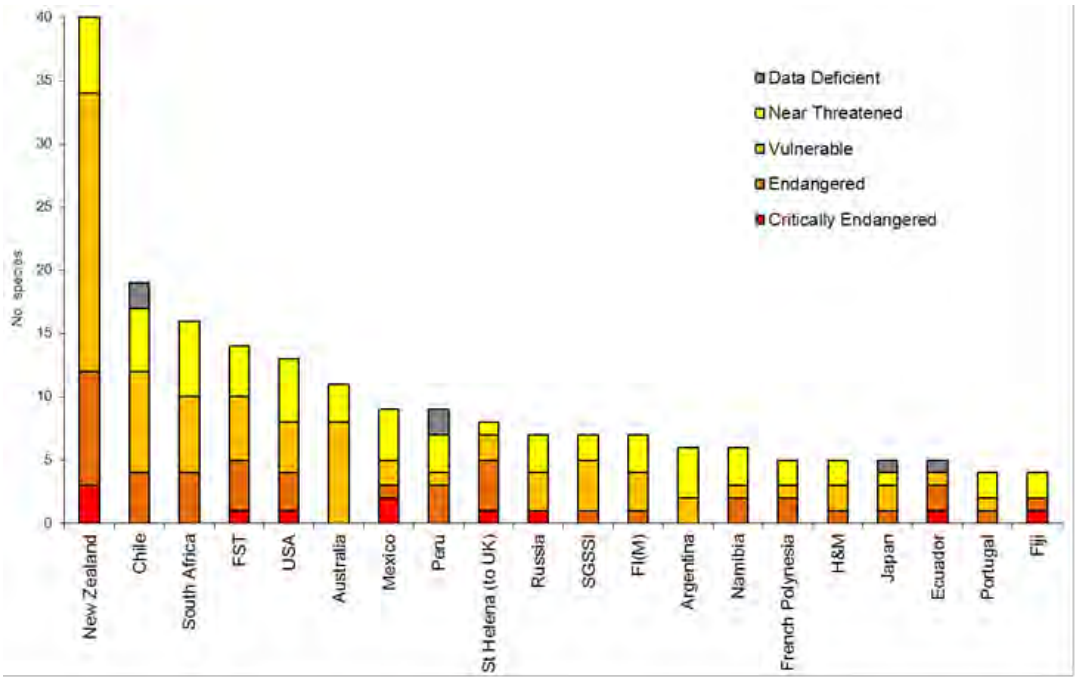


Figure 8.2: (from Croxall et al. 2012, supplementary material). The number of breeding and resident seabird species by country in each IUCN category (excluding Least Concern). FST, French Southern Territories; SGSSI, South Georgia and South Sandwich Islands; FI(M), Falkland Islands (Malvinas); H&M, Heard Island and McDonald Islands.

Table 8.2: (from Taylor 2000). Number of species (spp.) and taxa of seabirds of different families in New Zealand and worldwide in 2000. Additional taxa may have been recorded since.

Family	Common name	World breeding		NZ breeding		NZ visitors, vagrants	
		N spp.	N taxa	N spp.	N taxa	N spp.	N taxa
Spheniscidae	Penguins	17	26	6	10	8	10
Gaviidae	Divers, loons	4	6	–	–	–	–
Podicipedidae	Grebes	10	20	2	2	–	–
Diomedeidae	Albatrosses	24	24	13	13	7	7
Procellariidae	Petrels, shearwaters	70	109	28	31	20	23
Hydrobatidae	Storm-petrels	20	36	4	5	2	3
Pelecanoididae	Diving petrels	4	9	2	4	–	–
Phaethontidae	Tropicbirds	3	12	1	1	1	1
Pelecanidae	Pelicans	7	12	–	–	1	1
Sulidae	Gannets	9	19	2	2	1	1
Phalacrocoracidae	Shags	39	57	12	13	–	–
Fregatidae	Frigatebirds	5	11	–	–	2	2
Anatidae	Marine ducks	18	27	–	–	–	–
Scolopacidae	Phalaropes	2	2	–	–	2	2
Chionididae	Sheathbills	2	5	–	–	–	–
Stercorariidae	Skuas	7	10	1	1	4	4
Laridae	Gulls	51	78	3	3	–	–
Sternidae	Terns, noddies	43	121	10	11	8	8
Rynchopidae	Skimmers	2	4	–	–	–	–
Alcidae	Auks, puffins	22	45	–	–	–	–
Total		359	633	84	96	56	62

The management of fisheries to ensure the long-term viability of seabird populations requires an understanding of the risks posed by fishing and other anthropogenic drivers. Several studies have already estimated the number

of seabirds caught annually within the New Zealand Exclusive Economic Zone (EEZ) in a range of fisheries (e.g., Baird & Smith 2008, Waugh et al. 2008a, 2008b, Abraham et al. 2010b, 2016). Seabirds that breed in New Zealand die as a result of interactions with commercial or recreational fishing operations in waters under New Zealand jurisdiction, through interactions with New Zealand vessels or other nations' vessels on the High Seas and through interactions with commercial, recreational or artisanal fishing operations in waters under the jurisdiction of other states.

In order to evaluate whether the viability of seabird populations is jeopardised by incidental mortality from commercial fishing, the number of annual fatalities needs to be compared with the capacity of the populations to replace those losses; this depends on the size and productivity of each population. Sufficient data to build fully quantitative population models to assess risks and explore the likely results of different management approaches are available for only very few taxa (e.g., Fletcher et al. 2008, Francis & Bell 2010, Francis et al. 2008, Dillingham & Fletcher 2011). For this reason, seabird risk assessments are generally made using only expert knowledge (Level 1) or when available, semi-quantitative using a mix of expert knowledge and data (Level 2) (Hobday et al. 2007). Rowe (2013) described a Level 1 seabird risk assessment and Baird et al. (2006, updated by Baird & Gilbert 2010) described a semi-quantitative assessment for seabird taxa for which reasonable numbers of observed captures were available. These assessments could not be used directly to quantify risk for all seabird taxa and fisheries. More comprehensive and quantitative Level 2 risk assessments have since been conducted and are described in more detail in Section 8.4.2.3, *Seabird Spatially Explicit Fisheries Risk Assessment*. Furthermore, observer programmes overseas are enabling quantitative Level 2 risk assessments to be carried out over larger spatial scales. For example, current work supported by MPI aims to develop a Level 2 risk assessment for a selection of New Zealand seabirds across the entire southern hemisphere.

8.4.1 QUANTIFYING FISHERIES INTERACTIONS

Onboard captures recorded by observers represent the most reliable source of information for monitoring trends in total captures and capture rates, but these data have three main deficiencies with respect to estimating total fatalities, especially to species level. First, some captured seabirds are released alive (31% in trawl fisheries between 2002–03 and 2015–16, 26% in surface-longline fisheries, and 20% in bottom-longline fisheries), meaning that, all else being equal, estimates of captures may overestimate total fatalities, depending on the survival rate of those released. There is a trend in the percentage of albatross observed caught on trawl vessels that were released alive with a general increase from 2009–10; this trend is less apparent across all birds or in other methods (Table 8.4). Second, identifications by observers are not completely reliable and sometimes use generic codes rather than species codes. From 2002–03 to 2015–16, 68% of all observed seabird captures have either been returned for necropsy or photographs taken for confirmation of identification. As a result of the expert review, 22.4% of the species identifications made by observers were changed (Thompson et al. in prep). Third, not all birds killed or mortally wounded by fishing gear are recovered on a fishing vessel. Some birds caught on longline hooks fall off before being recovered, and birds that collide with trawl warps may be dragged under the water and drowned or injured to the extent that they are unable to fly or feed. Excluding this 'cryptic' mortality means that, all else being equal, estimates of captures will underestimate total fatalities, and the extent of underestimation will vary among taxa and fisheries. These deficiencies do not greatly affect the suitability of estimates of captures and capture rates for monitoring purposes, but they have necessitated the development of alternative methods for assessing risk and population consequences.

Information with which to characterise seabird interactions with fisheries comes from a variety of sources. Some is opportunistically collected, whilst other information collection is targeted at specifically describing the nature and extent of seabird captures in fisheries. This section is focused on the targeted information collection.

Many New Zealand commercial fisheries have MPI observer coverage, some of which is funded by DOC's CSP programme (e.g., Rowe 2009, 2010, Ramm 2011, 2012). Observers collect independent data on the number of captures of seabirds, the number of fishing events observed, and at-sea identification of the seabirds for these fisheries. Commercial fishers are legally required to provide effort data allowing estimation of the total number of fishing events in a fishery. In combination these data have

been used for many years to assess the nature and extent of seabird captures in fisheries (e.g., Abraham et al. 2010b, Abraham & Thompson 2009a, 2010, 2011a, 2011b, Ayers et al. 2004, Baird 1994, 1995, 1996, 1997, 1999, 2000, 2001a, 2001b, 2003, 2004a–c, 2005, Baird et al. 1998, 1999, Baird & Griggs 2004, Thompson & Abraham 2009). In this context, 'captures' include all seabirds observed by an observer to be brought onboard a fishing vessel, whether reported as live or dead, but exclude non-fishing-related events (e.g., birds striking the superstructure and landing on deck) and decomposed carcasses. Specimens and photographs (especially for birds released alive) are also collected allowing verification of at-sea identifications (from carcasses, tissue samples or photographs) and description of biological characters (sex, age, condition, etc., available only from carcasses).

Table 8.4: Percentage of observed captures that were released alive (http://data.dragonfly.co.nz/psc. Data version v2018001.)

	All birds			Albatross spp. only		
	Trawl	SLL	BLL	Trawl	SLL	BLL
2002–03	25	18	9	8	28	11
2003–04	9	30	32	4	31	80
2004–05	18	41	33	11	48	0
2005–06	18	38	49	7	40	43
2006–07	18	22	12	11	23	0
2007–08	19	38	10	15	38	30
2008–09	27	26	36	20	34	50
2009–10	36	30	54	30	32	0
2010–11	30	45	45	35	51	100
2011–12	24	16	70	23	18	88
2012–13	38	26	0	33	27	0
2013–14	39	25	14	25	26	10
2014–15	51	42	15	47	47	100
2015–16	29	10	8	29	9	20
2016-17	25	22	32	36	32	0

In some fisheries, observer data are temporally and spatially well stratified, whilst in others data are only available from a spatially select part of the fishery, or a limited part of the year. Where sufficient observer data are available, estimates of total seabird captures in the fishery are calculated. The methods currently used in estimating seabird captures in New Zealand fisheries are described in Abraham et al. (2016). In this context, captures include all seabirds recovered on a fishing vessel except birds that simply land on the deck or collide with a vessel's superstructure, decomposing animals, records of tissue fragments, and birds caught during trips carried out under special permit (e.g., for trials of mitigation methods). See section 8.1 for a definition of 'seabird capture'. Observer coverage has been variable in that some fisheries and areas have had much higher coverage than others. This complicates estimation of the total number of seabirds captured, especially when estimates include more than one fishery, because the distribution of birds and captures is also heterogeneous (Figure 8.3).

Fisher-reported captures (on NFPSCR forms available since 1 October 2008) have not been used to estimate total captures because the reported capture rates vary by fishery and the species identification can be less certain.

Abraham et al. (2016) made model-based estimates of captures in New Zealand trawl and longline fisheries for the following taxa or groups: sooty shearwater; white-chinned petrel; black petrel; grey petrel; white-capped albatross; Salvin's albatross; southern Buller's albatross; other albatrosses; and all other birds. These individual species were chosen because they are the most frequently caught in trawl and longline fisheries. Captures of other albatrosses are mostly Gibson's *Diomedea antipodensis gibsonii* or Antipodean albatrosses or Campbell Island albatross *Thalassarche impavida*. The 'other birds' category includes many taxa but grey, black, grey-faced *Pterodroma macroptera gouldi* , and Cape petrels (both subspecies but mostly southern Cape petrels, *Daption capense capense*), flesh-footed shearwater, and spotted shag *Stictocarbo punctatus punctatus* are relatively common observed captures (the latter based on few observations that included 31 captures in one event). Estimated captures up to and including the 2016–17 year are shown in Table 8.5 to Table 8.13.

Observed captures of seabirds in trawl fisheries were most common off both coasts of the South Island, along the Chatham Rise, on the fringes of the Stewart-Snares shelf, and around the Auckland Islands (Figure 8.4). This largely reflects the distribution of the major commercial fisheries for squid, hoki and middle-depth species, which have tended to have relatively high observer coverage. White-capped, Salvin's, and southern Buller's albatrosses have been the most frequently observed captured species of albatrosses, and white-chinned petrel and sooty shearwater have been the other species most frequently observed (Table 8.14). About 38% of observed captures were albatrosses.

Observed captures of seabirds in surface-longline fisheries were most common off the south-west coast of the South Island and the north-east coast of the North Island (Figure 8.5), again largely reflecting the distribution of the major commercial fisheries (for southern bluefin and other tunas). The charter fleet targeting tuna has historically had much higher observer coverage than the domestic fleet. Southern Buller's and white-capped albatrosses have been the most frequently observed captured species of albatrosses, and grey, white-chinned, and black petrels have been the other species most frequently observed (Table 8.15). About 80% of observed captures were albatrosses.

Observed captures of seabirds in bottom-longline fisheries were most common off the south coast of the South Island, along the Chatham Rise, scattered throughout the subantarctic, and off the north-east coast of the North Island, especially around the Hauraki Gulf (Figure 8.6). This distribution largely reflects the distribution of the ling and snapper longline fisheries that have received most observer coverage; other bottom-longline fisheries have had much less coverage. Salvin's and Chatham albatrosses have been the most frequently observed captured of the albatross species, and white-chinned petrel, flesh-footed shearwater, grey petrel, sooty shearwater, and black petrels have been the other species most frequently observed (Table 8.16). Only about 14% of observed captures were albatrosses.

Model-based estimates of captures can be compared across trawl and longline fisheries (Figure 8.7). Summed across all bird taxa, trawl, surface-longline and bottom-longline fisheries account for 41%, 17% and 42% of estimated captures, respectively, but there are substantial differences in these proportions among seabird taxa. A high proportion (81% between 2002–03 and 2016–17) of estimated total captures of white-capped albatross are from trawl fisheries with most of the remainder estimated from the surface-longline fisheries. The trawl fishery also accounts for 92% of the estimated captures of sooty shearwaters, with most of the remainder taken by bottom longliners. The proportion of estimated captures by trawl fisheries reduces to 12% for all other albatrosses (i.e., not including white-capped, Salvin's and Buller's albatrosses) combined, with 44% and 43% taken in surface- and bottom-longline fisheries, respectively. Bottom longline and trawl take similar proportions of the estimated total captures of white-chinned petrels (29% and 66%, respectively).

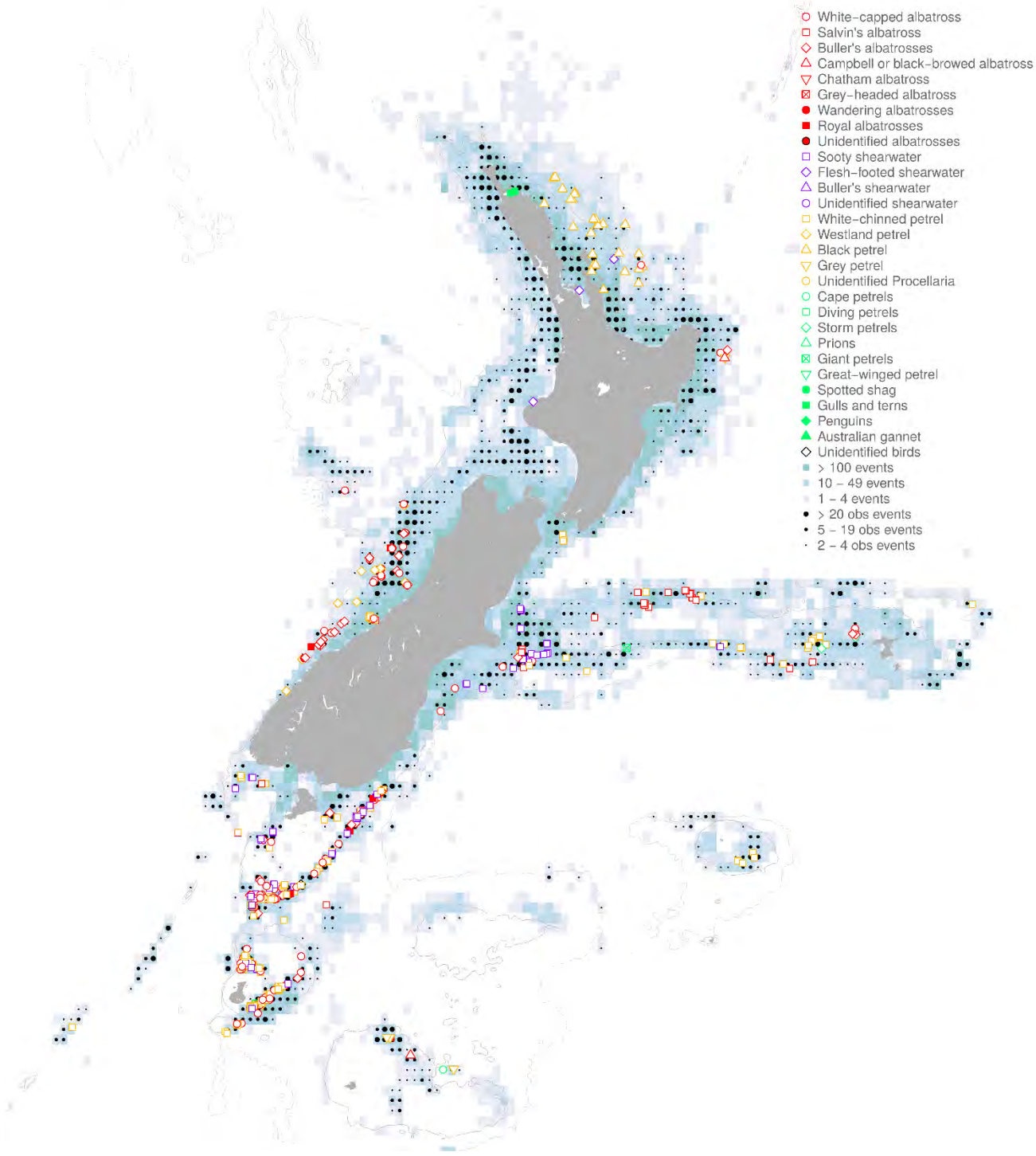


Figure 8.3: All observed seabird captures in trawl, surface-longline, bottom-longline, set-net and purse seine fishing within New Zealand region, between October 2016 and September 2017. The colour within each 0.2 degree cell indicates the number of fishing events (tows and sets, darker colours indicate more fishing) and the black dots indicate the number of observed fishing events (larger dots indicate more observations). The coloured symbols indicate the location of observed seabird captures, randomly jittered by 0.2 degrees. The 500 m and 100 m depth contours are shown. http://data.dragonfly.co.nz/psc Data version v2018001.

Table 8.5: Summary of model-estimated total captures of all seabirds combined by October fishing year in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries) (effort in tows), large (>28m) and small (<28m) surface-longline (effort in hooks) and large (>28m) and small (<28m) bottom-longline (effort in hooks) fisheries between 2007–08 and 2016–17. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed, capture rate, observed capture rate. http://data.dragonfly.co.nz/psc. Data version v2018001.

Year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All Effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						**Inshore trawl**				
2007-08	39 210	22.7	2.59	1 280	1 041-1 607	50 314	0.3	0.7	700	486-1 034
2008-09	35 927	22.4	5.07	1 568	1 314-1 920	51 625	3.2	3.09	741	535-1 050
2009-10	36 766	22.3	3.04	1 317	1 075-1 644	56 126	1.4	1.56	786	559-1 139
2010-11	35 522	18.9	5.42	1 589	1 321-1 958	50 563	1.4	0.14	689	490-977
2011-12	34 128	25.9	2.72	1 157	951-1 437	50 290	1	1.33	688	485-999
2012-13	33 321	36.5	5.75	1 433	1 260-1 682	50 517	0.4	0.45	708	500-1 014
2013-14	34 612	33.4	4.07	1 209	1 040-1 432	50 498	3.2	1.05	727	519-1 021
2014-15	34 737	33.1	5.26	1 431	1 241-1 692	44 030	4.7	0.63	602	427-853
2015-16	34 304	32	4	1 135	973-1 377	43 726	4.6	1.05	617	437-891
2016-17	33 891	30.9	3.96	1 145	968-1 396	44 281	7.3	0.28	616	430-879
Large BLL						**Small BLL**				
2007-08	14 127 896	21.8	0.01	410	175-992	27 376 411	1.8	0.04	1 837	1 253-2 930
2008-09	12 861 501	24.9	0	312	101-828	24 573 964	3.6	0.04	1 730	1 202-2 705
2009-10	13 602 940	12.6	0.01	348	134-874	26 845 521	2.7	0.08	1 813	1 271-2 806
2010-11	12 919 517	11.8	0.01	332	137-815	27 981 339	1	0.01	1 997	1 407-3 081
2011-12	11 571 447	17.5	0	208	74-531	26 312 456	0.3	0.07	1 853	1 254-3 035
2012-13	8 234 145	3.3	0	259	109-588	24 271 654	1.9	0.01	1 619	1 104-2 602
2013-14	16 459 721	11.7	0.02	682	332-1 422	24 419 994	4.1	0.06	1 481	1 064-2 174
2014-15	14 060 072	2.5	0.03	482	210-1 124	25 289 849	2.1	0.03	1 383	982-2 092
2015-16	18 604 396	10.8	0.04	607	319-1 223	24 887 264	2.5	0.04	1 328	918-2 105
2016-17	22 157 051	17.6	0	567	255-1 217	24 396 916	4.5	0.04	1 279	912-1 922
Large SLL						**Small SLL**				
2007-08	568 285	50.2	0.08	42	28-72	1 678 054	8.1	0.1	472	325-702
2008-09	809 230	97.2	0.05	44	42-52	2 306 403	6.5	0.1	599	416-870
2009-10	478 558	100	0.12	56	56-56	2 516 706	7.3	0.43	725	547-1 004
2010-11	503 370	100	0.06	29	29-29	2 684 809	6.4	0.11	741	521-1 099
2011-12	551 440	100.6	0.06	33	33-33	2 548 687	6.8	0.18	758	547-1 109
2012-13	487 520	100	0.01	5	5-5	2 389 412	3.1	0.3	723	528-1 026
2013-14	653 330	100	0.02	16	16-16	1 896 434	6.8	0.15	608	435-881
2014-15	622 300	99.4	0.04	22	22-23	1 790 036	6	0.15	508	357-743
2015-16	54 450	43.6	1.14	44	30-76	2 302 691	13	0.35	730	549-1 017
2016-17	0			0		2 092 486	16.5	0.15	579	426-814

Table 8.6: Summary of model-estimated total captures of white-caped albatross combined by October fishing year in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries) (effort in tows), large (>28m) and small (<28m) surface-longline (effort in hooks) and large (>28m) and small (<28m) bottom-longline (effort in hooks) fisheries between 2007–08 and 2016–17. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed, capture rate, observed capture rate. http://data.dragonfly.co.nz/psc. Data version v2018001.

Year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All Effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						**Inshore trawl**				
2007-08	39 210	22.7	0.47	187	136-255	50 314	0.3	0	140	76-229
2008-09	35 927	22.4	1.02	286	218-365	51 625	3.2	0.42	145	85-226
2009-10	36 766	22.3	0.46	181	128-248	56 126	1.4	0.39	166	94-262
2010-11	35 522	18.9	0.61	196	140-270	50 563	1.4	0.14	154	87-246
2011-12	34 128	25.9	0.73	227	173-299	50 290	1	1.14	154	88-244
2012-13	33 321	36.5	1.03	213	179-259	50 517	0.4	0.45	165	94-267
2013-14	34 612	33.4	0.61	154	122-196	50 498	3.2	0.18	162	90-255
2014-15	34 737	33.1	0.65	150	122-189	44 030	4.7	0	136	76-218
2015-16	34 304	32	0.95	190	158-232	43 726	4.6	0.15	143	81-228
2016-17	33 891	30.9	0.75	168	136-209	44 281	7.3	0	141	78-226
Large BLL						**Small BLL**				
2007-08	14 123 096	21.8	0	1	0-5	27 376 411	1.8	0	29	8-62
2008-09	12 861 501	24.9	0	1	0-4	24 573 964	3.6	0	30	9-63
2009-10	13 607 740	12.6	0	1	0-4	26 845 521	2.7	0	26	7-56
2010-11	12 914 717	11.8	0	1	0-5	27 981 339	1	0	37	11-78
2011-12	11 560 277	17.5	0	1	0-3	26 312 456	0.3	0.02	33	12-67
2012-13	8 240 515	3.3	0	2	0-6	24 271 654	1.9	0	27	8-58
2013-14	16 448 081	11.7	0	3	0-9	24 419 994	4.1	0	29	8-61
2014-15	14 077 799	2.5	0	3	0-7	25 289 849	2.1	0	25	8-52
2015-16	18 603 012	10.8	0	3	0-8	24 887 264	2.5	0	26	7-55
2016-17	22 150 093	17.6	0	2	0-7	24 396 916	4.5	0	30	10-63
Large SLL						**Small SLL**				
2007-08	568 285	50.2	0.01	6	3-14	1 678 054	8.1	0.01	29	9-61
2008-09	809 230	97.2	0	2	2-3	2 306 403	6.5	0.01	36	13-74
2009-10	478 558	100	0.02	11	11-11	2 516 706	7.3	0.1	63	37-105
2010-11	503 370	100	0.01	4	4-4	2 684 809	6.4	0	43	17-84
2011-12	551 440	100.6	0.01	6	6-6	2 548 687	6.8	0.01	121	58-219
2012-13	487 520	100	0	2	2-2	2 389 412	3.1	0.14	116	61-205
2013-14	653 330	100	0	0	0-0	1 896 434	6.8	0.05	95	46-169
2014-15	622 300	99.4	0	3	3-3	1 790 036	6	0.04	86	40-158
2015-16	54 450	43.6	0.46	15	11-29	2 302 691	13	0.09	118	69-197
2016-17	0					2 092 486	16.5	0.05	105	57-180

Table 8.7: Summary of model-estimated total captures of Salvin's albatross combined by October fishing year in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries) (effort in tows), large (>28m) and small (<28m) surface-longline (effort in hooks) and large (>28m) and small (<28m) bottom-longline (effort in hooks) fisheries between 2007–08 and 2016–17. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed, capture rate, observed capture rate. http://data.dragonfly.co.nz/psc. Data version v2018001.

Year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All Effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						**Inshore trawl**				
2007-08	39 210	22.7	0.09	146	81-249	50 314	0.3	0.7	105	45-204
2008-09	35 927	22.4	0.33	195	122-308	51 625	3.2	0.55	119	55-222
2009-10	36 766	22.3	0.43	226	149-342	56 126	1.4	0.91	130	59-244
2010-11	35 522	18.9	0.3	210	125-343	50 563	1.4	0	118	50-229
2011-12	34 128	25.9	0.29	211	135-330	50 290	1	0	110	46-215
2012-13	33 321	36.5	0.44	244	167-364	50 517	0.4	0	122	50-234
2013-14	34 612	33.4	0.44	257	175-380	50 498	3.2	0	136	58-263
2014-15	34 737	33.1	0.4	283	188-418	44 030	4.7	0	108	45-204
2015-16	34 304	32	0.32	216	144-332	43 726	4.6	0	92	36-180
2016-17	33 891	30.9	0.24	184	115-302	44 281	7.3	0	103	45-198
Large BLL						**Small BLL**				
2007-08	14 123 096	21.8	0	36	2-152	27 376 411	1.8	0	113	23-380
2008-09	12 861 501	24.9	0	28	2-121	24 573 964	3.6	0	107	25-330
2009-10	13 607 740	12.6	0	38	1-179	26 845 521	2.7	0	113	24-353
2010-11	12 914 717	11.8	0	25	4-96	27 981 339	1	0	145	30-473
2011-12	11 560 277	17.5	0	29	0-141	26 312 456	0.3	0	157	33-536
2012-13	8 240 515	3.3	0	48	7-162	24 271 654	1.9	0	144	28-514
2013-14	16 448 081	11.7	0	100	26-291	24 419 994	4.1	0	114	26-357
2014-15	14 077 799	2.5	0	50	8-156	25 289 849	2.1	0	109	22-370
2015-16	18 603 012	10.8	0	79	23-219	24 887 264	2.5	0	106	21-373
2016-17	22 150 093	17.6	0	79	13-267	24 396 916	4.5	0	75	15-254
Large SLL						**Small SLL**				
2007-08	568 285	50.2	0	0	0-1	1 678 054	8.1	0.01	12	4-24
2008-09	809 230	97.2	0	2	2-2	2 306 403	6.5	0.01	16	6-30
2009-10	478 558	100	0	0	0-0	2 516 706	7.3	0.01	19	8-36
2010-11	503 370	100	0	0	0-0	2 684 809	6.4	0	12	4-25
2011-12	551 440	100.6	0	0	0-0	2 548 687	6.8	0.01	9	3-18
2012-13	487 520	100	0	0	0-0	2 389 412	3.1	0	10	3-19
2013-14	653 330	100	0	0	0-0	1 896 434	6.8	0	7	1-14
2014-15	622 300	99.4	0	0	0-0	1 790 036	6	0	5	1-11
2015-16	54 450	43.6	0	0	0-1	2 302 691	13	0	7	2-15
2016-17	0					2 092 486	16.5	0	4	0-10

Table 8.8: Summary of model-estimated total captures of Buller's albatross combined by October fishing year in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries) (effort in tows), large (>28m) and small (<28m) surface-longline (effort in hooks) and large (>28m) and small (<28m) bottom-longline (effort in hooks) fisheries between 2007–08 and 2016–17. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed, capture rate, observed capture rate. http://data.dragonfly.co.nz/psc. Data version v2018001.

Year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All Effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						**Inshore trawl**				
2007-08	39 210	22.7	0.19	99	62-147	50 314	0.3	0	19	0-61
2008-09	35 927	22.4	0.22	76	48-112	51 625	3.2	0	19	0-59
2009-10	36 766	22.3	0.13	67	40-108	56 126	1.4	0	21	1-65
2010-11	35 522	18.9	0.3	84	55-123	50 563	1.4	0	17	0-53
2011-12	34 128	25.9	0.41	122	87-172	50 290	1	0	17	0-56
2012-13	33 321	36.5	0.48	105	84-136	50 517	0.4	0	17	0-54
2013-14	34 612	33.4	0.33	89	66-121	50 498	3.2	0	17	0-54
2014-15	34 737	33.1	0.3	90	66-122	44 030	4.7	0	14	0-42
2015-16	34 304	32	0.52	121	95-155	43 726	4.6	0	15	0-50
2016-17	33 891	30.9	0.22	69	48-98	44 281	7.3	0	15	0-48
Large BLL						**Small BLL**				
2007-08	14 123 096	21.8	0	17	6-40	27 376 411	1.8	0	51	18-107
2008-09	12 861 501	24.9	0	4	0-12	24 573 964	3.6	0	30	9-60
2009-10	13 607 740	12.6	0	9	1-24	26 845 521	2.7	0	33	11-72
2010-11	12 914 717	11.8	0	9	1-23	27 981 339	1	0	40	13-90
2011-12	11 560 277	17.5	0	4	0-13	26 312 456	0.3	0.04	41	14-98
2012-13	8 240 515	3.3	0	6	1-15	24 271 654	1.9	0	24	7-55
2013-14	16 448 081	11.7	0	11	3-24	24 419 994	4.1	0	24	7-52
2014-15	14 077 799	2.5	0	9	2-22	25 289 849	2.1	0	21	6-45
2015-16	18 603 012	10.8	0	12	4-25	24 887 264	2.5	0	20	6-43
2016-17	22 150 093	17.6	0	10	2-29	24 396 916	4.5	0	19	6-41
Large SLL						**Small SLL**				
2007-08	568 285	50.2	0.06	30	18-58	1 678 754	8.1	0.02	65	27-130
2008-09	809 230	97.2	0.04	29	28-38	2 305 503	6.5	0.01	94	43-185
2009-10	478 558	100	0.09	41	41-41	2 517 986	7.3	0.15	130	78-213
2010-11	503 370	100	0.05	23	23-23	2 683 529	6.4	0.02	105	53-184
2011-12	551 440	100.6	0.05	27	27-27	2 548 787	6.8	0.02	138	70-248
2012-13	487 520	100	0	2	2-2	2 389 462	3.1	0.11	118	62-208
2013-14	653 330	100	0.02	15	15-15	1 896 434	6.8	0.06	105	53-189
2014-15	622 300	99.4	0.03	18	18-19	1 790 036	6	0.03	85	40-164
2015-16	54 450	43.6	0.59	19	14-40	2 303 441	13	0.14	151	95-249
2016-17	0					2 092 486	16.5	0.04	114	61-206

Table 8.9: Summary of model-estimated total captures of white–chinned petrel combined by October fishing year in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries) (effort in tows), large (>28m) and small (<28m) surface-longline (effort in hooks) and large (>28m) and small (<28m) bottom-longline (effort in hooks) fisheries between 2007–08 and 2016–17. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed, capture rate, observed capture rate. http://data.dragonfly.co.nz/psc. Data version v2018001.

Year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All Effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						**Inshore trawl**				
2007-08	39 210	22.7	0.65	253	160-400	50 314	0.3	0	14	1-45
2008-09	35 927	22.4	1.29	298	206-437	51 625	3.2	0.06	12	2-36
2009-10	36 766	22.3	0.88	285	184-446	56 126	1.4	0	15	1-46
2010-11	35 522	18.9	1.9	414	279-623	50 563	1.4	0	15	1-49
2011-12	34 128	25.9	0.69	196	131-293	50 290	1	0	11	1-35
2012-13	33 321	36.5	2.41	409	353-499	50 517	0.4	0	14	1-43
2013-14	34 612	33.4	1.31	239	199-299	50 498	3.2	0	15	1-46
2014-15	34 737	33.1	2.4	397	342-480	44 030	4.7	0	13	1-44
2015-16	34 304	32	1.46	242	204-305	43 726	4.6	0	12	1-38
2016-17	33 891	30.9	1.36	272	209-370	44 281	7.3	0	11	1-36
Large BLL						**Small BLL**				
2007-08	14 127 896	21.8	0	257	55-810	27 376 411	1.8	0.01	462	115-1 446
2008-09	12 861 501	24.9	0	199	21-704	24 573 964	3.6	0	438	117-1 324
2009-10	13 602 940	12.6	0	181	18-662	26 845 521	2.7	0	413	106-1 201
2010-11	12 919 517	11.8	0.01	201	47-637	27 981 339	1	0	482	125-1 441
2011-12	11 571 447	17.5	0	94	6-381	26 312 456	0.3	0	467	107-1 479
2012-13	8 234 145	3.3	0	124	13-420	24 271 654	1.9	0	409	96-1 265
2013-14	16 459 721	11.7	0.02	458	144-1 163	24 419 994	4.1	0	295	81-837
2014-15	14 060 072	2.5	0.03	327	88-937	25 289 849	2.1	0	279	67-849
2015-16	18 604 396	10.8	0.04	434	179-1 020	24 887 264	2.5	0.01	327	86-936
2016-17	22 157 051	17.6	0	389	120-990	24 396 916	4.5	0.02	293	85-821
Large SLL						**Small SLL**				
2007-08	568 285	50.2	0.01	5	4-11	1 678 054	8.1	0	17	4-51
2008-09	809 230	97.2	0	2	2-3	2 306 403	6.5	0.01	22	5-60
2009-10	478 558	100	0	2	2-2	2 516 706	7.3	0.01	29	8-79
2010-11	503 370	100	0	2	2-2	2 684 809	6.4	0.02	28	10-67
2011-12	551 440	100.6	0	0	0-0	2 548 687	6.8	0.02	28	10-73
2012-13	487 520	100	0	1	1-1	2 389 412	3.1	0	36	8-119
2013-14	653 330	100	0	0	0-0	1 896 434	6.8	0	25	5-83
2014-15	622 300	99.4	0	0	0-0	1 790 036	6	0.02	35	7-126
2015-16	54 450	43.6	0	1	0-10	2 302 691	13	0	48	8-183
2016-17	0					2 092 486	16.5	0	26	4-94

Table 8.10: Summary of model-estimated total captures of sooty shearwaters combined by October fishing year in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries) (effort in tows), large (>28m) and small (<28m) surface-longline (effort in hooks) and large (>28m) and small (<28m) bottom-longline (effort in hooks) fisheries between 2007–08 and 2016–17. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed, capture rate, observed capture rate. http://data.dragonfly.co.nz/psc. Data version v2018001.

Year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All Effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						**Inshore trawl**				
2007-08	39 210	22.7	0.92	444	280-704	50 314	0.3	0	48	10-129
2008-09	35 927	22.4	1.9	562	385-856	51 625	3.2	0.06	52	13-136
2009-10	36 766	22.3	0.61	345	200-601	56 126	1.4	0	58	13-152
2010-11	35 522	18.9	1.73	496	328-760	50 563	1.4	0	58	13-150
2011-12	34 128	25.9	0.39	276	149-512	50 290	1	0	50	10-136
2012-13	33 321	36.5	1.08	309	214-510	50 517	0.4	0	55	12-143
2013-14	34 612	33.4	1.1	334	231-519	50 498	3.2	0	60	13-164
2014-15	34 737	33.1	1.25	372	257-581	44 030	4.7	0.05	46	11-121
2015-16	34 304	32	0.57	232	137-430	43 726	4.6	0	45	9-121
2016-17	33 891	30.9	1.24	326	225-516	44 281	7.3	0.09	51	
Large BLL						**Small BLL**				
2007-08	14 123 096	21.8	0	27	11-54	27 384 307	1.8	0	11	0-40
2008-09	12 861 501	24.9	0	24	1-76	24 569 215	3.6	0	12	1-40
2009-10	13 607 740	12.6	0	43	15-105	26 848 346	2.7	0	11	0-38
2010-11	12 914 717	11.8	0	20	0-67	27 975 994	1	0	14	1-47
2011-12	11 560 277	17.5	0	18	0-55	26 319 876	0.3	0	11	0-39
2012-13	8 240 515	3.3	0	18	1-60	24 275 944	1.9	0	9	0-32
2013-14	16 448 081	11.7	0	23	4-63	24 411 354	4.1	0	7	1-20
2014-15	14 077 799	2.5	0	14	0-45	25 287 149	2.1	0	6	0-19
2015-16	18 603 012	10.8	0	8	0-25	24 898 664	2.5	0	7	0-27
2016-17	22 150 093	17.6	0	12	0-40	24 385 436	4.5	0	6	0-21
Large SLL						**Small SLL**				
2007-08	568 285	50.2	0	0	0-1	1 678 054	8.1	0	1	0-3
2008-09	809 230	97.2	0	0	0-0	2 306 403	6.5	0	1	0-3
2009-10	478 558	100	0	0	0-0	2 516 706	7.3	0	1	0-5
2010-11	503 370	100	0	0	0-0	2 684 809	6.4	0	1	0-4
2011-12	551 440	100.6	0	0	0-0	2 548 687	6.8	0	0	0-3
2012-13	487 520	100	0	0	0-0	2 389 412	3.1	0	1	0-4
2013-14	653 330	100	0	0	0-0	1 896 434	6.8	0	1	0-3
2014-15	622 300	99.4	0	0	0-0	1 790 036	6	0	1	0-4
2015-16	54 450	43.6	0	0	0-1	2 302 691	13	0	1	0-5
2016-17	0					2 092 486	16.5	0	1	0-4

Table 8.11: Summary of model-estimated total captures of black petrels combined by October fishing year in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries) (effort in tows), large (>28m) and small (<28m) surface-longline (effort in hooks) and large (>28m) and small (<28m) bottom-longline (effort in hooks) fisheries between 2007–08 and 2016–17. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed, capture rate, observed capture rate. http://data.dragonfly.co.nz/psc. Data version v2018001.

Year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All Effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						**Inshore trawl**				
2007-08	39 210	22.7	0	1	0-4	50 314	0.3	0	42	23-66
2008-09	35 927	22.4	0	1	0-4	51 625	3.2	0	44	24-69
2009-10	36 766	22.3	0	1	0-6	56 126	1.4	0	44	24-68
2010-11	35 522	18.9	0	1	0-5	50 563	1.4	0	42	23-65
2011-12	34 128	25.9	0	1	0-4	50 290	1	0.19	41	23-64
2012-13	33 321	36.5	0	1	0-6	50 517	0.4	0	35	18-55
2013-14	34 612	33.4	0	1	0-5	50 498	3.2	0.31	38	22-56
2014-15	34 737	33.1	0	1	0-4	44 030	4.7	0.1	30	17-48
2015-16	34 304	32	0	1	0-4	43 726	4.6	0.65	42	28-59
2016-17	33 891	30.9	0	1	0-4	44 281	7.3	0.18	35	21-53
Large BLL						**Small BLL**				
2007-08	14 123 096	21.8	0	1	0-4	27 384 307	1.8	0.01	313	164-585
2008-09	12 861 501	24.9	0	1	0-5	24 569 215	3.6	0.01	290	155-533
2009-10	13 607 740	12.6	0	1	0-5	26 848 346	2.7	0.06	352	198-633
2010-11	12 914 717	11.8	0	1	0-5	27 975 994	1	0.01	323	171-586
2011-12	11 560 277	17.5	0	2	0-7	26 319 876	0.3	0	288	145-538
2012-13	8 240 515	3.3	0	0	0-2	24 275 944	1.9	0	208	115-363
2013-14	16 448 081	11.7	0	0	0-2	24 411 354	4.1	0.01	214	118-386
2014-15	14 077 799	2.5	0	0	0-1	25 287 149	2.1	0	193	102-345
2015-16	18 603 012	10.8	0	0	0-1	24 898 664	2.5	0	154	80-276
2016-17	22 150 093	17.6	0	0	0-1	24 385 436	4.5	0.01	188	104-345
Large SLL						**Small SLL**				
2007-08	568 285	50.2	0	0	0-0	1 678 754	8.1	0.01	66	29-148
2008-09	809 230	97.2	0	0	0-0	2 305 503	6.5	0.01	80	36-178
2009-10	478 558	100	0	0	0-0	2 517 986	7.3	0.03	72	35-143
2010-11	503 370	100	0	0	0-0	2 683 529	6.4	0.01	117	54-257
2011-12	551 440	100.6	0	0	0-0	2 548 787	6.8	0.01	93	42-201
2012-13	487 520	100	0	0	0-0	2 389 462	3.1	0	90	40-201
2013-14	653 330	100	0	0	0-0	1 896 434	6.8	0	76	32-173
2014-15	622 300	99.4	0	0	0-0	1 790 036	6	0	55	21-134
2015-16	54 450	43.6	0	0	0-2	2 303 441	13	0.02	73	35-163
2016-17	0					2 092 486	16.5	0.02	64	31-143

Table 8.12: Summary of model-estimated total captures of grey petrels combined by October fishing year in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries) (effort in tows), large (>28m) and small (<28m) surface-longline (effort in hooks) and large (>28m) and small (<28m) bottom-longline (effort in hooks) fisheries between 2007–08 and 2016–17. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed, capture rate, observed capture rate. http://data.dragonfly.co.nz/psc. Data version v2018001.

Year	Fishing effor		Capture	Modelled estimates		Fishing effor		Capture	Modelled estimates	
	All Effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						**Inshore trawl**				
2007-08	39 210	22.7	0.02	8	3-18	50 314	0.3	0	4	0-16
2008-09	35 927	22.4	0	6	0-17	51 625	3.2	0	4	0-16
2009-10	36 766	22.3	0.11	26	15-45	56 126	1.4	0	6	0-23
2010-11	35 522	18.9	0.1	20	11-33	50 563	1.4	0	4	0-17
2011-12	34 128	25.9	0.01	4	1-Oct	50 290	1	0	4	0-15
2012-13	33 321	36.5	0.07	12	9-17	50 517	0.4	0	3	0-15
2013-14	34 612	33.4	0.1	14	11-19	50 498	3.2	0	4	0-15
2014-15	34 737	33.1	0.04	7	5-12	44 030	4.7	0	3	0-12
2015-16	34 304	32	0.03	6	3-11	43 726	4.6	0	3	0-12
2016-17	33 891	30.9	0.04	7	4-12	44 281	7.3	0	3	0-12
Large BLL						**Small BLL**				
2007-08	14 127 896	21.8	0	33	6-127	27 384 307	1.8	0	160	42-496
2008-09	12 861 501	24.9	0	19	1-95	24 569 215	3.6	0	139	38-398
2009-10	13 602 940	12.6	0	27	0-152	26 848 346	2.7	0	181	49-527
2010-11	12 919 517	11.8	0	32	0-184	27 975 994	1	0	160	46-437
2011-12	11 571 447	17.5	0	10	0-66	26 319 876	0.3	0	140	42-389
2012-13	8 234 145	3.3	0	12	0-58	24 275 944	1.9	0	127	36-353
2013-14	16 459 721	11.7	0	12	0-68	24 411 354	4.1	0	142	40-392
2014-15	14 060 072	2.5	0	26	0-168	25 287 149	2.1	0.01	144	42-407
2015-16	18 604 396	10.8	0	18	0-107	24 898 664	2.5	0	124	33-367
2016-17	22 157 051	17.6	0	12	0-75	24 385 436	4.5	0	129	33-390
Large SLL						**Small SLL**				
2007-08	568 285	50.2	0	0	0-1	1 678 054	8.1	0.01	17	5-41
2008-09	809 230	97.2	0.01	5	5-5	2 306 403	6.5	0.01	18	6-40
2009-10	478 558	100	0	0	0-0	2 516 706	7.3	0.01	27	10-57
2010-11	503 370	100	0	0	0-0	2 684 809	6.4	0	22	7-49
2011-12	551 440	100.6	0	0	0-0	2 548 687	6.8	0.01	18	7-39
2012-13	487 520	100	0	0	0-0	2 389 412	3.1	0	20	7-42
2013-14	653 330	100	0	0	0-0	1 896 434	6.8	0.01	22	7-47
2014-15	622 300	99.4	0	0	0-0	1 790 036	6	0	19	6-43
2015-16	54 450	43.6	0	0	0-1	2 302 691	13	0	22	8-51
2016-17	0					2 092 486	16.5	0	23	8-53

Table 8.13: Summary of model-estimated total captures of flesh-footed shearwaters combined by October fishing year in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries) (effort in tows), large (>28m) and small (<28m) surface-longline (effort in hooks) and large (>28m) and small (<28m) bottom-longline (effort in hooks) fisheries between 2007–08 and 2016–17. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed, capture rate, observed capture rate. http://data.dragonfly.co.nz/psc. Data version v2018001.

Year	Fishing effor		Capture	Modelled estimates		Fishing effor		Capture	Modelled estimates	
	All Effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						**Inshore trawl**				
2007-08	39 210	22.7	0.07	51	20-117	50 314	0.3	0	42	17-80
2008-09	35 927	22.4	0.04	51	18-120	51 625	3.2	0	43	18-80
2009-10	36 766	22.3	0.01	64	23-152	56 126	1.4	0.13	45	20-84
2010-11	35 522	18.9	0.22	74	33-169	50 563	1.4	0	40	17-75
2011-12	34 128	25.9	0	42	12-111	50 290	1	0	40	17-76
2012-13	33 321	36.5	0	59	18-149	50 517	0.4	0	38	16-74
2013-14	34 612	33.4	0.02	49	16-125	50 498	3.2	0.43	43	22-79
2014-15	34 737	33.1	0.01	49	16-124	44 030	4.7	0.34	39	20-69
2015-16	34 304	32	0.01	54	17-136	43 726	4.6	0.05	33	13-67
2016-17	33 891	30.9	0.01	56	17-146	44 281	7.3	0	30	12-59
Large BLL						**Small BLL**				
2007-08	14 123 096	21.8	0	0	0-2	27 376 411	1.8	0	290	186-453
2008-09	12 861 501	24.9	0	2	0-12	24 573 964	3.6	0.02	302	198-456
2009-10	13 607 740	12.6	0	1	0-5	26 845 521	2.7	0.02	287	188-427
2010-11	12 914 717	11.8	0	0	0-3	27 981 339	1	0	318	207-480
2011-12	11 560 277	17.5	0	1	0-4	26 312 456	0.3	0	280	185-422
2012-13	8 240 515	3.3	0	4	0-14	24 271 654	1.9	0	292	187-453
2013-14	16 448 081	11.7	0	4	0-14	24 419 994	4.1	0.03	293	197-456
2014-15	14 077 799	2.5	0	4	0-12	25 289 849	2.1	0.02	266	170-423
2015-16	18 603 012	10.8	0	3	0-10	24 887 264	2.5	0.02	244	157-384
2016-17	22 150 093	17.6	0	3	0-12	24 396 916	4.5	0	239	150-385
Large SLL						**Small SLL**				
2007-08	568 285	50.2	0	0	0-0	1 678 754	8.1	0.01	162	58-362
2008-09	809 230	97.2	0	0	0-0	2 305 503	6.5	0	209	81-457
2009-10	478 558	100	0	0	0-0	2 517 986	7.3	0	205	77-438
2010-11	503 370	100	0	0	0-0	2 683 529	6.4	0.01	259	101-575
2011-12	551 440	100.6	0	0	0-0	2 548 787	6.8	0	205	68-495
2012-13	487 520	100	0	0	0-0	2 389 462	3.1	0	184	63-428
2013-14	653 330	100	0	0	0-0	1 896 434	6.8	0	160	51-395
2014-15	622 300	99.4	0	0	0-0	1 790 036	6	0.01	93	24-238
2015-16	54 450	43.6	0	2	0-12	2 303 441	13	0	135	43-332
2016-17	0					2 092 486	16.5	0	108	28-282

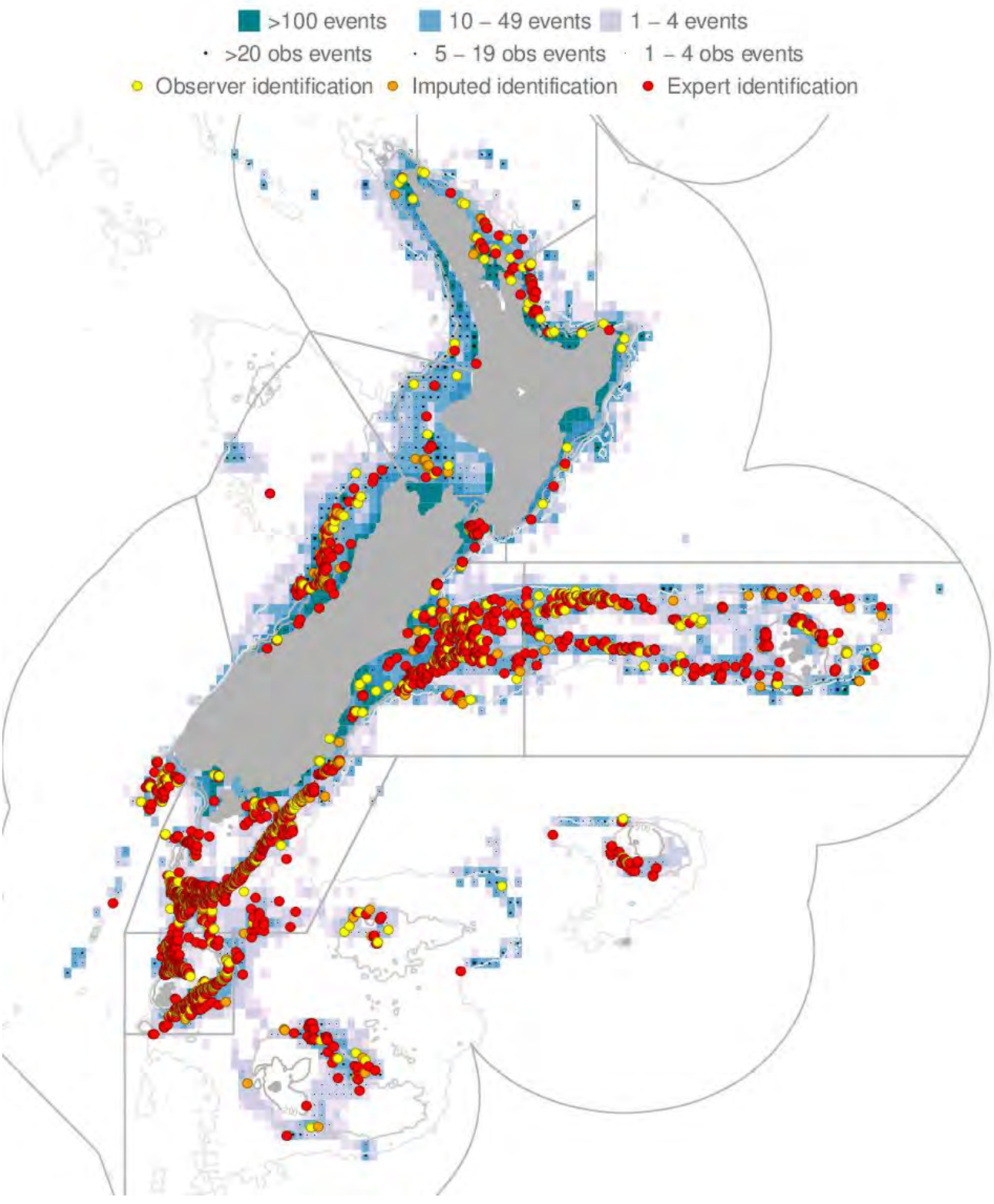


Figure 8.4: Map of trawl fishing effort and all observed seabird captures in trawls, October 2002 to September 2017. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by dots, red for those identified to species by experts, orange for those with identification imputed, and yellow for those using observer identification. Fishing is shown only if the effort could be assigned a latitude and longitude, and if there were three or more vessels fishing within a cell. http://data.dragonfly.co.nz/psc . Data version v20180001.

Table 8.14: Summary of seabirds observed captured in trawl fisheries 2002–03 to 2016–17. Declared target species are: SQU, arrow squid; HOK+, hoki, hake, ling; MID, other middle-depth species – silver, white, and common warehou, barracouta, alfonsinos, stargazer, redbait, rubyfish; SCI, scampi; ORH+, orange roughy and oreos; SBW, southern blue whiting; JMA, Jack mackerels; INS, other inshore species for which one or more captures have been observed – elephantfish, gemfish, gurnard, tarakihi, red cod, spiny dogfish, John dory, snapper; FLA, flatfishes. http://data.dragonfly.co.nz/psc.

Species name or group	Declared target species									
	SQU	HOK+	MID	SCI	ORH +	SBW	JMA	INS	FLA	ALL
New Zealand white-capped albatross	974	122	124	25	4	0	16	26	4	**1295**
Salvin's albatross	26	207	94	47	20	17	3	19	0	**433**
Southern Buller's albatross	157	128	58	10	4	1	6	1	0	**365**
Campbell black-browed albatross	1	13	0	1	0	4	0	0	0	**19**
Albatrosses	6	5	4	2	0	1	0	1	0	**19**
Southern royal albatross	7	2	2	0	1	2	0	0	0	**14**
Chatham Island albatross	0	2	1	1	8	0	0	0	0	**12**
Smaller albatrosses	0	2	0	1	0	1	0	0	0	**4**
Royal albatrosses	3	0	0	0	0	0	0	0	0	**3**
Black-browed albatross	1	1	0	0	0	0	0	0	0	**2**
Wandering albatrosses	0	1	1	0	0	0	0	0	0	**2**
Antipodean albatross	0	0	0	0	1	0	0	0	0	**1**
Buller's albatross	1	0	0	0	0	0	0	0	0	**1**
Gibson's albatross	0	0	0	0	1	0	0	0	0	**1**
Great albatrosses	0	1	0	0	0	0	0	0	0	**1**
Northern Buller's albatross	0	1	0	0	0	0	0	0	0	**1**
Northern royal albatross	0	0	0	0	1	0	0	0	0	**1**
All albatrosses	**1176**	**485**	**284**	**87**	**40**	**26**	**25**	**47**	**4**	**2174**
White-chinned petrel	1227	143	154	75	4	0	30	1	0	**1634**
Sooty shearwater	927	332	221	37	3	0	9	5	0	**1534**
Flesh-footed shearwater	0	5	0	37	0	0	0	18	0	**60**
Grey petrel	1	4	0	1	2	52	0	0	0	**60**
Cape petrel	1	26	0	3	9	1	0	0	0	**40**
Spotted shag	0	0	0	0	0	0	0	0	32	**32**
Westland petrel	0	24	2	0	0	0	1	2	0	**29**
Black petrel	0	0	0	0	0	0	0	28	0	**28**
Common diving petrel	8	6	5	1	2	1	2	2	0	**27**
Fairy prion	2	8	2	0	0	0	8	0	0	**20**
Snares Cape petrel	1	10	0	0	0	3	0	0	0	**14**
Antarctic prion	12	0	0	0	0	0	0	0	0	**12**
Northern giant petrel	0	8	2	1	1	0	0	0	0	**12**
New Zealand white-faced storm petrel	3	0	0	0	3	0	3	0	0	**9**
Fulmar prion	0	0	0	0	0	0	9	0	0	**9**
Grey-backed storm petrel	2	3	0	0	0	3	1	0	0	**9**
Cape petrels	0	6	0	0	0	0	2	0	0	**8**
Giant petrels	7	0	1	0	0	0	0	0	0	**8**
Short-tailed shearwater	0	5	2	0	1	0	0	0	0	**8**
Fulmars, petrels, prions and shearwaters	1	1	0	2	0	0	0	0	0	**4**
Petrels, prions, and shearwaters	0	0	0	0	1	2	0	1	0	**4**
Shearwaters	0	0	0	0	0	0	0	4	0	**4**

Table 8.14 [Continued]:

Species name or group	Declared target species									
	SQU	HOK+	MID	SCI	JMA	INS	SBW	ORH+	FLA	ALL
Large seabirds	0	1	0	0	0	1	1	0	0	**3**
Prions	2	1	0	0	0	0	0	0	0	**3**
Small seabirds	0	2	0	0	0	0	0	0	0	**2**
Australasian gannet	0	0	0	0	0	0	1	0	0	**1**
Wilson's storm petrel	0	0	0	0	0	0	0	1	0	**1**
Black-bellied storm petrel	0	1	0	0	0	0	0	0	0	**1**
Broad-billed prion	0	0	1	0	0	0	0	0	0	**1**
Gadfly petrels	1	0	0	0	0	0	0	0	0	**1**
Grey-faced petrel	0	0	0	0	0	0	0	1	0	**1**
Mid-sized petrels & shearwaters	1	0	0	0	0	0	0	0	0	**1**
Seabirds	1	0	0	0	0	0	0	0	0	**1**
Storm petrels	0	1	0	0	0	0	0	0	0	**1**
All other birds	**2197**	**587**	**390**	**157**	**26**	**63**	**67**	**63**	**32**	**3582**
Grand total	**3373**	**1072**	**674**	**244**	**66**	**89**	**92**	**110**	**36**	**5756**

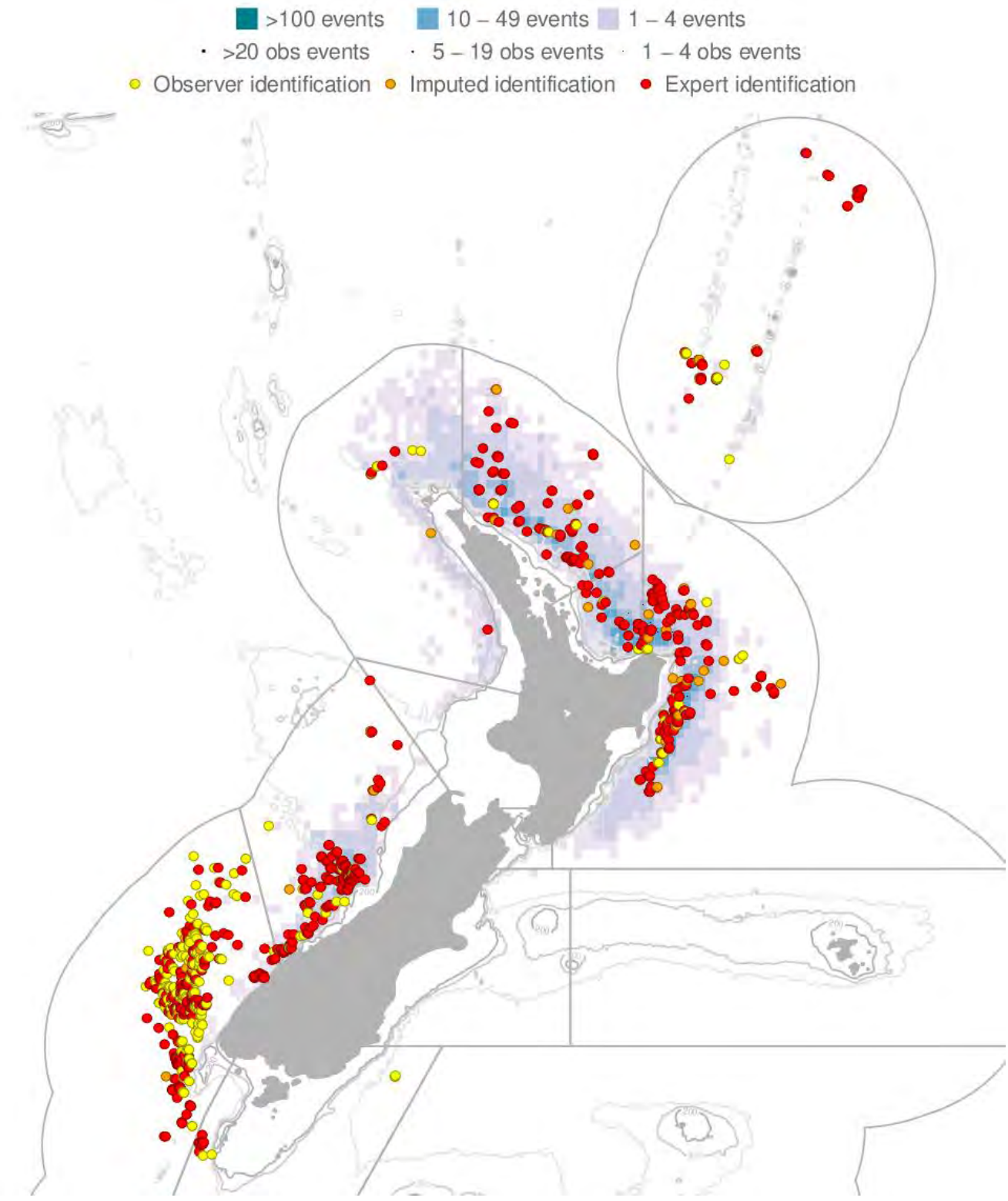


Figure 8.5: Map of surface-longline fishing effort and all observed seabird captures by surface longlines, October 2002 to September 2017. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by dots, red for those identified to species by experts, orange for those with identification imputed, and yellow for those using observer identification. Fishing is shown only if the effort could be assigned a latitude and longitude, and if there were three or more vessels fishing within a cell (here, 90.2% of effort is displayed). http://data.dragonfly.co.nz/psc. Data version v20180001.

Table 8.15: Summary of seabirds observed captured in surface-longline fisheries 2002–03 to 2016–17. Declared target species are: SBT, southern bluefin tuna; BIG, bigeye tuna; SWO, broadbill swordfish; ALB, albacore tuna. http://data.dragonfly.co.nz/psc. Data version v2018v1.

Species name or group	Declared target species				
	SBT	SWO	BIG	ALB	ALL
Southern Buller's albatross	445	1	11	7	**464**
New Zealand white-capped albatross	176	4	2	0	**182**
Campbell black-browed albatross	31	3	4	18	**56**
Gibson's albatross	13	12	10	7	**42**
Antipodean albatross	6	15	11	3	**35**
Albatrosses	0	33	1	0	**34**
Southern royal albatross	9	0	3	0	**12**
Salvin's albatross	4	0	4	1	**9**
Antipodean and Gibson's albatrosses	0	5	2	0	**7**
Wandering albatross	2	2	2	0	**6**
Wandering albatrosses	6	0	0	0	**6**
Black-browed albatross	5	0	0	0	**5**
Black-browed albatrosses	0	2	1	0	**3**
Northern royal albatross	0	0	2	0	**2**
Great albatrosses	1	0	0	0	**1**
Grey-headed albatross	1	0	0	0	**1**
Light-mantled sooty albatross	1	0	0	0	**1**
Northern Buller's albatross	1	0	0	0	**1**
Smaller albatrosses	1	0	0	0	**1**
All albatrosses	**702**	**77**	**53**	**36**	**868**
Grey petrel	42	3	0	5	**50**
White-chinned petrel	24	7	6	4	**41**
Westland petrel	28	1	0	1	**30**
Black petrel	0	3	24	2	**29**
Grey-faced petrel	1	2	3	16	**22**
Flesh-footed shearwater	0	2	11	0	**13**
Sooty shearwater	3	1	0	7	**11**
Large seabirds	3	0	0	0	**3**
Cape petrels	2	0	0	0	**2**
Southern giant petrel	2	0	0	0	**2**
White-headed petrel	0	0	0	2	**2**
Gadfly petrels	0	0	1	0	**1**
All other birds	**105**	**19**	**45**	**37**	**206**
Grand total	**807**	**96**	**98**	**73**	**1074**

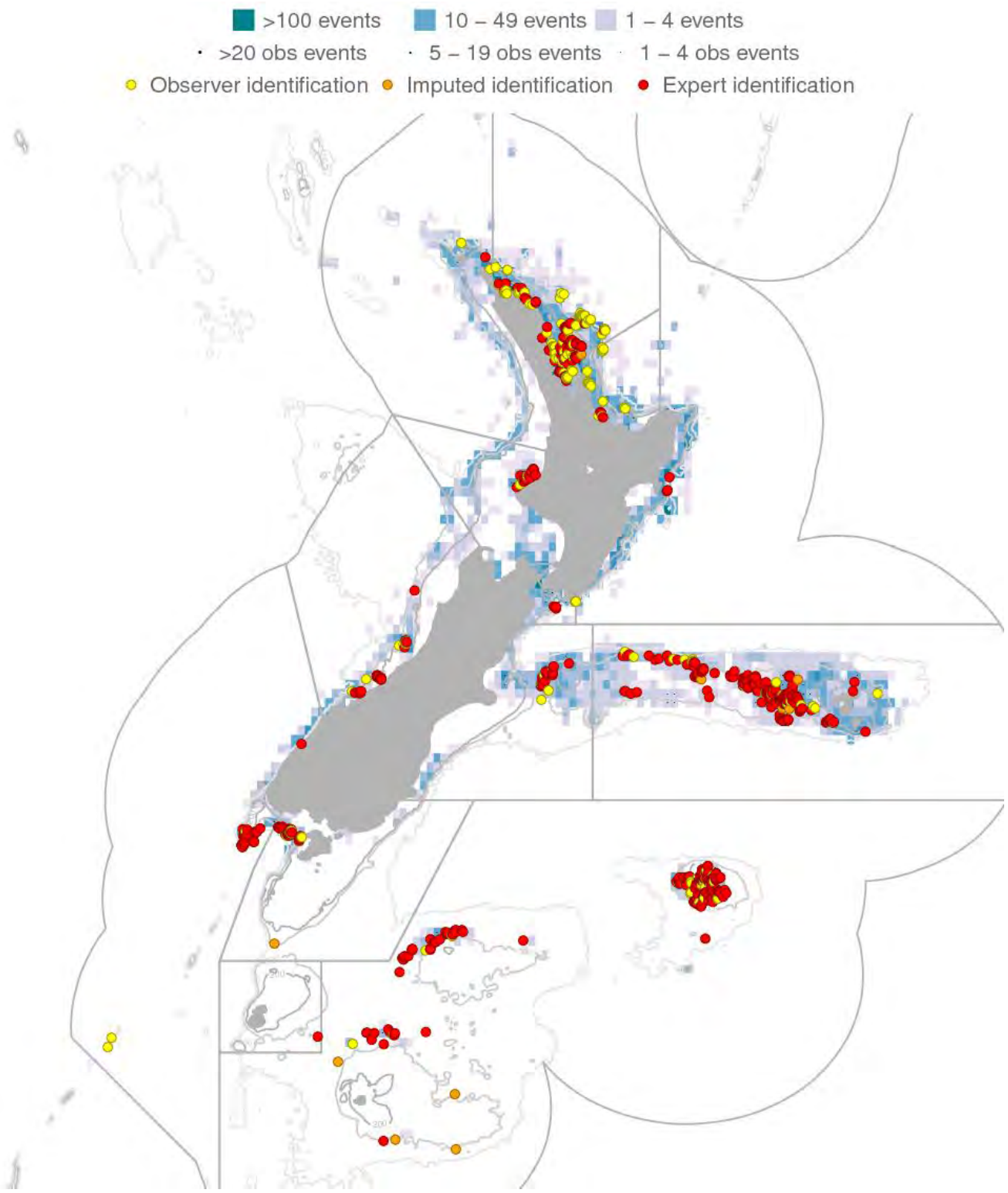


Figure 8.6: Map of bottom-longline fishing effort and all observed seabird captures by bottom longline, October 2002 to September 2017. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by dots, red for those identified to species by experts, orange for those with identification imputed, and yellow for those using observer identification. Fishing is shown only if the effort could be assigned a latitude and longitude, and if there were three or more vessels fishing within a cell (here, 96.4% of effort is displayed). http://data.dragonfly.co.nz/psc. Data version v20180001.

Table 8.16: Summary of seabirds observed captured in bottom-longline fisheries 2002–03 to 2016–17. Declared target species are: LIN, ling; SNA, snapper; BNS, bluenose; HPB, hapuku or bass; Other – gurnard, kahawai, toothfish, ribaldo, school shark, tarahiki. http://data.dragonfly.co.nz/psc. Data version v2018001.

Species name or group	Declared target species					
	LIN	SNA	BNS	HPB	Other	ALL
Salvin's albatross	61	0	0	0	3	**64**
Chatham Island albatross	19	0	0	0	0	**19**
Southern Buller's albatross	10	0	3	0	0	**13**
New Zealand white-capped albatross	7	0	0	0	0	**7**
Southern royal albatross	5	0	0	0	0	**5**
Campbell black-browed albatross	0	0	2	1	1	**4**
Albatrosses	3	0	0	0	0	**3**
Wandering albatrosses	2	0	1	0	0	**3**
Indian Ocean yellow-nosed albatross	1	0	0	0	0	**1**
Black-browed albatross	1	0	0	0	0	**1**
All albatrosses	**109**	**0**	**6**	**1**	**4**	**120**
White-chinned petrel	349	0	2	0	18	**369**
Flesh-footed shearwater	0	75	0	3	20	**98**
Black petrel	0	38	47	4	0	**89**
Grey petrel	70	0	0	0	0	**70**
Sooty shearwater	62	1	0	1	0	**64**
Cape petrel	20	0	0	0	0	**20**
Buller's shearwater	0	13	0	0	0	**13**
Fulmars, petrels, prions and shearwaters	0	10	0	0	0	**10**
Southern black-backed gull	0	5	0	0	3	**8**
Common diving petrel	7	0	0	0	0	**7**
Fluttering shearwater	0	5	0	0	1	**6**
Grey-faced petrel	0	0	0	6	0	**6**
Northern giant petrel	3	1	0	0	2	**6**
Prions	6	0	0	0	0	**6**
Cape petrels	4	0	0	0	0	**4**
Westland petrel	2	0	0	0	2	**4**
Australasian gannet	0	2	0	0	0	**2**
Pied shag	0	2	0	0	0	**2**
Red-billed gull	0	2	0	0	0	**2**
Seagulls	2	0	0	0	0	**2**
Small seabirds	2	0	0	0	0	**2**
Storm petrels	2	0	0	0	0	**2**
New Zealand white-faced storm petrel	1	0	0	0	0	**1**
Crested penguins	1	0	0	0	0	**1**
Giant petrels	1	0	0	0	0	**1**
Short-tailed shearwater	1	0	0	0	0	**1**
All other birds	**533**	**154**	**49**	**14**	**46**	**796**
Grand total	**642**	**154**	**55**	**15**	**50**	**916**

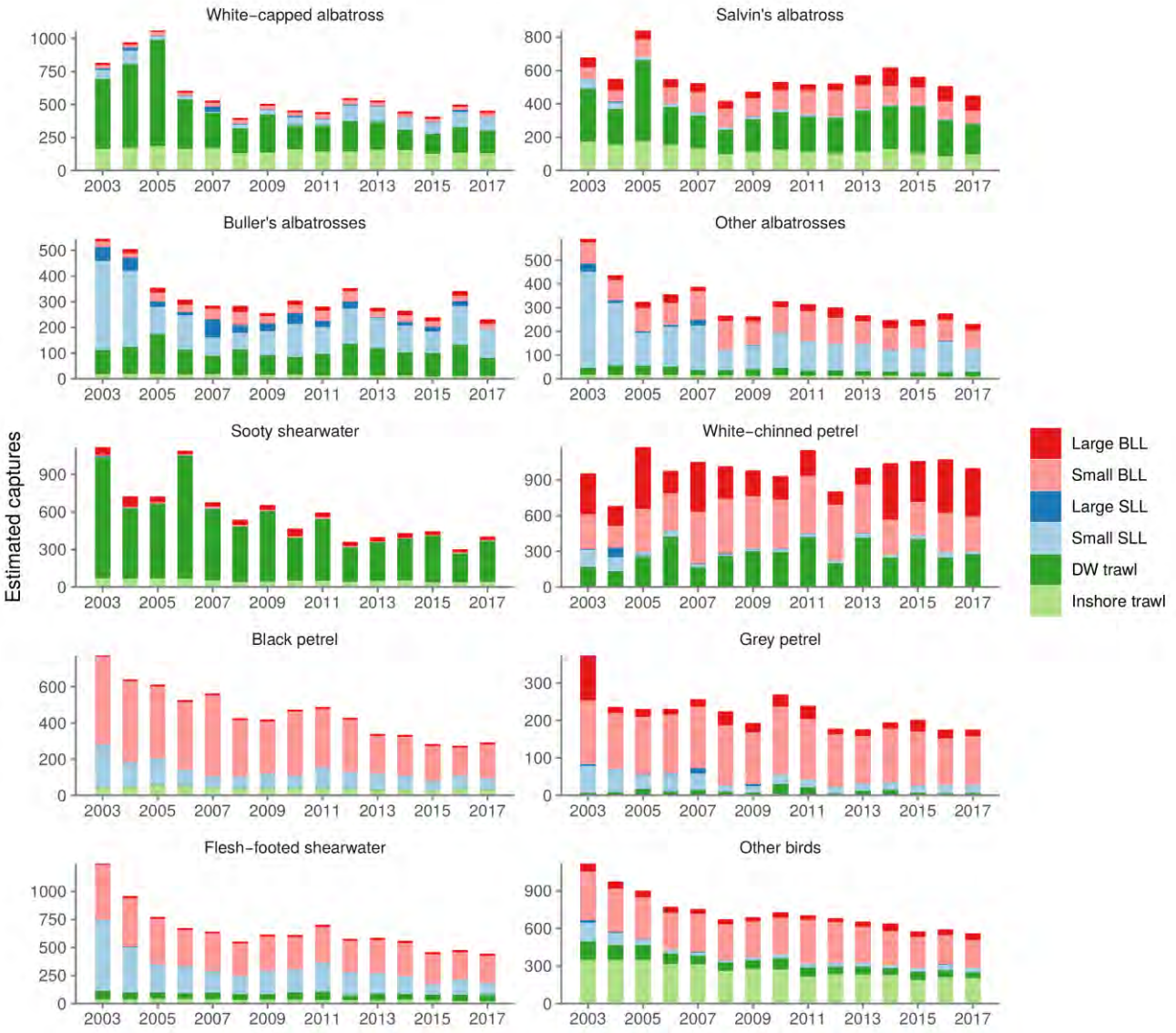


Figure 8.7: Model-based estimates of captures of the most numerous seabird taxa observed captured in inshore trawl (including inshore and flatfish target fisheries), DW trawl (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi and middle depth target fisheries), large (>28m) and small (<28m) surface-longline and large (>28m) and small (<28m) bottom-longline fisheries between 2002–03 and 2016–17. For confidence limits see Tables 8.10–8.13. Note that this level of aggregation conceals any different trends between areas. http://data.dragonfly.co.nz/psc. Data version v2018001.

Over the 2002–03 to 2016–17 period, there appear to have been downward trends (across all fisheries) in the estimated captures of white-capped albatross and non-albatross taxa other than white-chinned petrel (Figure 8.7). Estimated captures of other albatrosses and white-chinned petrel appear to have fluctuated without much trend, although there is some evidence for an increasing trend for white-chinned petrel, especially in trawl fisheries, although with large annual variations.

Because fishing effort often changes with time, estimates of total captures may not be the only index required for comprehensive monitoring. The number of captures is clearly more biologically relevant for birds, but capture rates by fishery may be more useful measures to assess fishery performance and the effectiveness of mitigation approaches. Dividing modelled catch estimates by the number of tows or hooks set in a particular fishery in each year provides catch rate indices by fishery. These are typically reported as the number of birds captured per 100

trawl tows or per 1000 longline hooks (Figure 8.8 to Figure 8.12).

For white-capped albatross, captures rates in the trawl fisheries for squid declined after 2004–05 (Figure 8.8) but showed no trend for inshore and middle depth trawlers, and increasing capture rates to a peak in 2015–16 for surface longliners targeting southern bluefin tuna. Together, these fisheries account for 73% of all estimated captures of white-capped albatross in these years.

For Salvin's albatross, captures rates have fluctuated without trend or increased in all fisheries taking substantial numbers of this species between 2002–03 and 2016–17, especially after 2006–07 (Figure 8.9). Capture rates were unusually high in the hoki trawl fishery in 2004–05. Together, these fisheries account for 65% of all estimated captures of Salvin's albatross in these years.

For Buller's albatross, estimated captures decreased in bigeye tuna target surface-longline fisheries between 2002–03 and 2016–17, while capture rates fluctuated. Captures and capture rates fluctuated with no trend in southern bluefin tuna target fisheries. The squid fishery shows some signs of an increasing trend, although had three relatively low capture rates in years 2012–13, 2014-15 and 2016–17 with a peak in 2015–16. The hoki trawl fishery in recent years has had higher captures and capture rates (Figure 8.10). Together these fisheries account for 64% of all estimated captures of southern Buller's albatross in these years.

For white-chinned petrel, captures rates increased between 2002–03 and 2016–17 in squid trawlers but showed little trend for bottom longliners targeting ling, bluenose and hapuka. High capture rates of white-chinned petrels alternate between the squid trawl and ling longline fisheries since 2012–13 (Figure 8.11). Together, these fisheries account for 77% of all estimated captures of white-chinned petrel in these years.

For sooty shearwaters, captures rates fluctuated without apparent trend between 2002–03 and 2016–17 in squid, middle-depth, hoki and inshore trawlers (Figure 8.12). Together, these fisheries account for 83% of all estimated captures of sooty shearwaters in these years.

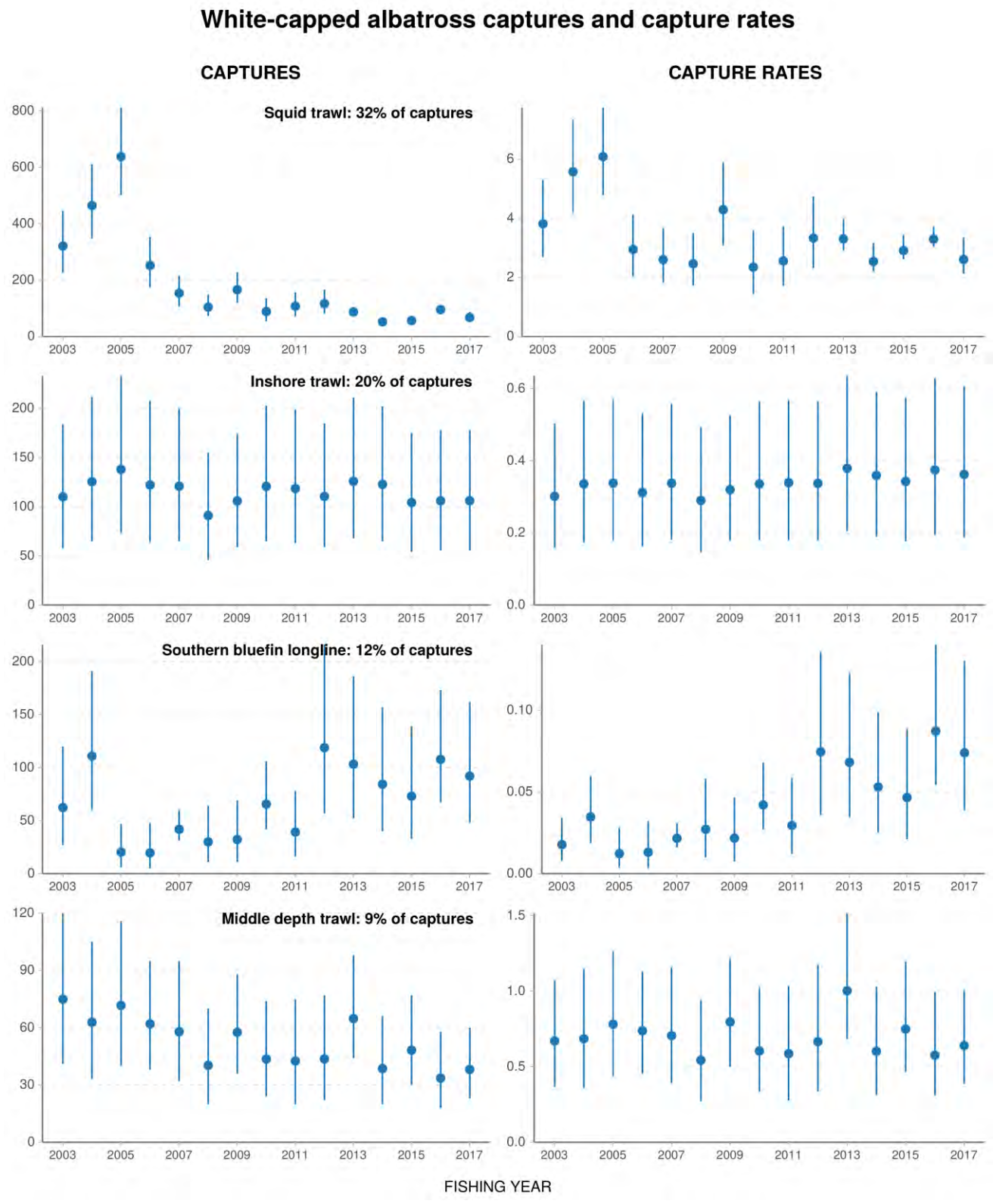


Figure 8.8: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of white-capped albatross in the four fisheries estimated to have taken the most captures between 2002–03 and 2016–17. http://data.dragonfly.co.nz/psc. Data version v2018001.

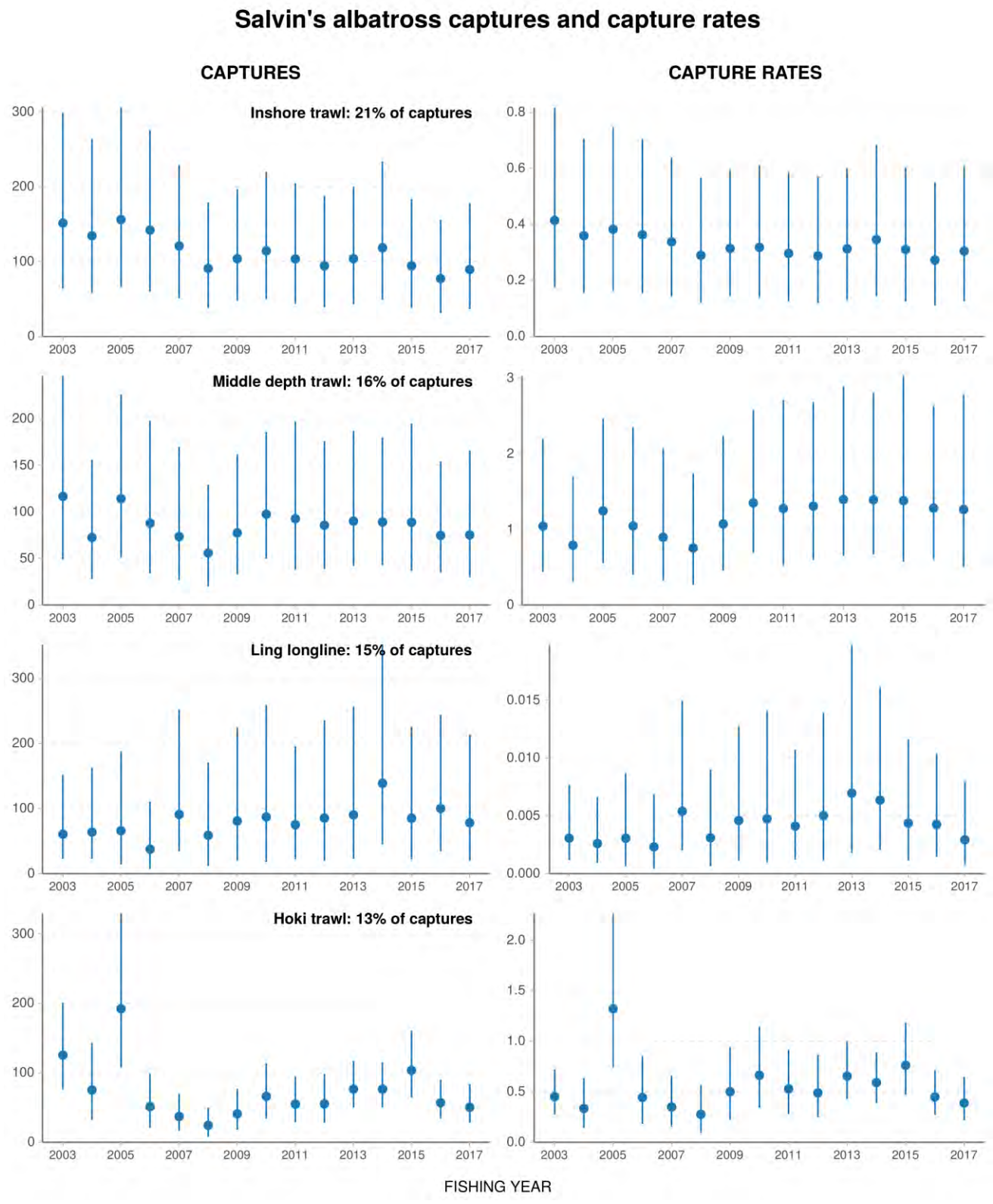


Figure 8.9: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of Salvin's albatross in the four fisheries estimated to have taken the most captures between 2002–03 and 2016–17. http://data.dragonfly.co.nz/psc. Data version v2018001.

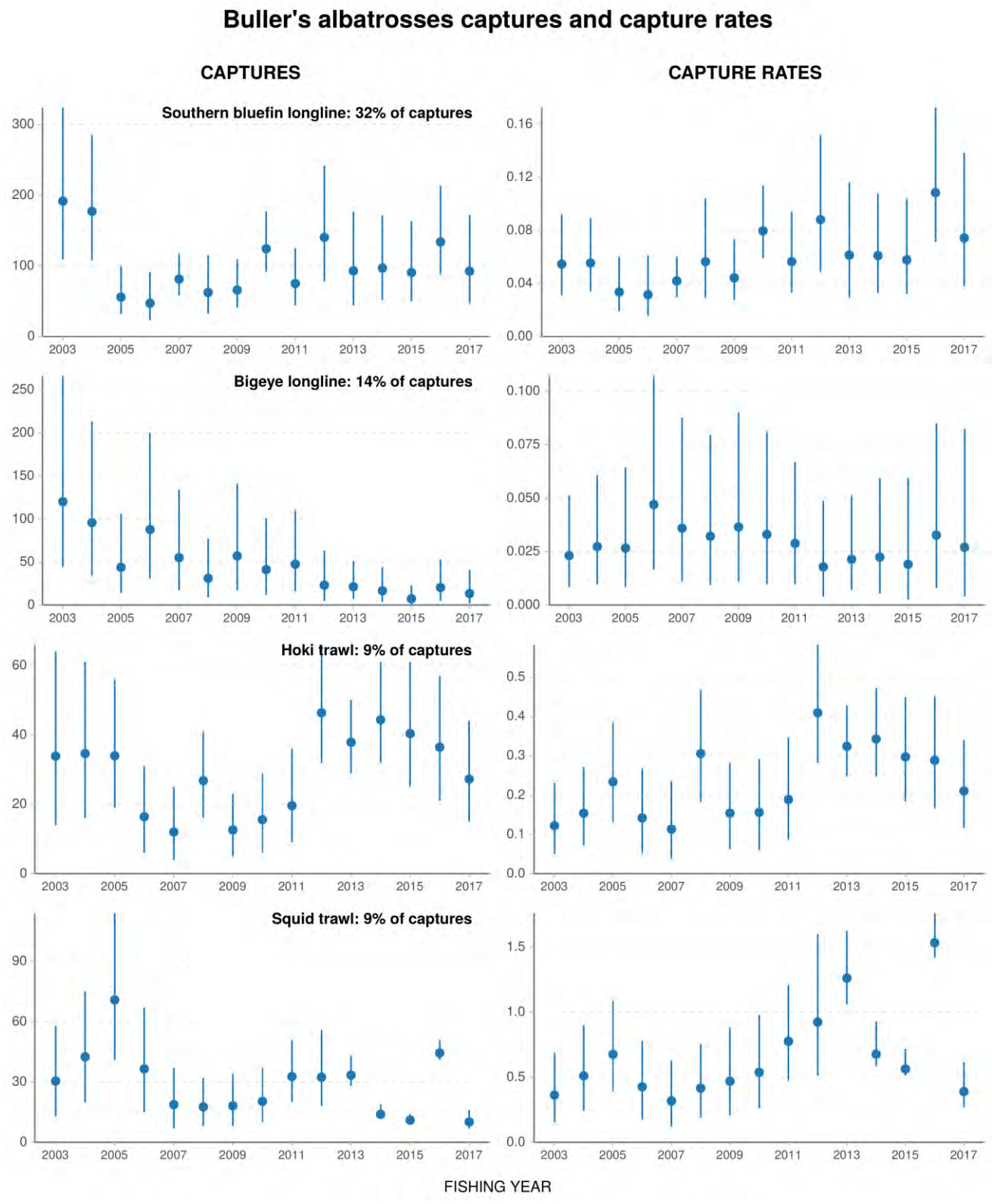


Figure 8.10: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of Buller's albatross in the four fisheries estimated to have taken the most captures between 2002–03 and 2016–17. http://data.dragonfly.co.nz/psc. Data version v20180001.

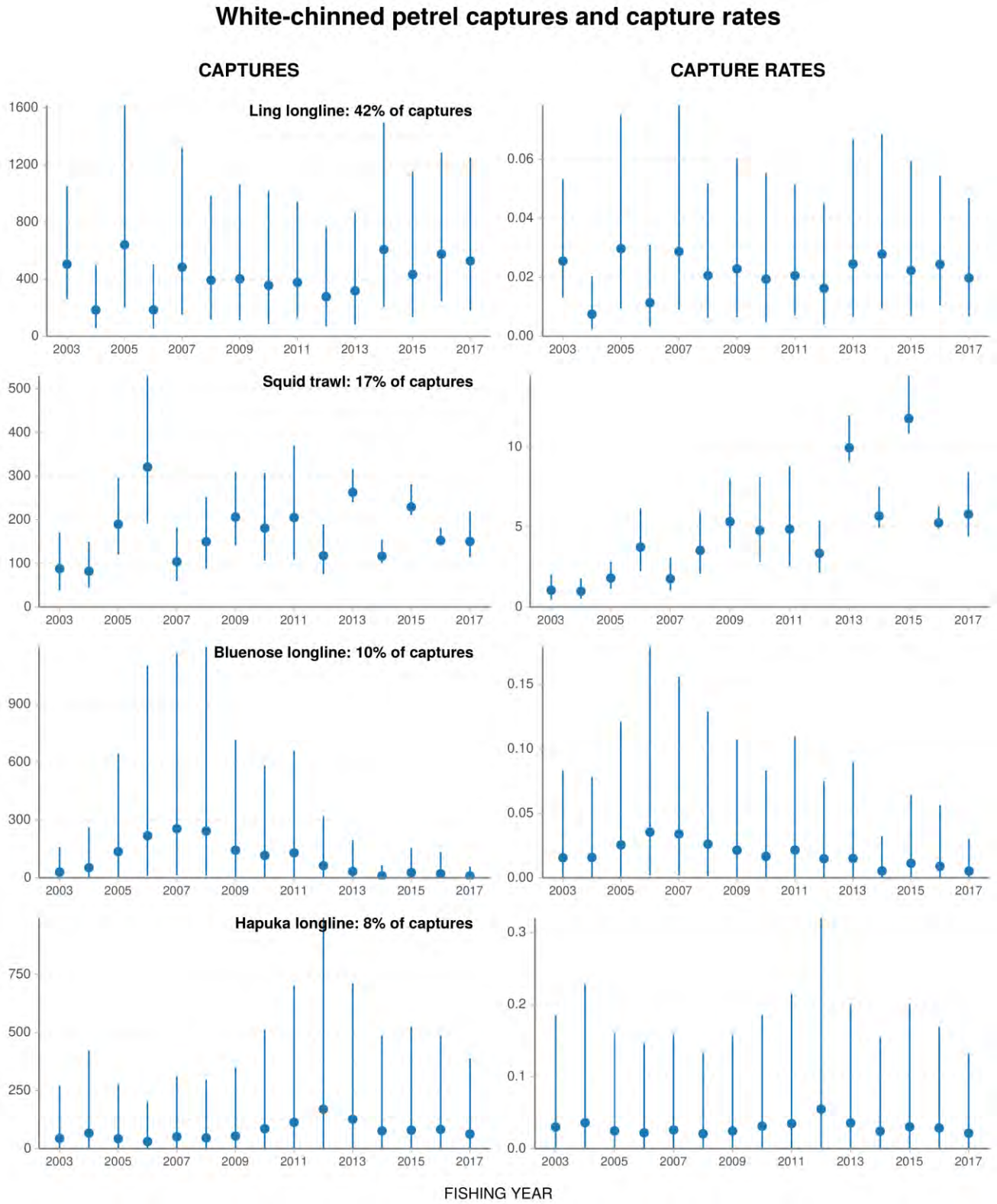


Figure 8.11: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of white-chinned petrels in the four fisheries estimated to have taken the most captures between 2002–03 and 2016–17. http://data.dragonfly.co.nz/psc. Data version v20180001.

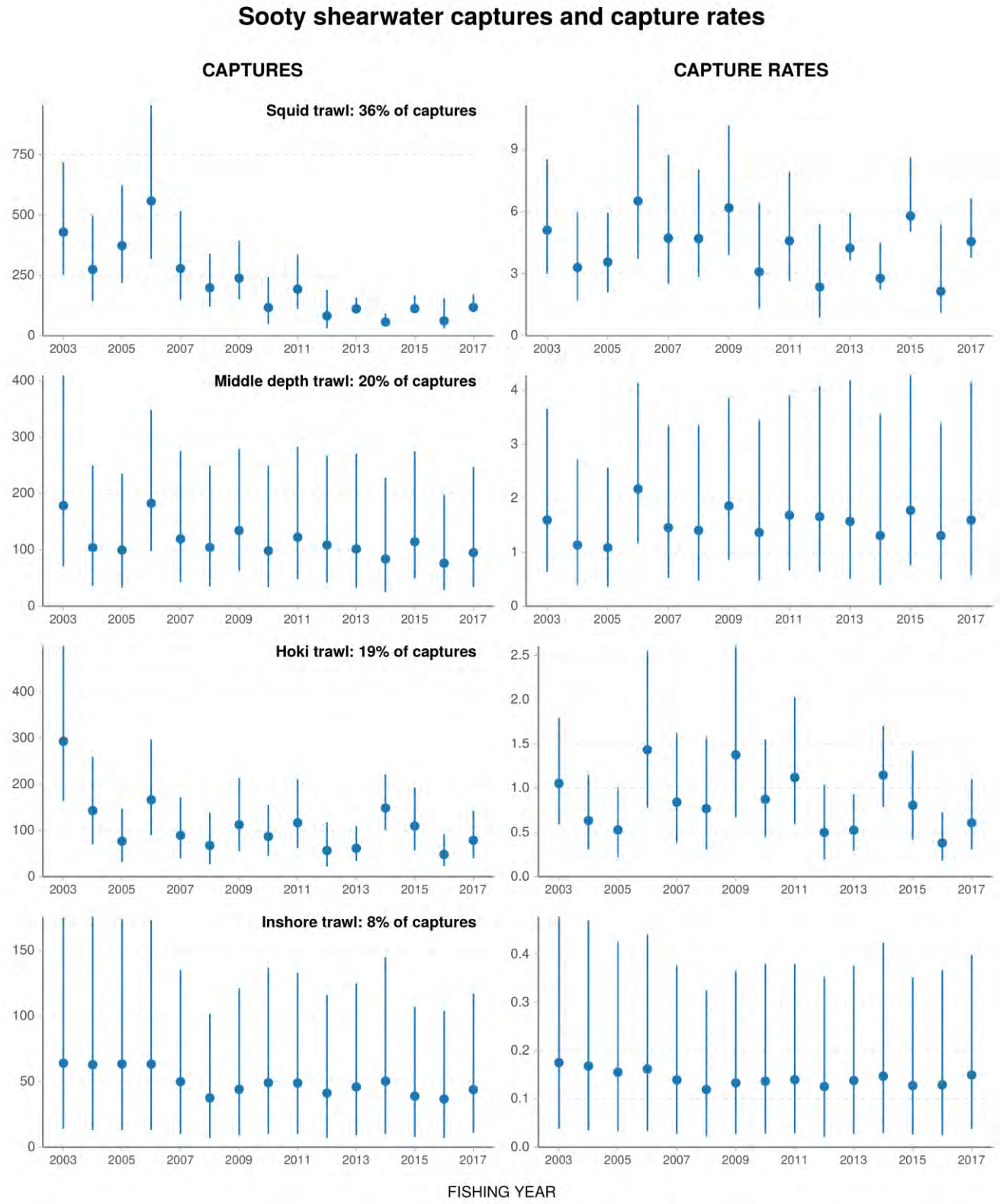


Figure 8.12: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of sooty shearwaters in the four fisheries estimated to have taken the most captures between 2002–03 and 2016–17. http://data.dragonfly.co.nz/psc. Data version v2018001.

8.4.2 MODELLING FISHERIES INTERACTIONS AND ESTIMATING RISK

8.4.2.1 HIERARCHICAL STRUCTURE OF RISK ASSESSMENTS

Hobday et al. (2007) described a hierarchical framework for ecological risk assessment in fisheries (see Figure 8.13). The hierarchy included three levels: Level 1 qualitative, expert-based assessments (often based on a Scale, Intensity, Consequence Analysis, SICA); Level 2 semi-quantitative analysis (often using some variant of Productivity Susceptibility Analysis, PSA); and Level 3 fully quantitative modelling including uncertainty analysis. The hierarchical structure is designed to 'screen out' potential effects that pose little or low risk for the least investment in data collection and analysis, escalating to risk treatment or higher levels in the hierarchy only for those potential effects that pose non-negligible risk. This structure relies for its effectiveness on a low potential for false negatives at each stage, thereby identifying and screening out activities that are 'low risk' with high certainty. This focuses effort on remaining higher-risk activities. In statistical terms, risk assessment tolerates Type I errors (false positives, i.e., not screening out activities that may actually present a low risk) in order to avoid Type II errors (false negatives, i.e., incorrectly screening out activities that actually constitute high risk), and it is important to distinguish this approach from normal estimation methods. Whereas normal estimation strives for a lack of bias and a balance of Type I and Type II errors, risk assessment is designed to answer the question 'how bad could it be?' The divergence between the risk assessment approach and normal, unbiased estimation approaches should diminish at higher levels in the risk assessment hierarchy, where the assessment process should be informed by good data that support robust estimation.

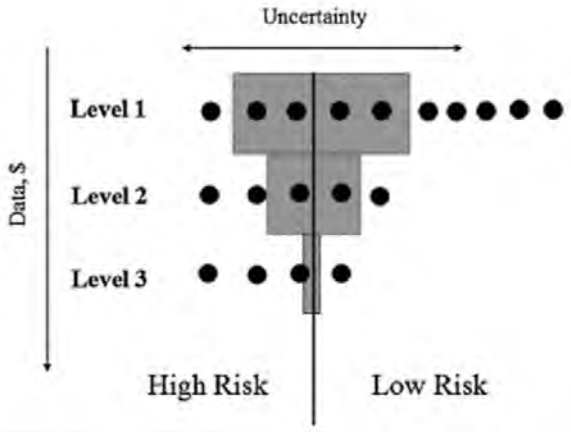


Figure 8.13: (from Hobday et al. 2007). Diagrammatic representation of the hierarchical risk assessment process where activities that present low risk are progressively screened out by assessments of increasingly high data content, sophistication and cost.

8.4.2.2 QUALITATIVE (LEVEL 1) RISK ASSESSMENT

Rowe (2013) summarised an expert-based, qualitative (Level 1) risk assessment, commissioned by DOC, for the incidental mortality of seabirds caused by New Zealand fisheries. The main focus was on fisheries operating within the NZ EEZ and on all seabirds absolutely or partially protected under the Wildlife Act 1953. New Zealand flagged vessels fishing outside the EEZ were included, but risk from non-New Zealand fisheries and other human causes were not included.

The panel of experts who conducted the Level 1 risk assessment assessed the threat to each of 101 taxa posed by 26 fishery groups, scoring exposure and consequence independently (details in Rowe 2013). The risk for a given taxon posed by a given fishery was calculated as the product of exposure and consequence scores. Potential risk was estimated as the risk posed by a fishery assuming no mitigation was in place, and residual risk (called 'optimum

risk' by Rowe 2013) was estimated assuming that mitigation was in place throughout a given fishery and deployed correctly. The panel also agreed a confidence score for each taxon-fishery interaction.

Total potential and residual risk for a seabird taxon was estimated by summing the scores across all fisheries (Table 8.17) shows taxa with an aggregate score of 30 or higher), and total potential and residual risk posed by a fishery group was estimated by summing the scores across all seabird taxa (Table 8.18 shows the results for all 26 fishery groups).

White-chinned petrel, sooty shearwater, black petrel, Salvin's albatross, white-capped albatross, and flesh-footed shearwater were all estimated by this procedure to have an aggregate risk score of 90 or higher (range 92 to 123) even if mitigation was in place and deployed properly across all fisheries. Of the 101 seabird taxa considered, the aggregate risk score was less than 30 for 70 taxa with respect to potential risk and for 72 taxa with respect to residual risk.

Table 8.17: Potential and residual risk scores for each seabird taxon with a potential risk score of 30 or more in Rowe (2013). Residual risk ('optimal risk' in Rowe 2013, not tabulated therein for grey-faced petrel or light-mantled albatross) is estimated assuming mitigation is deployed and correctly used throughout all interacting fisheries.

Taxon	Potential score	Residual score	Percent reduction
White-chinned petrel	159	123	23
Sooty shearwater	126	108	14
Black petrel	139	106	24
Salvin's albatross	161	106	34
White-capped albatross	141	94	33
Flesh-footed shearwater	117	92	21
Southern Buller's albatross	123	85	31
Grey petrel	123	84	32
Black-browed albatross	114	80	30
Northern Buller's albatross	107	72	33
Chatham albatross	114	71	38
Campbell albatross	97	66	32
Westland petrel	89	59	34
Antipodean albatross	89	55	38
Gibson's albatross	89	55	38
Wandering albatross	89	55	38
Southern royal albatross	79	49	38
King shag	48	48	0
Pitt Island shag	46	46	0
Chatham Island shag	45	45	0
Hutton's shearwater	37	35	5
Northern giant petrel	62	35	44
Pied shag	35	35	0
Indian yellow-nosed albatross	58	34	41
Southern giant petrel	61	34	44
Fluttering shearwater	34	32	6
Spotted shag	31	31	0
Stewart Island shag	31	31	0
Yellow-eyed penguin	30	30	0
Grey-faced petrel	31	–	–
Light-mantled albatross	30	–	–

Set-net and inshore trawl fisheries groups posed the greatest residual risk to seabirds (summed across all taxa); both had aggregate scores of over 200 and had no substantive mitigation. Surface- and bottom-longline fisheries and middle-depth trawl fisheries for finfish and squid also had aggregate risk scores of 100 or more. These

risk scores were substantially reduced if mitigation was assumed to be deployed throughout these fisheries (reductions of 24 to 56%), but all remained above 100. Trawling for southern blue whiting and deepwater species, inshore drift net, various seine methods, ring net, diving, dredging and hand gathering all had aggregate risk scores of 40 or less if mitigation was assumed to be deployed throughout these fisheries. Diving, dredging and hand gathering were all judged by the panel to pose essentially no risk to seabirds.

Table 8.18: Cumulative potential risk and residual risk scores across all seabird taxa for each fishery from Rowe (2013). Residual risk ('optimal risk' in Rowe 2013) is estimated assuming mitigation is deployed and correctly used throughout a given fishery.

Fishery group	No. taxa	Potential risk	Residual risk	Percent reduction
Set net	42	374	374	0
Inshore trawl	44	225	225	0
Surface longline: charter	25	313	191	39
Surface longline: domestic	25	302	184	39
Bottom longline: small	33	354	154	56
Bottom longline: large	32	311	139	55
Mid-depth trawl: finfish	22	160	122	24
Mid-depth trawl: squid	21	156	118	24
Mid-depth trawl: scampi	23	94	94	0
Hand line	27	68	68	0
Squid jig	44	62	62	0
Dahn line	29	61	61	0
Pots, traps	17	61	61	0
Trot line	29	61	61	0
Pelagic trawl	27	63	51	19
Troll	23	50	50	0
Mid-depth trawl: southern blue whiting	21	53	40	25
Deepwater trawl	21	46	35	24
Inshore drift net	12	33	33	0
Danish seine	15	32	32	0
Beach seine	16	29	29	0
Purse seine	11	22	22	0
Ring net	12	13	13	0
Diving	0	0	0	–
Dredge	0	0	0	–
Hand gather	0	0	0	–

8.4.2.3 SEABIRD SPATIALLY EXPLICIT FISHERIES RISK ASSESSMENT

The Spatially Explicit Fisheries Risk Assessment (SEFRA) approach used by MPI was developed first for measuring the risk to multiple seabird species starting in 2009. See Chapter 3 for more details.

The SEFRA method developed by MPI is a generalisation of the spatial overlap approach described by Kirby & Hobday (2007) and applies the 'exposure-effects' approach, where exposure refers to the number of fatalities arising from an activity, and effect refers to the consequence of that exposure for the population. The SEFRA approach arose initially from an expert workshop hosted by the then Ministry of Fisheries in 2008 and attended by experts with specialist knowledge of New Zealand fisheries, seabird-fishery interactions, seabird biology, population modelling, and ecological risk assessment. The overall framework is described in Sharp et al. (2011) and has been variously applied and improved in multiple iterations for seabirds (Waugh et al. 2008a, 2008b, developed further by Sharp 2009, Waugh & Filippi 2009, Filippi et al. 2010, Richard et al. 2011, Richard & Abraham 2013b, Richard & Abraham 2015, Richard et al. 2017). The latest iteration of the risk assessment, in draft form, was reported by Abraham & Richard (2018).

Previous versions of this chapter, together with the references cited above, contain considerable detail about the evolution and refinement of the risk assessment approach and the reader is directed to those sources for a

comprehensive record of the risk assessment framework development, and its outputs, over time. Here, the most recent outputs and summary details are provided, as reported by Abraham & Richard (2018).

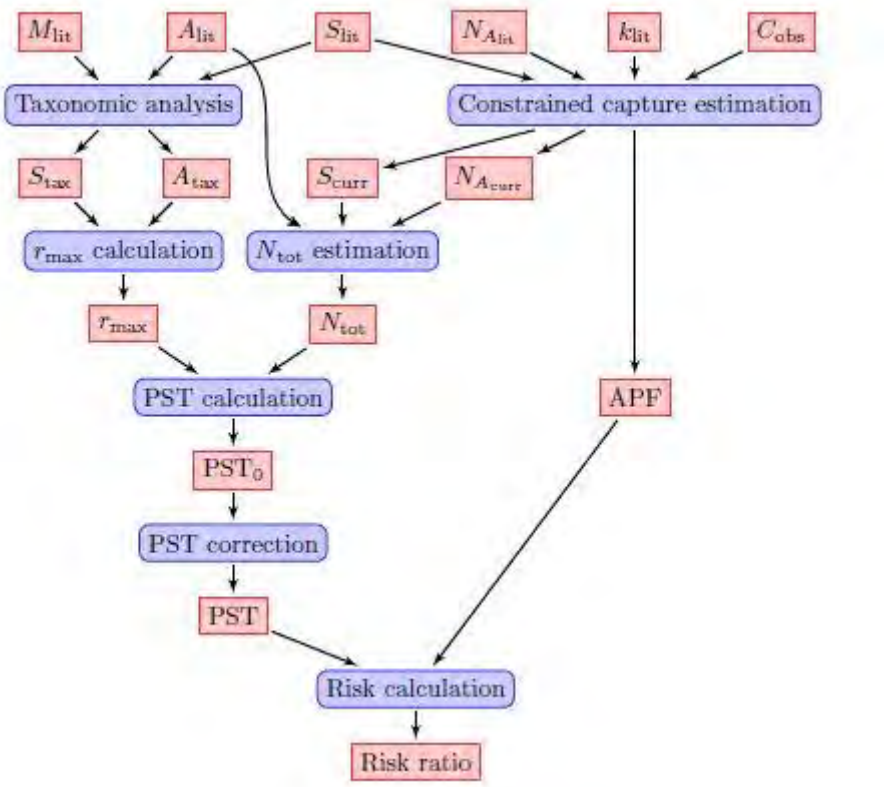


Figure 8.14: Schematic process of the estimation of risk in the current seabird risk assessment. M : body mass; A: age at first reproduction; S: adult survival rate; NA: adult population size; k: cryptic mortality multiplier; C: seabird captures; rmax: maximum net productivity rate; Ntot: total population size; PST: Population Sustainability Threshold; APF: annual potential fatalities. For the indices: lit: from the literature or expert-based; obs: recorded by observers; tax: from the taxonomic analysis; curr: representing current conditions, corrected by the model; tot: total; 0: prior to correction.

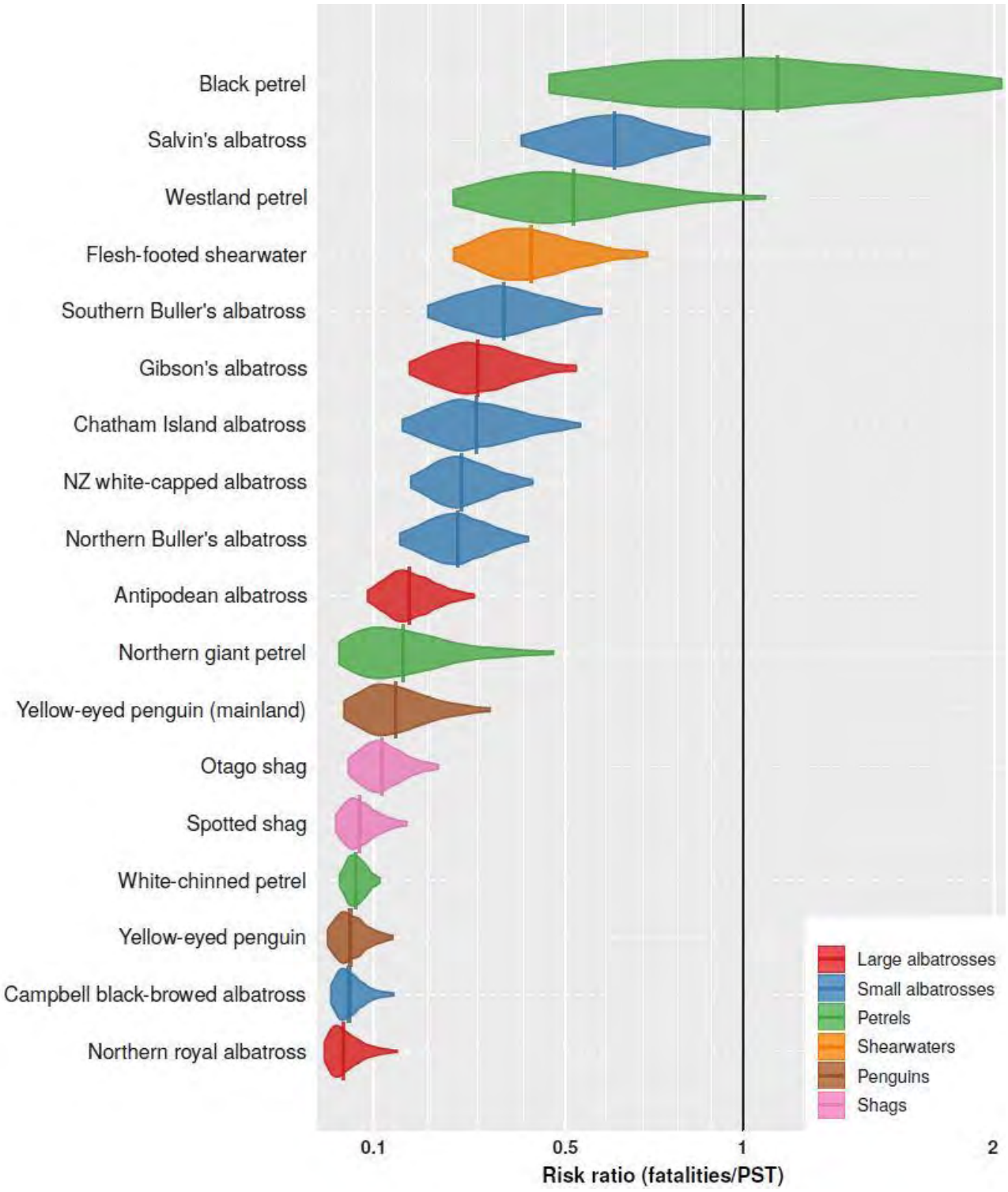


Figure 8.15: Risk ratio for different seabird taxa, based on data between 2006–07 and 2016–17. The risk ratio is displayed on a logarithmic scale, with the threshold of the number of potential bird fatalities equalling the Population Sustainability Threshold (PST) represented by the black vertical line, and the distribution of the risk ratios within their 95% credible interval indicated by the coloured shapes, including the median risk ratio (vertical line). Seabird taxa are listed in decreasing order of the median risk ratio. Taxa with a risk ratio of almost zero were not included (95% upper limit less than 0.05). The risk ratio of yellow-eyed penguin refers to the mainland population only, based on the assumption that all estimated fatalities were of the mainland population, and the number of annual breeding pairs in 2016/17 was between 273 and 374 (YEP Trust pers. comm.).

Table 8.19: Population Sustainability Threshold (PST), total annual potential fatalities (APF) in trawl, longline, and set-net fisheries, risk ratio with $f = 1$ (RR = APF/PST), and the probability that APF > PST for seabird taxa in the current risk assessment. Taxa are ordered in decreasing order of the median risk ratio. The risk to yellow-eyed penguin was assessed for the entire New Zealand population, but also for the mainland population only, based on the assumption that all estimated fatalities were of the mainland population, and the number of annual breeding pairs between 600-800. Taxa names are coloured according to their risk category. Red: risk ratio with a median over 1 or upper 95% credible limit (u.c.l.) over 2; dark orange: median over 0.3 or u.c.l. over 1; light orange: median over 0.1 or u.c.l. over 0.3; yellow: u.c.l. over 0.1. PST and APF values were rounded to three significant digits.

	PST		APF		Risk ratio		P(APF > PST)
	Mean	95% c.i.	Mean	95% c.i.	Median	95% c.i.	
Black petrel	433	204–841	449	275–721	1.11	0.46–2.04	0.60
Salvin's albatross	3 450	2 590–4 850	2 110	1 560–2 810	0.62	0.40–0.89	0.00
Westland petrel	349	235–521	185	100–348	0.52	0.25–1.07	0.03
Flesh-footed shearwater	1 440	1 010–2 000	610	413–904	0.42	0.25–0.71	0.00
Southern Buller's albatross	1 370	898–2 170	476	353–645	0.36	0.20–0.59	0.00
Gibson's albatross	498	329–741	149	95–216	0.30	0.16–0.53	0.00
Chatham Island albatross	425	295–627	128	72–204	0.30	0.15–0.54	0.00
NZ white-capped albatross	10 900	7 650–15 600	2 900	2 140–3 960	0.27	0.17–0.42	0.00
Northern Buller's albatross	1 630	1 070–2 610	407	319–514	0.26	0.15–0.41	0.00
Antipodean albatross	370	256–522	62	37–94	0.16	0.09–0.29	0.00
Northern giant petrel	344	160–806	51	17–114	0.15	0.04–0.47	0.00
Yellow-eyed penguin (mainland)	120	79–181	17	6–34	0.14	0.05–0.33	0.00
Otago shag	283	182–426	33	17–52	0.11	0.06–0.22	0.00
Spotted shag	3 690	1 810–6 760	263	172–378	0.07	0.03–0.16	0.00
White-chinned petrel	25 700	16 000–41 100	1 700	1 410–2 020	0.07	0.04–0.11	0.00
Yellow-eyed penguin	285	190–425	17	6–34	0.06	0.02–0.13	0.00
Campbell black-browed albatross	2 000	992–3 580	111	63–191	0.06	0.03–0.14	0.00
Northern royal albatross	730	345–1 420	35	13–76	0.05	0.01–0.14	0.00
Foveaux shag	207	131–314	6	2–13	0.03	0.01–0.07	0.00
Grey petrel	5 450	3 150–9 100	141	86–218	0.03	0.01–0.05	0.00
Southern royal albatross	853	607–1 180	22	9–40	0.02	0.01–0.05	0.00
Snares Cape petrel	1 570	612–3 910	23	5–65	0.01	0.00–0.06	0.00
Northern little penguin	1 500	913–2 340	12	3–24	0.01	0.00–0.02	0.00
Fluttering shearwater	35 700	15 000–72 900	224	112–383	0.01	0.00–0.02	0.00
White-flippered little penguin	469	276–748	3	0–8	0.01	0.00–0.02	0.00
Little black shag	340	155–658	2	0–9	0.01	0.00–0.03	0.00
Pied shag	1 110	704–1 670	7	0–23	0.00	0.00–0.02	0.00
Grey-headed albatross	697	355–1 230	5	0–21	0.00	0.00–0.03	0.00
Fiordland crested penguin	628	294–1 200	4	0–15	0.00	0.00–0.03	0.00
Southern little penguin	1 500	915–2 360	6	1–14	0.00	0.00–0.01	0.00
Common diving petrel	136 000	47 200–306 000	341	59–1 170	0.00	0.00–0.01	0.00
Grey-faced petrel	30 300	19 100–51 600	60	25–111	0.00	0.00–0.00	0.00
Sooty shearwater	612 000	287 000–1 190 000	1 110	634–2 020	0.00	0.00–0.00	0.00
Light-mantled sooty albatross	874	668–1 140	2	0–12	0.00	0.00–0.01	0.00
Hutton's shearwater	14 900	9 190–23 000	16	2–59	0.00	0.00–0.00	0.00
Chatham Island little penguin	1 520	928–2 420	1	0–8	0.00	0.00–0.01	0.00
White-headed petrel	34 500	16 400–67 300	9	2–18	0.00	0.00–0.00	0.00
NZ white-faced storm petrel	328 000	140 000–678 000	86	17–245	0.00	0.00–0.00	0.00
Buller's shearwater	55 800	34 100–99 800	12	4–27	0.00	0.00–0.00	0.00
Australasian gannet	9 490	4 280–19 400	3	0–12	0.00	0.00–0.00	0.00
Little shearwater	21 800	13 700–33 300	4	0–9	0.00	0.00–0.00	0.00
Fairy prion	325 000	211 000–490 000	77	11–310	0.00	0.00–0.00	0.00
Snares crested penguin	6 870	4 770–9 580	1	0–5	0.00	0.00–0.00	0.00
Southern black-backed gull	336 000	138 000–720 000	46	15–101	0.00	0.00–0.00	0.00
Broad-billed prion	68 500	44 700–104 000	8	1–27	0.00	0.00–0.00	0.00
Black-bellied storm petrel	15 500	8 770–26 200	2	0–9	0.00	0.00–0.00	0.00
Cook's petrel	48 600	27 900–86 600	7	0–39	0.00	0.00–0.00	0.00
Antarctic prion	155 000	75 800–281 000	9	2–25	0.00	0.00–0.00	0.00
Erect-crested penguin	17 700	12 500–24 700	1	0–4	0.00	0.00–0.00	0.00
Mottled petrel	47 100	30 400–78 100	4	0–25	0.00	0.00–0.00	0.00
Auckland Island shag	482	204–974	0	0–1	0.00	0.00–0.00	0.00
Bounty Island shag	26	14–44	0	0–0	0.00	0.00–0.00	0.00
Subantarctic skua	67	44–100	0	0–0	0.00	0.00–0.00	0.00
Caspian tern	171	95–286	0	0–0	0.00	0.00–0.00	0.00
Chatham Island shag	76	47–117	0	0–3	0.00	0.00–0.03	0.00
Campbell Island shag	480	228–925	0	0–0	0.00	0.00–0.00	0.00
Eastern rockhopper penguin	11 100	6 770–17 500	1	0–3	0.00	0.00–0.00	0.00
White-bellied storm petrel	229	106–428	0	0–0	0.00	0.00–0.00	0.00
White tern	26	15–43	0	0–0	0.00	0.00–0.00	0.00
South Georgian diving petrel	10	5–19	0	0–1	0.00	0.00–0.07	0.00
NZ king shag	39	24–61	0	0–2	0.00	0.00–0.05	0.00
Kerm. storm petrel	11	4–25	0	0–0	0.00	0.00–0.00	0.00
Masked booby	53	29–92	0	0–0	0.00	0.00–0.00	0.00
NZ storm petrel	53	6–200	0	0–1	0.00	0.00–0.02	0.00
Pitt Island shag	104	63–164	0	0–2	0.00	0.00–0.02	0.00
Chatham petrel	41	23–74	0	0–0	0.00	0.00–0.00	0.00
Chatham Island taiko	2	1–4	0	0–0	0.00	0.00–0.00	0.00
Pycroft's petrel	413	243–732	0	0–1	0.00	0.00–0.00	0.00
Soft-plumaged petrel	499	135–1 320	0	0–0	0.00	0.00–0.00	0.00
Wedge-tailed shearwater	5 980	3 060–10 500	0	0–0	0.00	0.00–0.00	0.00
Kerm. petrel	779	499–1 330	0	0–1	0.00	0.00–0.00	0.00
White-naped petrel	7 010	3 210–13 500	0	0–0	0.00	0.00–0.00	0.00

Table 8.20: Mean species-level risk estimates for seabird taxa, disaggregated by target fisheries. Highlighted cells (increasing red) identify fisheries that are responsible for an increasing proportion of species-level risk. Species are ordered by total risk ratio in descending order, and fisheries are ordered with the fishery with highest total risk ratio on the left. Target fisheries with zero risk to all species (rounded to two decimal places) are not shown; these include: albacore SLL, minor SLL, jack mackeral trawl, and grey mullet set net. Likewise, species for which mean total risk ratio rounds to zero are not shown.

Species	Inshore trawl	Hoki trawl	Small STN SLL	Small ling BLL	Swordfish SLL	Bigeye SLL	Flatfish trawl	Snapper BLL	Middle depth trawl	Scampi trawl	Minor BLL	Deepwater trawl	Hapuka BLL	Squid trawl	Bluenose BLL	Shark setnet	Large ling BLL	Ling trawl	Flatfish setnet	Minor setnet	SBW trawl	Hake trawl	Minor SLL	Large STN SLL	TOTAL RISK RATIO
Black petrel	0.54	0.02	0	0	0.04	0.18	0	0.12	0.01	0	0.01	0	0.01	0	0.08	0	0	0	0	0	0	0	0	0	1.11
Salvin's albatross	0.2	0.11	0	0.08	0	0	0.02	0	0.07	0.08	0	0.02	0	0	0	0	0.01	0.01	0	0	0.01	0	0	0	0.62
Westland petrel	0.09	0.05	0.14	0.04	0.01	0.02	0.03	0	0.02	0	0.02	0	0.01	0	0	0.01	0	0	0.01	0	0	0	0	0	0.52
Flesh-footed shearwater	0.13	0	0	0	0	0.01	0	0.13	0	0.03	0.04	0	0.02	0	0	0	0	0	0.01	0	0	0	0	0	0.42
Southern Buller's albatross	0.02	0.11	0.06	0.01	0	0	0.01	0	0.05	0.01	0	0	0	0.05	0	0	0	0	0	0	0	0	0	0	0.36
Chatham Island albatross	0	0.01	0	0.17	0	0	0	0	0	0	0.01	0.07	0	0	0	0	0	0	0	0	0	0	0	0	0.3
Gibson's albatross	0	0	0.07	0	0.19	0.02	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.3
NZ white-capped albatross	0.1	0.03	0.01	0	0	0.01	0.04	0	0.02	0.01	0	0	0	0.03	0	0	0	0	0	0	0	0	0	0	0.27
Northern Buller's albatross	0.01	0.03	0.05	0.01	0	0.08	0	0	0.02	0.03	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.26
Antipodean albatross	0	0	0.04	0	0.09	0.01	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.16
Northern giant petrel	0	0.02	0	0	0	0	0	0	0.01	0.01	0.02	0.01	0.03	0	0	0	0	0	0	0	0	0	0	0	0.15
Otago shag	0.01	0	0	0	0	0	0.1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.11
Spotted shag	0.01	0	0	0	0	0	0.05	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.07
White-chinned petrel	0	0.01	0	0.03	0	0	0	0	0	0.01	0	0	0	0.01	0	0	0.01	0	0	0	0	0	0	0	0.07
Campbell albatross	0	0.01	0.01	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.06
Yellow-eyed penguin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.03	0	0	0.01	0.01	0	0	0	0	0.06
Northern royal albatross	0	0	0.01	0.01	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.05
Foveaux shag	0	0	0	0	0	0	0.03	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.03
Grey petrel	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.03
Southern royal albatross	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02
Snares Cape petrel	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01
Northern little penguin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01
White-flippered little penguin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01
Fluttering shearwater	0	0	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01
Little black shag	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01
Fishery group total (all species)	1.23	0.44	0.42	0.41	0.37	0.35	0.34	0.28	0.23	0.2	0.14	0.13	0.12	0.11	0.09	0.06	0.04	0.03	0.03	0.03	0.02	0.02	0.01	0.01	5.26

8.4.2.4 SPECIES-SPECIFIC, FULLY QUANTITATIVE MODELLING

Fully quantitative population modelling has been conducted only for southern Buller's albatross, black petrel, white-capped albatross, Gibson's albatross and Antipodean albatross. Data for other species or populations appear unlikely to be adequate for comprehensive population modelling. The poor estimates of observable and cryptic fishing-related mortality have restricted such work to comprehensive population modelling rather than formal assessment of risk. Summaries of the fully quantitative modelling outlined here are included in section 8.4.3 that focusses on the 13 most at-risk species (see Table 8.19: species with a risk ratio of 0.1 or greater from the most recent iteration of the seabird risk assessment, Abraham & Richard 2018).

8.4.2.5 OTHER QUANTITATIVE MODELS

This section is not intended to cover all quantitative modelling of seabird populations, rather to focus on recent studies that sought to assess the impact of fishing-related mortality.

Fletcher et al. (2008) sought to assess the potential impact of fisheries on Antipodean and Gibson's wandering albatrosses; black petrel and southern royal albatross *Diomedea epomophora*. Because of problems with the available fisheries and biological data, they were unable to use their models to predict the impact of a change in fishing effort on the population growth rate of a given species. Instead, they used the models to estimate the impact that changes in demographic parameters like annual survival are likely to have on population growth rate. They found that: reducing breeder survival rate by k percentage points will lead to a reduction in the population growth rate of about 0.3k percentage points (0.4 for black petrel); and a reduction of k percentage points in the survival rate for each stage in the lifecycle (juvenile, pre-breeder, non-breeder and breeder) will lead to a reduction in the population growth rate of approximately k percentage points. Fletcher et al. (2008) also made estimates of PBR for 23 New Zealand seabird taxa and summarised and tabulated non-fishing-related threats for 38 taxa.

Newman et al. (2009) combined survey data with demographic population models to estimate the total population of sooty shearwaters within New Zealand. They estimated the total New Zealand population between 1994 and 2005 to have been 21.3 (95% c.i.: 19.0–23.6) million birds. The harvest of 'muttonbirds' was estimated to be 360 000 (320 000–400 000) birds per year, equivalent to 18% of the chicks produced in the harvested areas and 13% of chicks in the New Zealand region. This directed harvest is much larger than estimates of captures in key fisheries or potential fatalities in the Level 2 risk assessment. Newman et al. (2009) did not assess the likely impact of fishing-related mortality and did not consider the different population-level impacts of adult mortality in fisheries and chick mortality in the directed harvest but concluded that the much larger directed harvest was not an adequate explanation for the observed declines in the past three decades.

8.4.2.6 GENERAL CONCLUSIONS FROM QUANTITATIVE MODELLING

Fully quantitative modelling has now been conducted for five seabird populations for which apparently suitable data are available (see individual species sections below). This modelling suggests very strongly that one population had been increasing steadily (southern Buller's albatross, but note that this trend may have since reversed), while a further population is declining quite rapidly (Antipodean albatross). White-capped albatross and black petrel were both assessed at the time of the modelling to be more likely to be declining than not but, even for these relatively data-rich populations, the conclusions were uncertain. Higher counts have been recorded for both species since the modelling was conducted. General conclusions from the modelling conducted to date, therefore, can be summarised as:

- Very few seabird populations have sufficient data for fully quantitative modelling.
- Except for the most complete datasets (southern Buller's albatross, Gibson's albatross and Antipodean albatross) it has been difficult

to draw firm conclusions about trends in population size from model outputs.

- Information from surveys or census counts is much more powerful for detecting trends in population size than data from the tagging programmes and plot monitoring implemented for New Zealand seabirds to date.
- The available information on incidental captures in fisheries have not allowed rigorous tests of the role of fishing-related mortality in driving population trends.
- Although comprehensive modelling provides additional information to allow interpretation, we will have to rely on Level 2 risk assessment approaches for much of our understanding of the relative risks faced by different seabird taxa and posed by different fisheries.

8.4.2.7 SOURCES OF UNCERTAINTY IN RISK ASSESSMENTS

There are several outstanding sources of uncertainty in modelling the effects of fisheries interactions on seabirds, especially for the complete assessment of risk to individual seabird populations.

8.4.2.7.1 SCARCITY OF INFORMATION ON CAPTURES AND BIOLOGICAL CHARACTERISTICS OF AFFECTED POPULATIONS

These sources of uncertainty can be explored within the analytical framework of the Level 2 risk assessment (Richard et al. 2011, Richard & Abraham 2013b, 2015), noting that the results of that exploration are constrained by the structure of that analysis. Richard & Abraham (2015) provided plots of such an exploration for nine taxa (Figure 8.16). It can be concluded from this analysis that better estimates of average adult survival would lead to substantially more precise estimates of risk for a wide variety of taxa, including most of the species estimated to be at most risk. More precise estimates of risk would be available for black petrel, Salvin's albatross, New Zealand white-capped albatross, Chatham Island albatross, and Antipodean albatross if better estimates of potential fatalities were available, and better estimates of survival would be useful for all nine taxa. This analysis was not applied at this iteration of the risk assessment to the spatial distribution of seabirds and fisheries, although it is acknowledged that this is extremely important for the proper implementation of any spatial overlap method. Noting this limitation, this type of sensitivity analysis is a powerful way of assessing the priorities for collection of new information, including research.

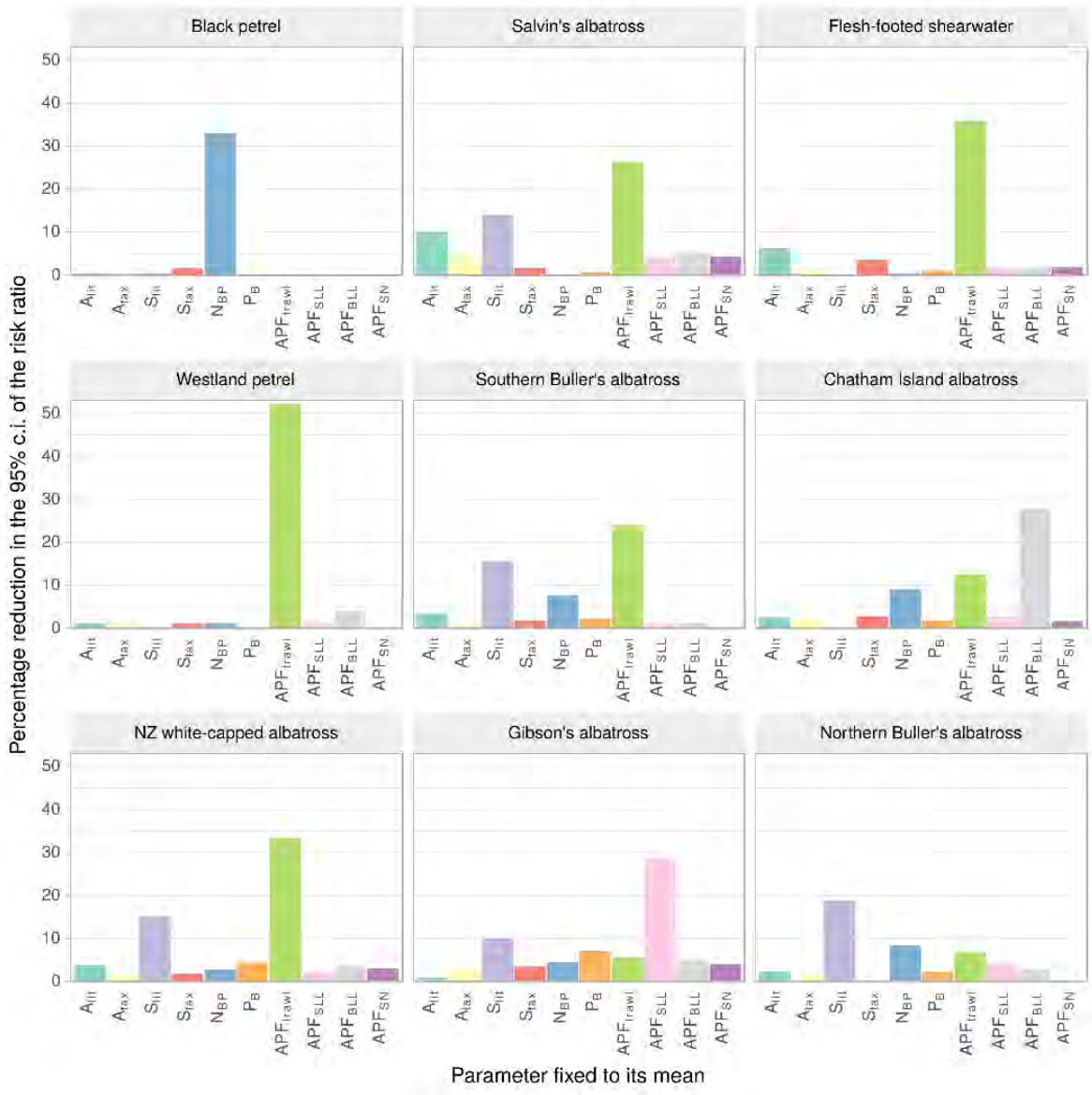


Figure 8.16: (reproduced from Richard et al. 2017). Sensitivity of the uncertainty in the risk ratio for the nine seabird species with the highest risk ratio. For each seabird type, the sensitivity to the uncertainty in the following parameters is considered: annual potential fatalities in trawl, bottom-longline, surface-longline and set-net fisheries (TWL, BLL, SLL, SN, respectively); the cryptic multipliers (CM); age at first reproduction (A); adult survival (SA); the number of annual breeding pairs (N_{BP}); and the proportion of adults breeding (PB). The sensitivity is defined as the percentage of reduction in the 95% confidence interval of the risk ratio that occurs when the parameter is set to its arithmetic mean.

8.4.2.7.2 SCARCITY OF INFORMATION ON CRYPTIC MORTALITY

Cryptic mortality is particularly poorly understood but has substantial influence on the results of the risk assessment. Richard et al. (2011) provided a description of the method used to incorporate cryptic mortality into their estimates of potential fatalities in the Level 2 risk assessment (their appendix B authored by B. Sharp, MPI). This method builds on the published information from Brothers et al. (2010) for longline fisheries and Watkins et al. (2008) and Abraham (2010a) for trawl fisheries. Brothers et al. (2010) observed almost 6000 seabirds attempting to take longline baits during line setting, of which 176 (3% of attempts) were seen to be caught. Of these, only 85 (48%) were retrieved during line hauling. They concluded that using only observed captures to estimate seabird fatalities grossly

underestimates actual levels in pelagic longline fishing. Similarly, Watkins et al. (2008) observed 2454 interactions between seabirds and trawl warps in the South African hake fishery over 189.8 hours of observation. About 11% of those interactions (263) involved birds, mostly albatrosses, being dragged under the water by the warps, and 30 of those submersions were observed to be fatal. Of the 30 birds observed killed on the warps, only two (both albatrosses) were hauled aboard and would have been counted as captures by an observer in New Zealand. Aerial collisions with the warps were about eight times more common but appeared mostly to have little effect (although one white-chinned petrel suffered a broken wing, which would almost certainly have fatal consequences). Parker et al. (2013) presented some preliminary data on cryptic mortality associated with a fishing vessel operating around the Falkland Islands in the south Atlantic Ocean. Of a total of 2250 contacts between seabirds (almost all black-browed albatross *Thalassarche melanophris* and northern giant petrel *Macronectes halli*) and warp cables or bird scaring lines, 371 (17%) were considered heavy contacts, and of these 26 (7%) were recorded as being of unknown outcome. Overall, Parker et al. (2013) estimated that at least 23% of total mortalities (including severe injuries that were deemed to ultimately be fatal) recorded were not observed from the fishing vessel.

Given the relatively small sample sizes in both of these trials, there is substantial (estimable) uncertainty in the estimates from the trials themselves and additional (non-estimable) uncertainty related to the extent to which these trials are representative of all fishing of a given type, particularly as both trials were undertaken overseas. The binomial 95% confidence range (calculated using the Clopper-Pearson 'exact' method) for the ratio of total fatalities to observed captures in Brothers et al.'s (2010) longline trial is 1.8–2.5 (mean 2.1), and that for Watkins et al.'s trawl warp trial is 5–122 (mean 15.0 fatalities per observed capture). Abraham (2010a) estimated that there were 244 (95% c.i.: 190–330) warp strikes by large birds for every one observed captured, and 6440 (3400–20 000) warp strikes by small birds for every one observed captured (although small birds tend to be caught in the net rather than by warps). There is also uncertainty in the relative frequencies and consequences of different types of encounters with trawl warps in New Zealand fisheries (Abraham 2010a, Richard et al. 2011 Appendix B). Some of this uncertainty is included and propagated in the most recent published risk assessment (Richard et al. 2017).

A review of available information on cryptic mortality was commissioned under CSP project INT2013-05 and supported by MPI project PRO2012-17 (Pierre et al. 2015). Pierre et al. (2015) recommended four 'nest steps' to progress the improvement of cryptic mortality scalars as applied to New Zealand fisheries:-

- Amend the definition of cryptic mortality applied in New Zealand, such that the definition in use is better aligned with international approaches. Pierre et al. (2015) suggested the following definition: 'seabird mortalities that are unobserved or unobservable and directly or indirectly result from interactions with fishing gear or fishing operations'.

- Examine existing datasets identified in this report, that are available internationally and in New Zealand, to improve estimates of cryptic mortality for New Zealand species, or species groups, caught in surface longline and trawl fisheries.

- Amend data collection protocols used by New Zealand fisheries observers such that potential cryptic mortalities will be documented routinely, and,

- Develop a data collection programme to support the estimation of method-specific scalars for bottom longline fisheries, especially vessels less than 34 m in overall length.

8.4.2.7.3 MORTALITIES IN NON-COMMERCIAL FISHERIES

Little is known about the nature and extent of incidental captures of seabirds in non-commercial fisheries, either in New Zealand or globally (Abraham et al. 2010a). In New Zealand, participation in recreational fishing is high and 2.5% of the adult population are likely to be fishing in a given week (mostly using rod and line). Because of this high participation rate, even a low rate of interactions between individual fishers and seabirds could have population-level

impacts. A boat ramp survey of 765 interviews at two locations during the summer of 2007–08 revealed that 47% of fishers recalled witnessing a bird being caught some time in the past. Twenty-one birds were reported caught on the day of the interview at a capture rate of 0.22 (95% c.i.: 0.13–0.34) birds per 100 hours of fishing. Observers on 57 charter trips recorded seabird captures at rate of 0.36 (0.09–0.66) birds per 100 fisher hours. The most frequently reported type of bird caught in rod and line fisheries were petrels and gulls. Captures of albatrosses, shags, gannets, penguins and terns were also recalled.

The ramp surveys reported by Abraham et al. (2010a) were limited and covered only two widely separated parts of the New Zealand coastline. However, they also report two other pieces of information that suggest that non-commercial captures are likely to be very widespread. First, the Ornithological Society of New Zealand's beach patrol scheme records seabird hookings and entanglements as a common occurrence throughout New Zealand. Second, returns of banded birds caught in fisheries (separating commercial and non-commercial fisheries is very difficult) are very widely distributed around the coast (Figure 8.17).

Noting that our understanding of seabird capture rates in amateur fisheries is very sketchy, it is possible to make first-order estimates of total captures using information on fishing effort. For example, in the north-eastern region where most of Abraham et al.'s (2010a) interviews were conducted, there were an estimated 4.8 (4.4–5.2) million fisher hours rod and line fishing from trailer boats in 2004–05 (Hartill et al. 2007). Applying Abraham et al.'s (2010a) capture rate leads to an estimate of 11 500 (6600–17 200) captures per year in this area. Based on estimates of nationwide recreational fishing effort, this could increase to as many as 40 000 bird captures annually. Most birds captured by amateur fishers were reported to have been released unharmed (77% of the incidents recalled) and only three people reported incidents where the bird died. Because of likely recall biases and the qualitative nature of the survey, the fate of birds that are captured by amateur fishers remains unclear.

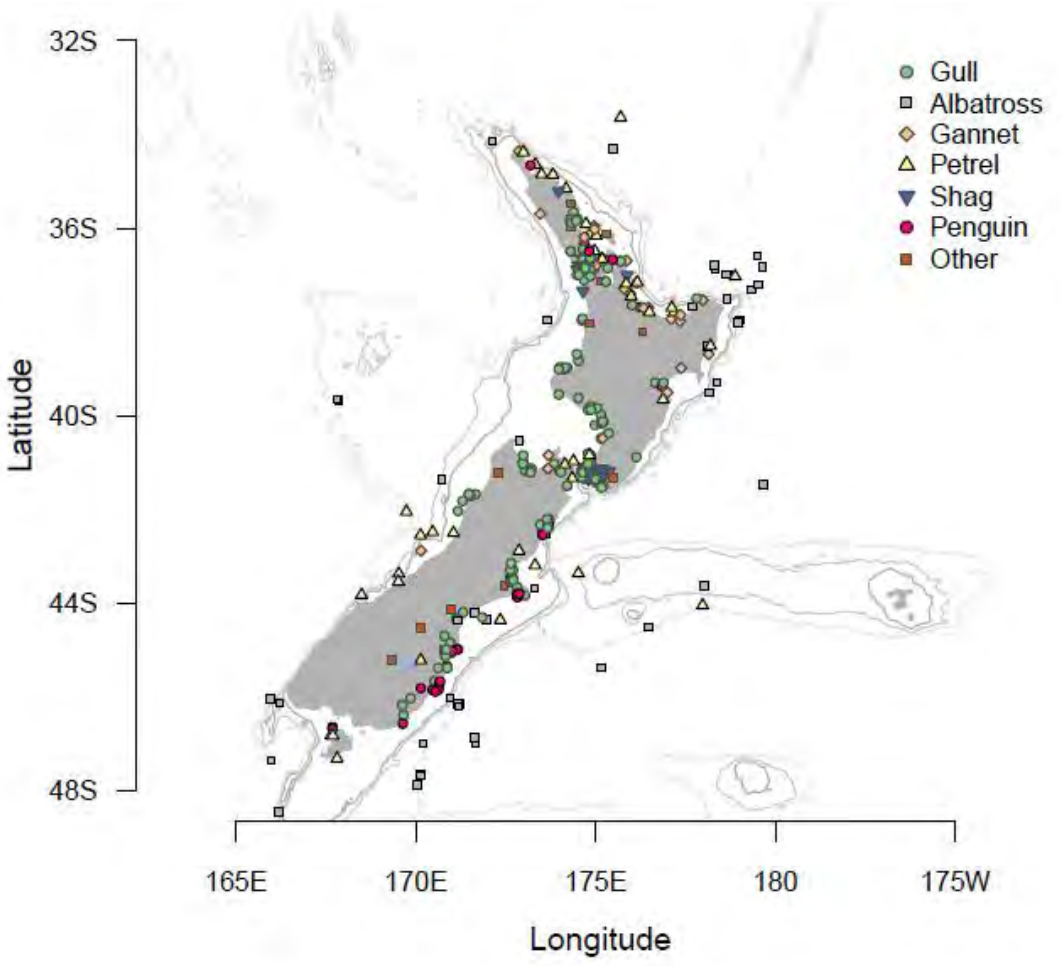


Figure 8.17: (from Abraham et al. 2010a). Distribution of the reported capture locations for banded seabirds reported as being captured in fishing gear, 1952–2007. Note, band recovery locations are reported with low spatial precision and some of the inland locations may be correct.

Non-commercial fishers are allowed to use set nets in New Zealand and two studies suggest that these have an appreciable bycatch of seabirds. A study of captures in non-commercial set nets in Portobello Bay, Otago Harbour, between 1977 and 1985 (Lalas 1991) suggested that spotted shags were the most frequently caught taxa (82 recorded, compared with 14 Stewart Island shags and two little shags). Lalas (1991) suggested that up to 800 spotted shags (20% of the local population) may have been caught in the summer of 1981–82. A broader-scale study of yellow-eyed penguin mortality in set nets in southern New Zealand (Darby & Dawson 2000) suggested non-negligible captures of this species by non-commercial fishers, also reporting other seabirds like spotted shags and little blue penguin.

8.4.2.7.4 OUT OF ZONE MORTALITY

Robertson et al. (2003) mapped the distribution of the 25 breeding (mainly endemic) New Zealand seabird taxa they considered most at risk outside New Zealand waters. These ranged widely: four used the South Atlantic; four the Indian Ocean; 22 Australian waters and the Tasman Sea; 15 used the South Pacific Ocean as far afield as Chile and Peru; and six used the North Pacific Ocean as far north as the Bering Sea. These taxa therefore use the national waters of at least 18 countries. For example, the Level 2 risk assessment described by Richard et al. (2011) includes only that part of the range of each taxon contained within New Zealand waters, but many, including commonly caught seabirds like white-capped albatross and white-chinned petrel, range much further and are vulnerable to fisheries in other parts of the world. For instance, fatalities of white-capped albatross outside the New Zealand EEZ greatly exceed

fatalities within the zone (Baker et al. 2007a, Francis 2012, Table 8.21), and more than 10 000 white-chinned petrel are killed off South America each year (Phillips et al. 2006), noting that reliable records are not available for most of the fisheries involved. Also note that white-chinned petrels also breed on Prince Edward and Falkland Islands, South Georgia, Iles Crozet, and the Kerguelen group, so South American captures may be from other populations other than New Zealand's. Based on similar analyses, Moore & Zydelis (2008) concluded that a population-based, multi-gear and multi-national framework is required to identify the most significant threats to wide-ranging seabird populations and to prioritise mitigation efforts in the most problematic areas. To that end, the Agreement for the Conservation of Albatrosses and Petrels (ACAP) adopted a global prioritisation framework at the Fourth Session of the Meeting of the Parties (MoP4) in April 2012 (ACAP 2012).

Table 8.21: (from Francis 2012). Estimates of the number of white-capped albatrosses killed annually, by fishery. The first two columns are from Baker et al. (2007a) (mid-point where a range was presented), including their assessment of reliability (L = low, M-H = medium-high, H = high). Updated estimates are from Watkins et al. (2008, *) and Petersen et al. (2009, **). Estimates not already corrected for cryptic mortality are either doubled to allow for this (***) or replaced by estimates of potential fatalities from Richard et al. (2011, ****), noting that potential fatalities may considerably overestimate actual fatalities.

Fishery	From Baker et al. 2007a		Updated	Incl. cryptic mortality
South African demersal trawl	4 750	(L)	* 6 650	6 650
Asian distant-water longline	1 255	(L)	–	*** 2 510
Namibian demersal trawl	910	(L)	* 1 270	1 270
Namibian pelagic longline	180	(L)	** 195	*** 390
NZ hoki and squid trawl	513	(MH)	–	**** 4 920
NZ longline	60	(MH)	–	**** 199
Australian (line fisheries)	15	(MH)	–	*** 30
South African pelagic longline	570	(H)	** 570	*** 1 140
Total	8 210	–	–	17 110

8.4.2.7.5 OTHER SOURCES OF ANTHROPOGENIC MORTALITY

Taylor (2000) listed a wide range of threats to New Zealand seabirds including introduced mammals, avian predators (weka), disease, loss of nesting habitat, competition for nest sites, coastal development, human disturbance, commercial and cultural harvesting, volcanic eruptions, pollution, plastics and marine debris, oil spills and exploration, heavy metals or chemical contaminants, global sea temperature changes, marine biotoxins, and fisheries interactions. Relatively little is known about most of these factors, but the parties to ACAP have agreed a formal prioritisation process to address and prioritise major threats (ACAP 2012). Croxall et al. (2012) identified the main priorities as: protection of Important Bird Area (IBA) breeding, feeding, and aggregation sites; removal of invasive, especially predatory, alien species as part of habitat and species recovery initiatives. Lewison et al. (2012) identified similar research priorities (in addition to direct fishing-related mortality), including: understanding spatial ecology; tropho-dynamics; response to global change; and management of anthropogenic impacts such as invasive species, contaminants, and protected areas. Non-fishing-related threats to seabirds in New Zealand are largely the mandate of the Department of Conservation and a detailed description is beyond the scope of this document (although causes of mortality other than fishing are clearly relevant to the interpretation of risk assessment restricted to the direct effects of fishing). These threats are identified in DOC's Action Plan for Seabird Conservation in New Zealand (Taylor 2000) and various Threatened Species Recovery Plans.

8.4.2.8 FUTURE DEVELOPMENT OF THE RISK ASSESSMENT FRAMEWORK

The following steps were identified in the NPOA-seabirds 2013 (MPI 2013) in order to improve the risk assessment framework that supports the implementation of the NPOA-seabirds 2013:

- implementation of a framework and process to consolidate different risk assessment and population monitoring results into an integrated assessment, including:
- checking the algorithmic Level 2 assessment results for particular high-risk species-fishery interactions, in light of other available data or identifiable structural biases on a case-by-case basis,
- a mechanism to incorporate issues associated with seabird mortalities outside the EEZ and recreational fisheries risk in future assessments, and
- the use of species population models or census data to constrain input parameters or interpret estimates of risk;
- routine update of the integrated fisheries risk assessment with relevant new information; and
- periodic review and update of risk management priorities in light of current risk estimates.

8.4.3 AT-RISK SEABIRDS: DEMOGRAPHIC, DISTRIBUTION AND FULLY QUANTITATIVE MODELLING STUDIES

Previous versions of this chapter included species accounts for a smaller selection of species than that presented here. Furthermore, previous species accounts were included for those species that were included as study species in project MPI PRO200601. Here 13 species accounts are included for those seabirds that have a risk ration of 0.1 of greater in the latest iteration of the SEFRA (see Table 8.19. Abraham & Richard 2018), black petrel to Otago shag inclusive (those species coloured-coded red or shades of orange in Table 8.19)

8.4.3.1 BLACK PETREL

Black petrel is a medium-sized endemic seabird with breeding confined to Great Barrier and Little Barrier islands in the outer Hauraki Gulf. Black petrel has consistently been the most at-risk seabird from commercial fishing activity within New Zealand's EEZ (Richard & Abraham 2013, 2015). A long-term study was initiated in the 1995-96 breeding season, based upon three 40 m x 40 m census grids located on Mount Hobson, Great barrier Island, in which all black petrel burrows were monitored. Additionally, burrows within 10 m of the summit area walking tracks were also monitored. In 1998-99 the number of census grids was increased to six and then to nine in 1999-00. By the end of the 2016-17 breeding season, a total of 448 study burrows were monitored (Bell et al. 2018).

There have been a relatively large number of reports produced describing the work undertaken on black petrels, mostly on Great Barrier Island. Summary information from the most recent of these reports (Bell et al. 2018) is reproduced here, together with further information on black petrels on Little Barrier Island from Bell et al. (2016).

Long-term trends in black petrel population size have been determined in two ways. Firstly, burrows and burrow density were determined from the census grids, which occur in high-grade habitat, and these results extrapolated across the 4.6 ha of high-grade habitat within the Mount Hobson study area. These data indicated that the population within the high-grade habitat has been relatively stable since 1995 with a mean of 291 breeding pairs (Figure 8.18, Bell et al. 2018).

Secondly, 20 random transects were surveyed within the 35 ha study area producing 176 burrows with active breeding attempts. Stratifying for habitat quality produced a population estimate of 2 427 breeding pairs (95% c.i. 1 713-3 141) within the 35 ha study area. Similar data from 2004-05, 2009-10 and 2012-13 were compared to those from 2016-17 (Figure 8.19), revealing an apparent population increase from approximately 1 000 pairs (2004-05 and 2009-10) to approximately 2 500 pairs (2012-12 and 2016-17). Bell et al. (2018) noted that this apparent increase may be due, in part, to increased likelihood of burrow detection. Further, Bell et al. (2018) recommended that an island-wide population estimate be generated, incorporating the use of a seabird dog, to include breeding birds beyond the boundaries of the study area.

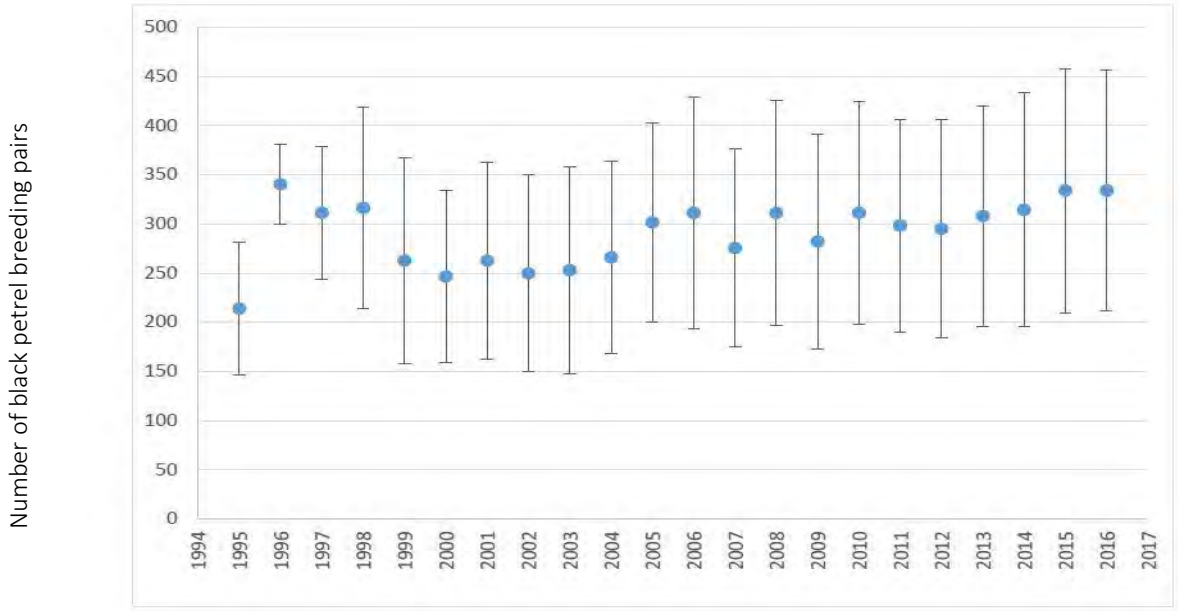


Figure 8.18: (taken from Bell et al. 2018). Annual black petrel breeding pairs population estimate for the high–grade petrel habitat derived from census grids for the Mt Hobson/Hirakimata study area 1995–2016.

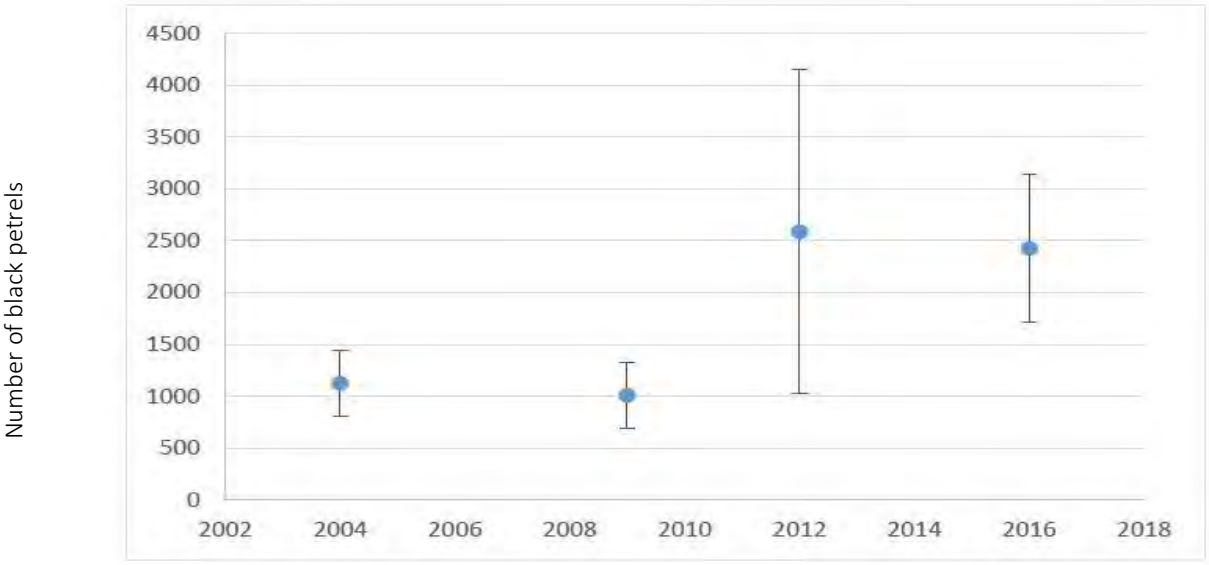


Figure 8.19: (taken from Bell et al. 2018). Breeding population estimates of black petrel in the 35 ha Mt Hobson/Hirakimata study area 2004–05 to 2016–17 breeding seasons.

Estimates of adult survival have been made for each year from 1995-96 to 2016-17 (Bell et al. 2018), revealing relatively high and consistent adult survival over the most recent five years (approximately 0.95), but with several years where adult survival was close to 0.80, and additional years where survival was less than 0.90 (Figure 8.20). Interestingly, the relatively high recent estimates of adult survival reported by Bell et al. (2018) determined using a Cormack–Jolly–Seber model in program MARK contrast with an estimate of 0.87 reported by Zhang et al. (unpublished data, project PRO201705A) using a Bayesian demographic (population) assessment model (NIWA's SeaBird model). The difference between these two approaches is relatively large and finding an explanation for this difference will be important.

The rates of return of banded chicks, by cohort, were presented in Bell et al. (2016). Overall, the rates of return of birds banded as chicks to the Great Barrier Island colony are relatively low (approximately 6-13%, depending on cohort), with chicks banded in more recent years (2013-14 to 2015-16) yet to be recaptured at the colony (Table 8.22).

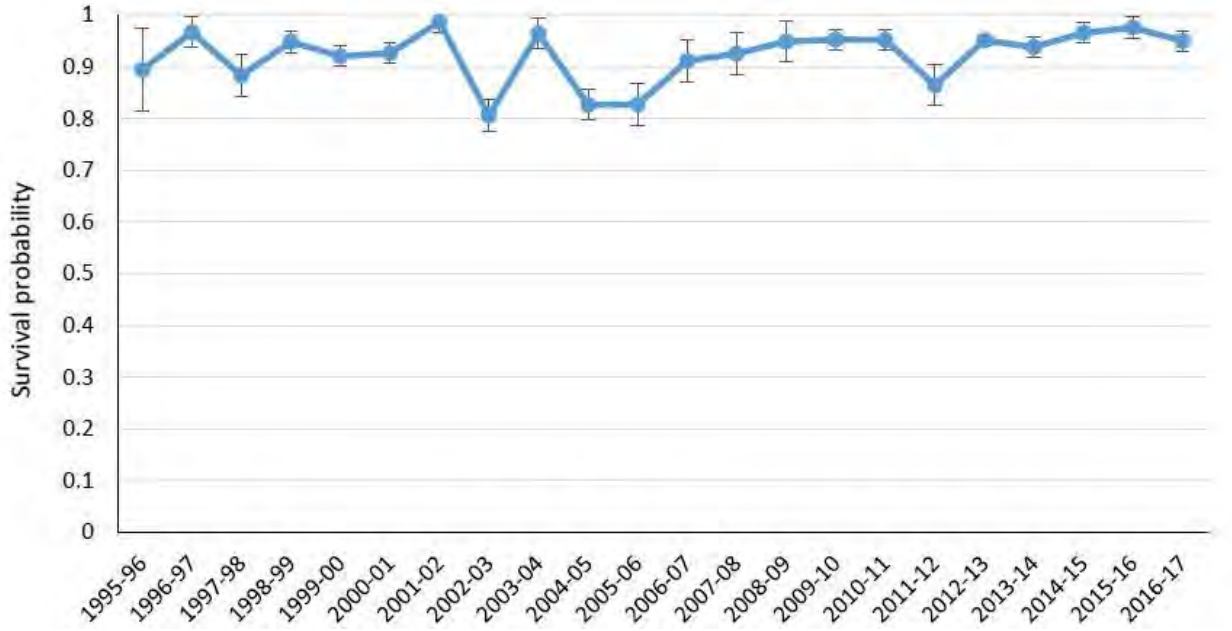


Figure 8.20: (taken from Bell et al. 2018). Trend in annual adult survival among black petrels banded at the Mt Hobson/Hirakimata colony between 1995 and 2017 (error bars represent the standard error of the mean).

Table 8.22: (from Bell et al. 2016). Number of black petrel chicks banded each season and the proportion of those chicks that have been recaptured within the study site on Great Barrier Island/Aotea since the 1995-96 breeding season.

	Total number of banded chicks	Total number of returned chicks	Proportion (%) of returned chicks
Pre-1995	209	26	12.4
1995/96	59	4	6.8
1996/97	69	7	10.1
1997/98	85	11	12.9
1998/99	116	15	12.9
1999/00	137	16	11.7
2000/01	137	9	6.6
2001/02	160	18	11.3
2002/03	62	7	11.3
2003/04	110	9	8.2
2004/05	184	22	12.0
2005/06	143	18	12.6
2006/07	215	22	10.2
2007/08	191	18	9.4
2008/09	203	18	8.9
2009/10	171	15	8.8
2010/11	144	10	6.9
2011/12	163	3	1.8
2012/13	219	1	0.5
2013/14	199	0	0
2014/15	215	0	0
2015/16	171	0	0
TOTAL	**3362**	**249**	**7.4**
MEAN (± SEM)	152.8 ± 11.0	11.3 ± 1.7	8.0 ± 1.0

Between 1998-99 and 2016-17, breeding success in black petrels in the Mount Hobson study area has ranged from approximately 60% (2010-11) to approximately 82% (2006-07, Figure 8.21), with an overall mean across 19 years of 74% (Bell et al. 2018).

In 2015-16, a total of 149 black petrel study burrows were monitored on Little Barrier Island, including 92 which were established in 1997. Breeding success in these burrows was 85% (Bell et al. 2016). Additionally, 49 breeding burrows were detected along 36 transects and a further 121 active burrows were detected along 52.5 km of seabird-detector dog surveys. Based on the area of Little barrier island above 500 m above sea level (where the majority of active burrows were located), Bell et al. (2016) estimated the Little Barrier island black petrel population to be approximately 620 pairs. Acoustic monitoring techniques were deployed in the Moehau Range, Coromandel Peninsula between November 2015 and January 2016. No black petrel calls were recorded (Bell & Stewart 2016).

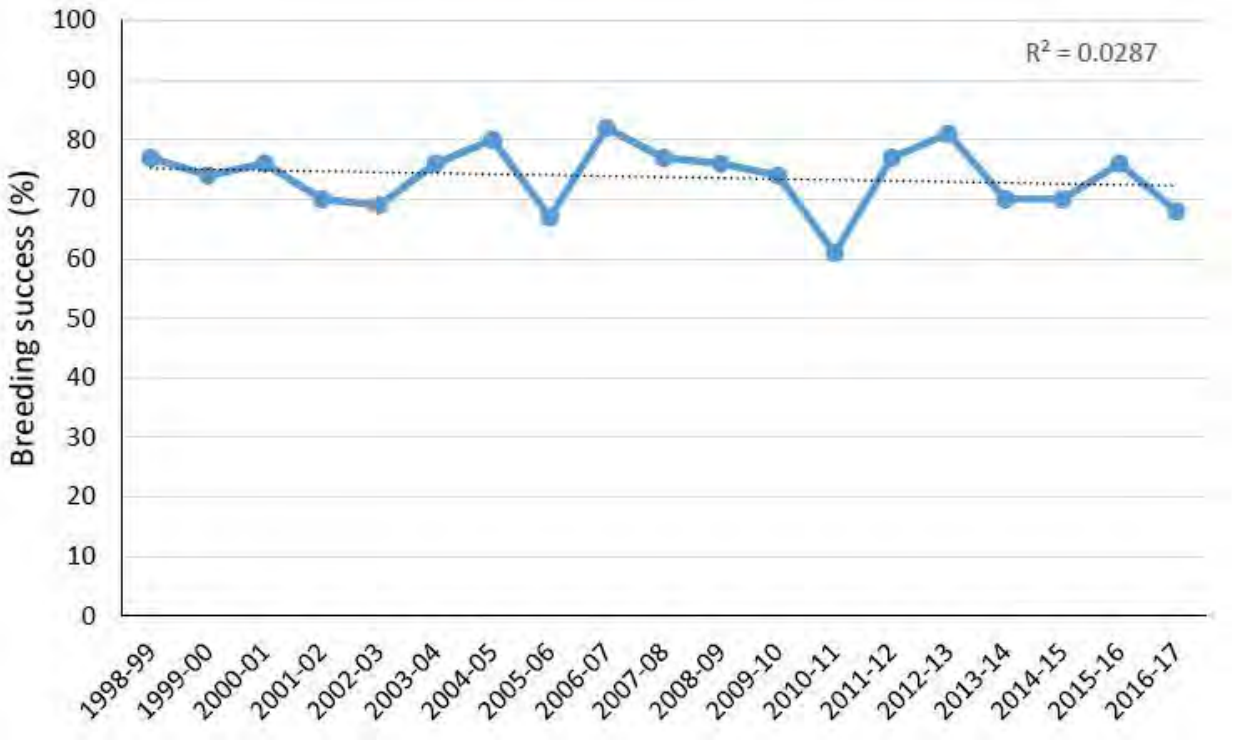


Figure 8.21: (taken from Bell et al. 2018). Trend in breeding success (percentage of breeding burrows that fledge a chick) among black petrel study burrows at Mt Hobson/Hirakimata on Great Barrier Island between 1998 and 2017.

8.4.3.1.1 QUANTITATIVE MODELS FOR BLACK PETREL

Francis & Bell (2010) analysed data from the main population of black petrel at Great Barrier Island. Abundance data from transect surveys were used to infer that the population was probably increasing at a rate between 1.2% and 3.1% per year. Mark-recapture data were useful in estimating demographic parameters, like survival and breeding success, but contained little information on population growth rates. Fishery bycatch data from observers were too sparse and imprecise to be useful in assessing the contribution of fishing-related mortality. Francis & Bell (2010) suggested that, because the population was probably increasing, there was no evidence that fisheries posed a risk to the population at that time. They cautioned that this did not imply that there was clear evidence that fisheries do not pose a risk.

Subsequent analysis (Bell et al. 2012) included an additional line transect survey in 2009–10 in which the breeding population was estimated to be about 22% lower than in 2004–05 (the latest available to Francis & Bell, 2010). Updating the model of Francis & Bell (2010) made little difference to estimates of demographic parameters such as adult survival, age at first breeding, and juvenile survival (which had 95% confidence limits of 0.67 and 0.91). The uncertainty in juvenile survival gave rise to uncertainty in the estimated population trend, with a mean rate of population growth over the modelling period ranging from -2.5% per year (if juvenile survival = 0.67) to +1.6% per year (if juvenile survival = 0.91, close to the average annual survival rate for older birds) (Figure 8.22). Bell et al. (2012) concluded that the mean rate of change of the population over the study period had not exceeded 2% per year, though the direction of change was uncertain. The latest counts have increased, due mainly to increases in breeding rate (Bell et al. 2013), suggesting even more uncertainty about population trend than when the quantitative modelling was last updated.

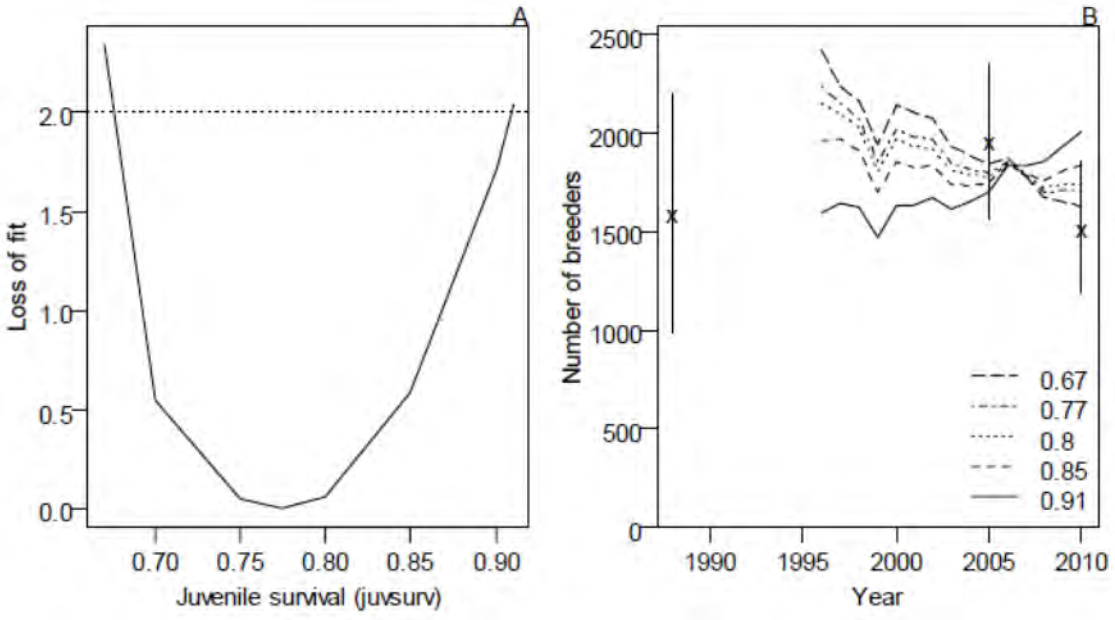


Figure 8.22: (from Bell et al. 2012). Likelihood profile for annual probability of juvenile survival of black petrel, showing: A, the loss of fit (the horizontal dotted line shows a 95% confidence interval for this parameter); and B, population trajectories corresponding to different values of juvenile survival, together with population estimates from transect counts (crosses with vertical lines indicating 95% confidence intervals. Note that the 1988 population estimate was not used in the model.

8.4.3.2 SALVIN'S ALBATROSS

8.4.3.2.1 BOUNTY ISLANDS

Salvin's albatross is endemic to New Zealand, breeding only on the Bounty Islands and the Western Chain of The Snares. The Bounty Islands are a group of bare rocky islands/islets situated 659 km south-east of New Zealand's South Island. In October 2010, Baker et al. (2010a) completed an aerial survey of the Bounty Islands to photograph all albatross colonies. This was the first complete population survey of Salvin's albatross on the Bounty Islands. Photo montages were created from the aerial photography and the number of nesting birds was counted. From these data, Baker et al. (2010a) estimated the total count of nesting Salvin's albatrosses in the Bounty Islands in October 2010 to be 41 101 (95% c.i.: 40 696–41 506).

This estimate may be biased high by the presence of 'loafers' (non-breeding birds) as it was not possible to ground truth the aerial photography or detect the proportion of loafers within the colony from close-up photography (because of the general lack of nest pedestals resulting from low availability of nesting material on the island). Conversely, the estimate may be biased low because aerial photography was not possible on some small areas of steep cliff where albatross nests may have been missed (Baker et al. 2012).

A review of existing ground counts was reported by Amey & Sagar (2013). To estimate population trends and examine the accuracy of ground counts, whole-island surveys of Salvin's albatross breeding at Proclamation Island, Bounty Islands, were undertaken during November in 1997, 2004 and 2011. These counts suggest that the numbers of Salvin's albatross nests on Proclamation Island declined by 14% between 1997 and 2004, by 13% between 2004 and 2011, and overall by 30% between 1997 and 2011. Counts of nests on Depot Island decreased by 10% between 2004 and 2011.

Baker et al. (2014a) conducted a repeat aerial survey of the Bounty Islands in October 2013. Using the same correction factor applied to the 2010 counts, they estimated the total annual breeding pairs at 39 995 (95% c.i.: 39 595–40 395) compared to the corrected estimate for 2010 of 31 786 (95% c.i: 31 430–32 143).

DOC Conservation Services Programme have been reviewing the methodology for undertaking a survey of the

Salvin's albatross on the Bounty islands (Debski & Hjörvarsdóttir 2017). In October 2018, a ground count of breeding Salvin's albatross was completed at Proclamation Island, together with ground-truthing of an aerial survey and deployment of a suite of tracking devices. Further work is planned for October 2019.

8.4.3.2.2 WESTERN CHAIN, SNARES ISLANDS

In 2008, a three-year study of Salvin's albatrosses was initiated at the Snares Western Chain. The three main objectives of the Salvin's albatross field work were:

- to estimate the breeding population size from counts of occupied nests;
- to determine foraging locations and activity by retrieving geolocator tracking devices deployed in 2008; and
- to estimate annual survival rates of banded adult birds from recapture analyses.

Totals of 1195 and 1116 breeding pairs were counted on Toru and Rima Islets during October 2008 (Charteris et al. 2009) and September–October 2009, respectively (Carroll et al. 2010) (Table 8.23). Only Toru Island was sampled in 2010.

In order to estimate the adult survival of Salvin's albatross, a total of 257 occupied nests were counted within a clearly defined study area established in October 2008 (Charteris et al. 2009). Within this area, 116 birds banded in previous years were recaptured, and a further 20 breeding birds were banded in the study area during October 2010. Among the recaptured birds were 13 that had been banded as chicks on Toru Islet during 1986, and 23 of the 123 birds banded as breeding adults in 1995. These recapture rates lead to an estimated adult survival probability of 0.967 for Salvin's albatross, one of the highest estimates for any species of annual-breeding albatross (Sagar et al. 2011), which is noteworthy given the location of the colony in an area of relatively high fishing activity..

Twenty-four of the 35 geolocation loggers deployed on breeding birds during October 2008 were retrieved. Data were processed by the British Antarctic Survey and a preliminary assessment of the distribution of Salvin's albatrosses during the entire year is presented in Figure 8.21. None of the 24 birds tracked was within the New Zealand EEZ during April; 23 were in South American waters between Tierra del Fuego and northern Peru and one was in eastern Bass Strait and along the eastern coast of Tasmania (Figure 8.21a). Birds began to return to New Zealand waters during May and this continued throughout June and July. The tracks of birds exiting South American waters originated from either the Peruvian or southern Chilean coasts. During this period, birds recently arrived in New Zealand waters occurred primarily east of the Chatham Islands, off Puysegur and on the Stewart-Snares Shelf (Figure 8.21b). Eggs are laid starting in August and all of the birds occurred within Australasian waters throughout August to October, primarily on the Challenger Plateau, off Puysegur, the Stewart-Snares Shelf, and Campbell Plateau (Figure 8.21c). During this period these birds from the Snares Western Chain occupy a relatively narrow longitudinal range between 160°E and 175°E and appear to avoid, or be excluded from, the area around the Bounty Islands, where there is another colony of Salvin's albatross. Beginning in mid-October, chicks hatch and, between November and March, presumed successful breeders foraged primarily on the Challenger Plateau, off Puysegur, the Stewart-Snares Shelf, and Campbell Plateau (Figure 8.21d). There was some movement across the Pacific in each of the months between November and March with presumed failed breeders leaving the New Zealand EEZ during the earlier part of this period and presumed successful breeders migrating east during March (Sagar et al. 2011).

Further research has been recently conducted on the Salvin's albatross on the Snares Western Chain (Baker et al. 2015). This research included a ground-based census, an aerial survey (including ground truthing) and collection of information on tagged birds. The aerial survey estimated 1486 (95% c.i.: 1409–1563) annual breeding pairs in 2014–15, which was 32% higher than the ground counts undertaken on the same day of the aerial survey (Baker et al. 2015b)

Table 8.23: (from Sagar et al. 2011). Numbers of Salvin's albatross pairs breeding on Toru and Rima Isles, Western Chain, The Snares, 2008–10. Failed nests are those assessed to contain fresh egg fragments. No count was made on Rima Islet in 2010.

Islet	Date	Adult + egg	Obvious failed nest	Total
Toru	6–7 October 2008	828	70	898
	2 October 2009	783	51	834
	28–29 September 2010	780	49	829
Rima	16 October 2008	279	18	297
	30 September 2009	265	17	282

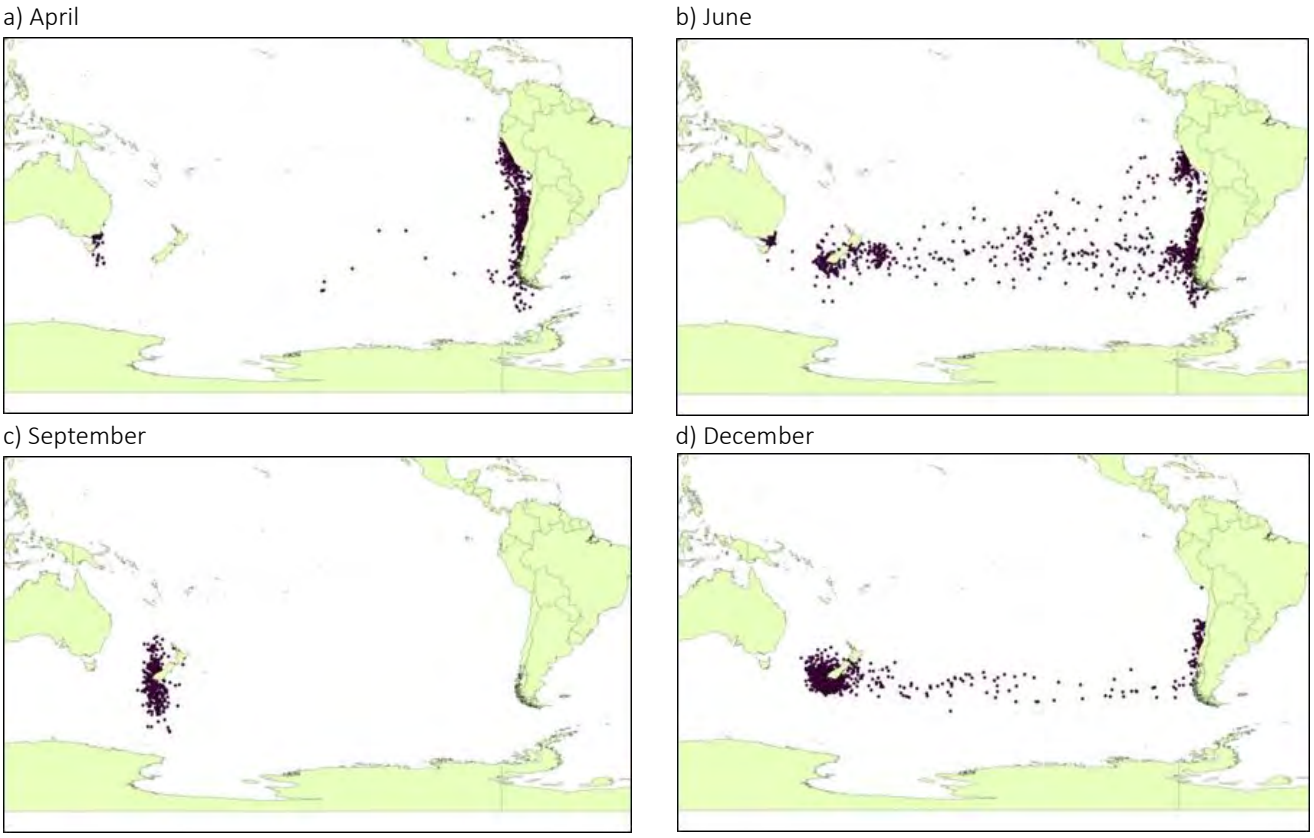


Figure 8.21: (from Sagar et al. 2011). Distribution of Salvin's albatrosses *Thalassarche salvini* from the Snares Western Chain tagged with geolocators at four times of the year: a) April, after the completion of their breeding season, b) June, showing their return tracks from South American waters to New Zealand waters prior to egg laying, c) September, when their partners were incubating an egg, and d) December, the birds around New Zealand are presumed to be foraging for food for themselves and their chick, whilst the birds crossing the Pacific and in South American waters are presumed to be failed breeders.

8.4.3.3 WESTLAND PETREL

The Westland petrel is endemic to New Zealand and nests in burrows in dense rainforest near Punakaiki, Westland. This species is poorly studied, probably largely due to burrow-nesting, inhabiting dense forest, and attendance at their nests only at night. As for the flesh-footed shearwater, a survey methodology for estimating population size and assessing long-term trends for the Westland petrel was designed (Baker & Double 2007). Once a colony was located, Baker et al. (2007b, 2008a, 2011a) estimated population size through a three-stage process. First, burrow densities were determined in each colony by using 2 m-wide strip 'colony transects', and mapped burrows along each transect. These transects differed from search transects in that they were confined to identified colonies and were randomly placed within the colonies. Second, the proportion of active nests per burrow was estimated using burrow scopes and 'inspection by hand' (inserting an arm down burrows to determine occupancy and feel for eggs, chicks, adult birds or nesting material). Finally, the area of each colony was measured by exploring the approximate boundaries on foot and mapping the densely inhabited area

and this area multiplied by the density to arrive at a population estimate for each colony.

Although Westland petrels breed throughout a 16 square kilometre area near Punakaiki, which has been designated as a Special Conservation Area, sampling effort was concentrated on estimating the population in high density areas, noting the challenges posed by the rugged terrain and often adverse weather conditions (Baker et al. 2007b, 2008a, 2011a). Baker et al. (2007b, 2008a, 2011a) estimated the number of potential burrows in all Westland petrel colonies to total 6846 (95% c.i.: 6389–7302) during the period 2007 to 2011. Of these, an estimated 2827 (2143–3510) were occupied. The rugged terrain and inclement weather made it difficult to ensure that the permanent transects were replicated exactly each year and hence raises some doubts about the comparability of counts. Wood & Otley (2012) estimated colony and population sizes of Westland petrel between 2002 and 2005, and found between 2 954 and 5 137 breeding pairs, and an overall total of nearly 13 000 potential burrows.

Waugh et al. (2018) described the foraging ranges of Westland petrels during the pre-laying phase in 2011, incubation phase in 2011, 2012, 2015 and 2016 and during the chick-rearing phase in all years except 2011. Birds tended to use the same areas across years and breeding stages, with core areas off Hokitika, in Cook Strait and to the south-west off Haast (Figure 8.22).

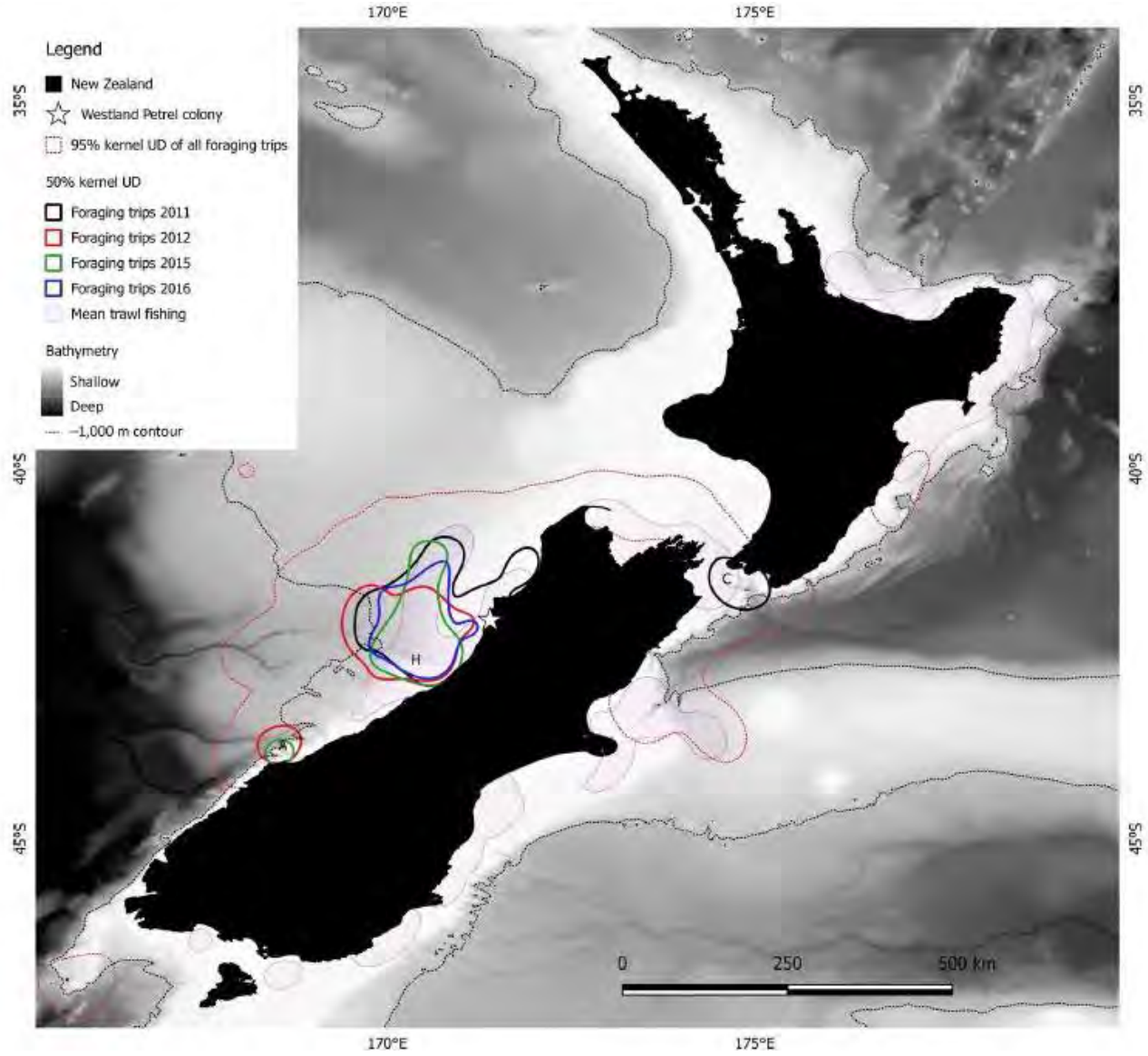


Figure 8.22: (from Waugh et al. 2018). Main foraging areas of Westland petrels, showing core foraging zones, bathymetry and the 1 000 m depth contour. Areas of intensive usage are associated with steep bathymetric slopes south of the colony at Arawata and Haast canyons (A) (within the 2012 and 2015 50% kernel utilisation distributions [UD]; at Hokitika Canyon (H) in the central zone in all years; and in Cook Strait over Nicholson Canyon (C) between the North Island and South Island (2011 only).

Landers et al. (2011) used light-based geolocation tags to describe the long-distance migration patterns of Westland petrel. Based on data from eight birds, Landers et al. (2011) found that post-breeding migration occurred in November and birds travelled east to waters off South America, covering approximately 7 000 km in six days (range 4-7 days). The return trip to New Zealand ahead of the breeding season occurred in April, taking ten days (range 8-13 days). Six of the eight birds remained off Chile during the non-breeding period, but the remaining two birds travelled to the east coast of South America to waters off southern Argentina Landers et al. 2011).

8.4.3.3.1 QUANTITATIVE MODELS FOR WESTLAND PETREL

Waugh et al. (2015) modelled the population dynamics and demographic characteristics of Westland petrel using a data set spanning 1970 to 2014. They found that the population showed a slowly increasing trend from the early 1970s, underpinned by relatively high adult survival, high breeding success (60% of eggs laid produced chicks that fledged), a mean age of first return to the colony of 7.7 years, a recruitment of fledged chicks to first return of 36% and negligible emigration. Annual adult survival for breeding birds was the same for both sexes (95.4%, 95% c.i. 91.8-97.5%) and was constant across years (Waugh et al. 2015).

8.4.3.4 FLESH-FOOTED SHEARWATER

Flesh-footed shearwaters breed around Australia and New Zealand and migrate to the northern hemisphere in the non-breeding season. In New Zealand, they nest in burrows on islands around the North Island and in Cook Strait. Of the breeding sites identified by DOC staff (G. Taylor unpublished, cited in Baker et al. in prep) eight major breeding islands for the flesh-footed shearwater were chosen for re-survey: Lady Alice, West Chicken, Whatupuke and Coppermine (Hen and Chickens Group); Green (Mercury Group), Ohinau (Ohena sub-group of Mercury Group), Karewa (Bay of Plenty) and Titi (Cook Strait). In addition, it is estimated that Middle Island (Mercury Group) held approximately 3000 pairs in 2003 (Waugh & Taylor 2012).

Baker & Double (2007) designed a survey methodology for estimating population size and assessing long-term trends for the flesh-footed shearwater. Surveys using this design were undertaken at the eight major breeding areas by Baker et al. (2008b, 2009a, 2010a, in prep.). Field work was focused on visiting all of the eight sites at least once during the five years of the study to estimate the number of pairs breeding at each site. A few sites were visited annually to estimate population trends. Baker et al. (2008b, 2009a, 2010a, 2011) searched these sites by locating ridgelines and systematically searching from the ridgeline to the sea or, where unsuitable terrain such as a cliff was encountered, using a series of 2 m-wide search transects. These search transects were established by following a compass bearing downhill from the ridgeline. When potential burrows were located, their location of that colony from the start point of the search transect was recorded, and the number of potential burrows subsequently found 1 m either side of the transect line counted. At some sites, colony transects were well marked to permit follow-up surveys in future years. The origin points for transects were randomly located along a central line or 'backbone', which was run through the colony. In practice, most colonies were centred on ridgelines or located on steep slopes, and the backbone was located along a ridgeline.

All colony areas, with the exception of those on Karewa, were mapped by using transect data and a hand-held GPS. On Karewa Island, the sensitive nature of the substrate meant that sampling was curtailed to working from boards laid on the surface along a sandy track used by DOC for park management purposes. This access point was used as a long transect, with other shorter transects established either side as permitted by the terrain encountered.

The density of potential burrows was scaled up to the estimated area of each colony to derive an estimate of the number of burrows for each colony (Table 8.24). Baker et al. (2011) estimated the total count of burrows on the eight islands surveyed to be 20 945 (95% c.i.: 19 019–22 871), notably fewer than Taylor's (2000) estimate of 25 000–50 000 pairs. Baker et al. (2011) stated that their estimates generally accord with the indicative population estimates

developed by Graeme Taylor (cited in Baker et al.2011) with the exception of that for Coppermine and Ohinau islands. Baker et al.'s (2011) estimate of 1425 occupied burrows (1059–1791) for Coppermine is much lower than Taylor's indicative estimate of 10 000 (presumably breeding pairs). In contrast, Baker et al.'s (2011) estimate of 2071 occupied burrows (943–3200) for Ohinau greatly exceeds Taylor's indicative estimate.

In January 2017, Bell & Boyle (2017) assessed the flesh-footed shearwater population at Middle Island in the Mercury Islands group and estimated 5 822 (95% c.i. 2 400-9 244) breeding pairs (corrected for burrow occupancy), the largest breeding population of flesh-footed shearwaters in New Zealand. Bell & Boyle (2017) reappraised a 2003 population estimate for flesh-footed shearwater at Middle Island undertaken by Graeme Taylor. Bell & Boyle (2017) reanalysed Taylor's data and estimated 4 441 breeding pairs in 2003.

Waugh et al. (2014) assessed the feasibility of gaining improved estimates of key flesh-footed shearwater population parameters and investigated the at-sea distribution of flesh-footed shearwaters. Study plots were established at Lady Alice/Mauimua, Titi Island and Ohinau Island, with burrow mapping by GPS and hand-drawn maps. The occupancy of burrows and size of breeding population at each colony was assessed. Occupancy was assessed by burrow-scoping and through inspection of burrow contents through study hatches.

Table 8.24: (from Baker et al. 2011). Estimated number of potential and occupied burrows for flesh-footed shearwater for eight New Zealand islands surveyed 2007–08 to 2010–11. Note that some colonies on Lady Alice and Coppermine were visited in all years, and for these colonies the highest estimate was used to derive the island total. The number of occupied burrows can reasonably be considered an estimate of annual breeding pairs for each island.

Island	No. potential burrows	Lower 95% c.i.	Upper 95% c.i.	No. occupied burrows	Lower 95% c.i.	Upper 95% c.i.
West Chicken	193	-2	388	15	0	210
Lady Alice	2 763	2 079	3 447	921	237	1 605
Whatupuke	2 941	1 767	4 115	1 210	36	2 384
Coppermine	2 290	1 924	2 656	1 425	1 059	1 791
Titi	2 814	2 201	3 427	337	0	950
Green	132	82	182	74	24	124
Ohinau	3 883	2 755	5 011	2 071	943	3 200
Karewa	5 929	4 420	7 438	2 561	1 052	4 070
Total	20 945	19 019	22 871	8 614	6 689	10 540

Analysis of island-wide population survey information, collected from 2011–12 to 2013–14 compared with previous surveys conducted from 2007–10 (Baker et al. 2008b, 2009a, 2010a, 2011) indicated a probable decline for the population on Ohinau Island, and stable populations on Lady Alice Island/Mauimua and Titi Island. Adult annual survival was within the range reported for other shearwaters, at 0.93 for Kauwahaia Island and 0.94 for burrow-caught birds at Lady Alice/Mauimua (Waugh et al. 2014, Jamieson & Waugh 2015). Crowe et al. (2017) determined an overall breeding success of 49.1% for flesh-footed shearwaters at Lady Alice/Mauimua and Ohinua Island for the 2016-17 breeding season. Competition for burrows from other seabirds may be an issues for flesh-footed shearwaters at some locations (G Taylor pers. comm.).

Tracking of flesh-footed shearwaters from Ohinau and Titi islands using GPS loggers showed that birds were foraging several hundreds of kilometres from their breeding site over deep oceanic waters to the east of the New Zealand region during incubation. During the early chick-rearing period, the flesh-footed shearwaters contracted their range with a higher concentration of activity in waters near the breeding site and at zones of upwelling and relative high productivity within 400 km of the breeding site (Figure 8.23). The overlap of foraging activity with trawl, longline and gillnet fisheries indicated highest intensity of overlap when the breeding birds were foraging close to the breeding site during early chick rearing (Waugh et al. 2014, 2016).

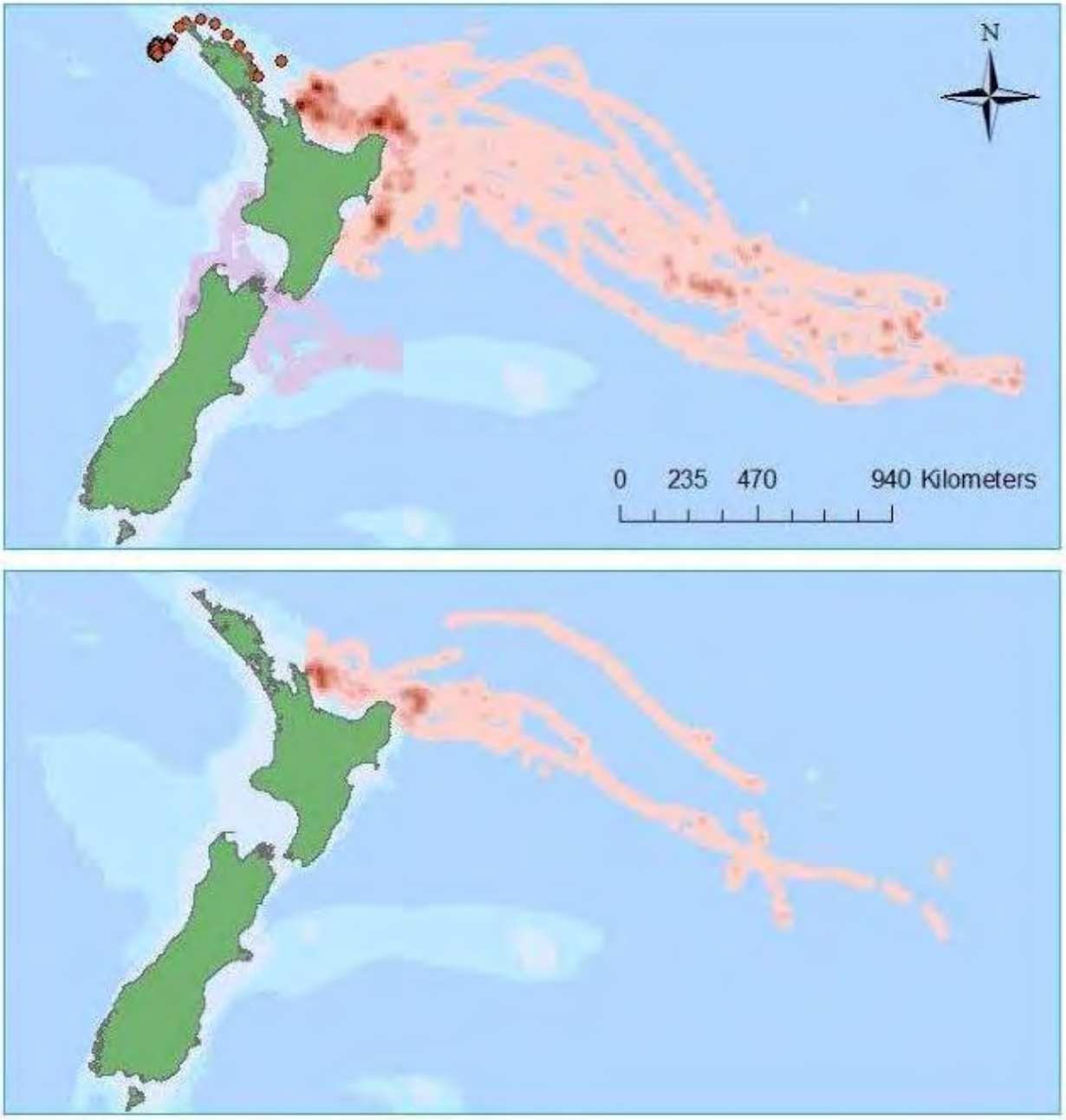


Figure 8.23: (from Waugh et al. 2014). Breeding season locations of foraging trips for Flesh-footed Shearwaters tracked with GPS loggers from Ohinau and Titi Islands. The upper and lower figures represent kernelled density plots during the incubation period (January) and chick-rearing period (February), respectively. The dotted lines represent the 1000m bathymetry contour.

More recent GPS tracking work of breeding flesh-footed shearwaters at Lady Alice during January and February 2017, and January 2018, revealed a more northerly at-sea distribution, with the majority of foraging effort centred along the east and west coasts of the north of the North Island (Figure 8.24 and Figure 8.25, Kirk et al. 2017 and Crowe 2018, respectively).

The Department of Conservation has an ongoing project, POP2015-02, to update the population size, estimate demographic parameters and gather at-sea distribution information for the flesh-footed shearwater (Mischler 2016).

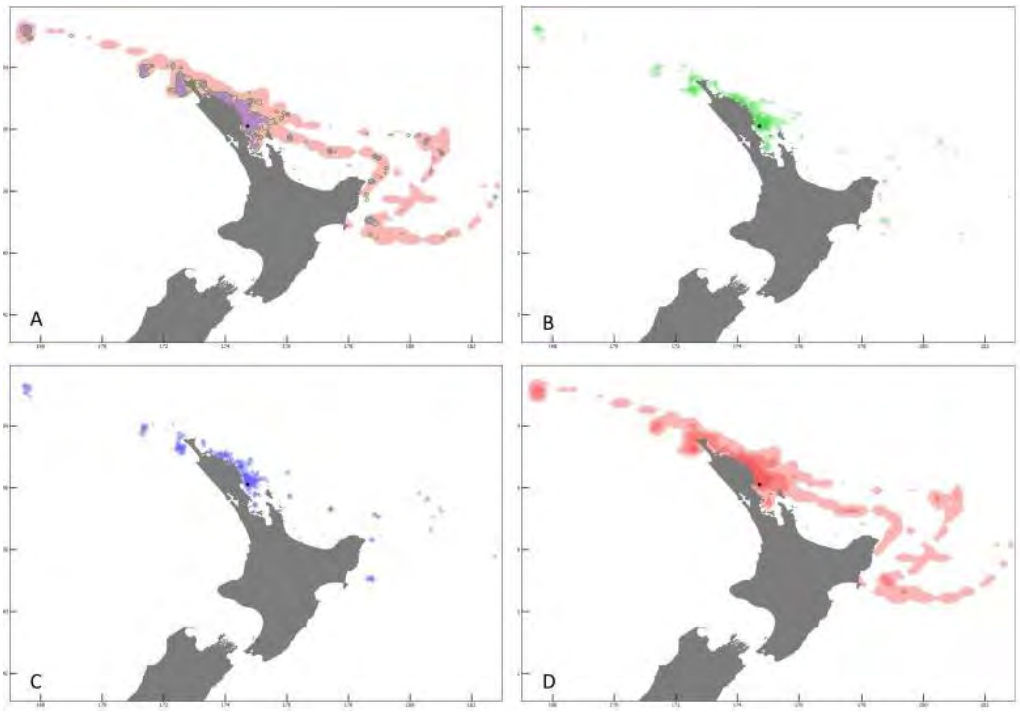


Figure 8.24: (from Kirk et al. 2017). Behavioural distributions of flesh-footed shearwaters tracked from Lady Alice Island in February 2017. A – 95% occupancy kernels for flight (red), rest (blue) and foraging (green line) behaviour. B, C and D – Occupancy kernels for each behaviour at the 50%, 75% and 95% levels (foraging behaviour is green, resting blue and flight red). Location of Lady Alice Island is shown with a black dot.

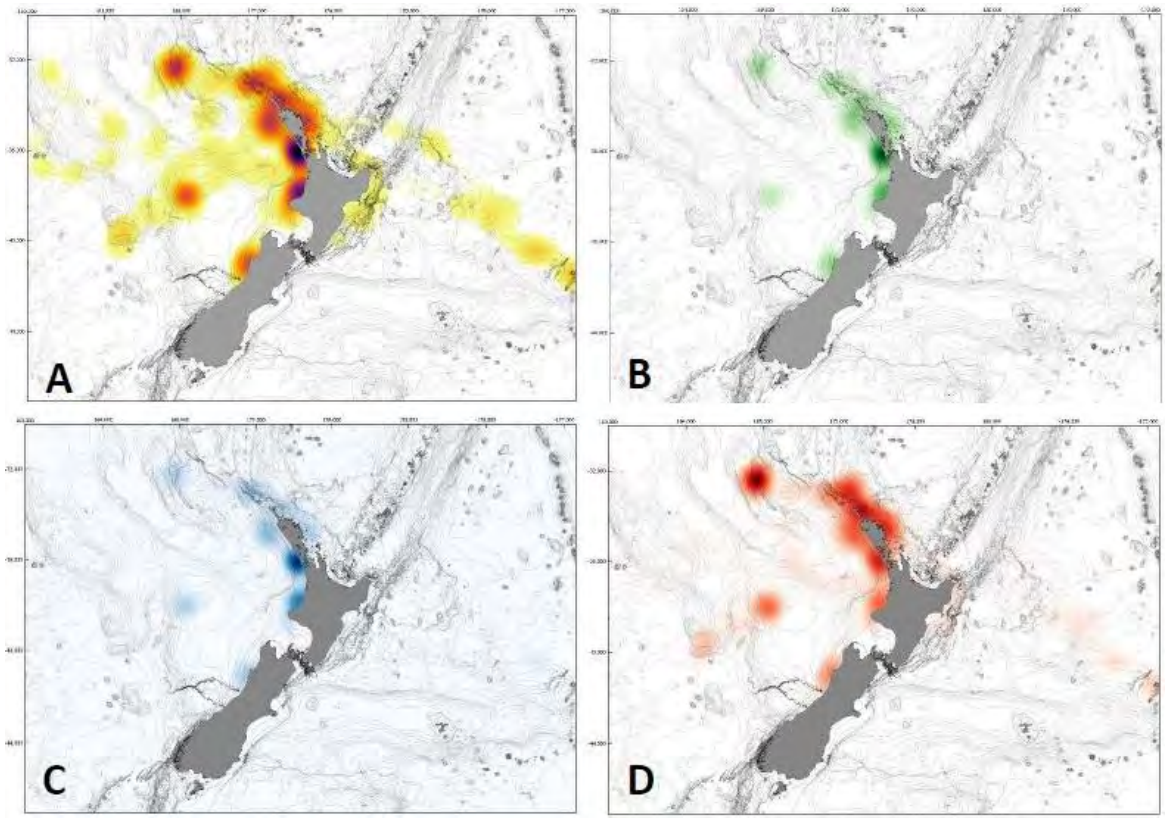


Figure 8.25: (from Crowe 2018). Heatmaps showing the behavioural distributions of flesh-footed shearwaters tracked from Lady Alice Island during the incubation period. Each behaviour type is represented by a different subset: All behaviour types combined (4A); Foraging (4B); Rest (4C) and Flight (4D). Darker colours indicate higher occupanc

8.4.3.5 SOUTHERN BULLER'S ALBATROSS

Southern Buller's albatross is endemic to New Zealand and breeds at The Snares (North East and Broughton islands) and the Solander (main Solander and Little Solander) islands only. Most research effort has been focused at the Snares, with study area population estimates conducted annually between 1992 and 2017 and whole-island counts conducted on five occasions (1969, 1992, 1997, 2002 and 2014). In contrast, whole-island population estimates for the Solander Islands have occurred on three occasions (1996, 2002 and 2016).

Sagar (2014) provided a summary of whole-island counts for The Snares, which showed that the total breeding population, based on ground-based counts, increased from 4 448 in 1969 to 7683 in 1992, 8242 in 1997, 8713 in 2002 and 8704 in 2014 (Table 8.25).

Thompson et al. (2017) provided a summary of counts of breeding southern Buller's albatross at the Solander Islands (Table 8.26). The total breeding population in February 2016 was estimated to be 5 620 pairs, higher than the totals for 2002 (4 912 pairs) and 1996 (4 147 pairs).

Table 8.25: (from Sagar 2014). Numbers of occupied nests of southern Buller's albatross counted in different areas of The Snares, 1969-2014. Values in parentheses assume rates of change on Broughton Island in 1969 and 2002, when no counts (NC) were made, are equal to those in the subtotal for North East Island in those years. Data for 1969-2002 are from Sagar & Stahl (2005).

Area/Year	1969	1992	1997	2002	2014
North Promontory	509	1108	1400	1643	1508
West Coast	121	262	317	205	146
North side, South-West Promontory	305	785	520	739	427
South side, South-West Promontory	763	1236	1410	1025	1201
Alert Stack	112	193	223	267	305
South Coast	1425	2095	2161	2554	2425
East Coast	789	1465	1693	1732	1733
Total North East Island + Alert Stack	**4024**	**7144**	**7724**	**8165**	**8047**
Broughton Island	NC	539	518	NC	657
Totals	**(4448)**	**7683**	**8242**	**(8713)**	**8704**

Table 8.26: (from Thompson et al. 2017). Numbers of breeding southern Buller's albatrosses at the Solander Islands in 2016, 2014, 2002 and 1996. Data for Little Solander Island in 2014 are from an aerial survey (baker & Jensz 2014), those from 2002 and 1996 are from a combination of ground, vantage and aerial counts (see Sagar & Stahl 2005).

Area	2016	2014	2002	1996
Solander Island	5280		4579	3885
East Bay	666		876	709
North East to North West headlands	778		1162	1086
West Bay	819		489	387
West Bay to South West Bay	481		362	306
South West Bay to South East Peninsula	2536		1690	1397
Little Solander Island	340	305	333	262
Totals for Solander group	5620		4912	4147

Combining the 2014 Snares population estimate of 8 704 pairs (Sagar 2014) with the 2016 Solander population estimate of 5 620 pairs (Thompson et al. 2017) gives an overall population estimate for southern Buller's albatross of approximately 14 300 annual breeding pairs.

Stahl & Sagar (2000a), using satellite telemetry, described the movements and distributions of southern Buller's albatross from The Snares in 1995-97. Birds made long trips to the Tasman Sea or the east coast of the South Island during incubation, but short trips east of The Snares during the guard stage. Trip length increased again during chick-rearing. Torres et al. (2013) tracked southern Buller's albatross from The Snares during the guard stage in 2008 to 2011 using GPS tags and found that generally birds travelled both to the east and west of The Snaes, but that in 2010 nearly all birds travelled to the east. Similarly, satellite telemetry of southern Buller's albatrosses at the Solander Islands revealed long trips to the Tasman Sea and both coasts of the South Island during incubation, with shorter trips during the guard stage (Stahl & Sagar 2000b). Males tended to travel east from the Solander Islands, whereas females tended to travel west from the Solander Islands (Stahl & Sagar 2000b). Waugh et al. (2017) used GPS tags to track breeding southern Buller's albatrosses from the Solander Islands in May 2016 and found that some birds travelled north to Cook Strait and the southern half of the North Island.

8.4.3.5.1 QUANTITATIVE MODELS FOR SOUTHERN BULLER'S ALBATROSS

Francis et al. (2008, see also Francis & Sagar 2012) assessed the status of the Snares Islands population of southern Buller's albatross. They estimated (see also Sagar & Stahl 2005) that the adult population had increased about five-fold since about 1950 (Figure 8.26) at a rate of about 2% per year and concluded from this that the risk to the viability of this population posed by fisheries had been small. This conclusion depends critically on the reliability of the first census of nesting birds conducted in 1969, but Francis et al. (2012) gave compelling reasons to trust that information. In summary, the later censuses did not find any concentrations of nests that were not present on the maps prepared during the 1969 census and the increase in counts after 1969 occurred in all census subareas and in five colonies where counts were made in many non-census years.

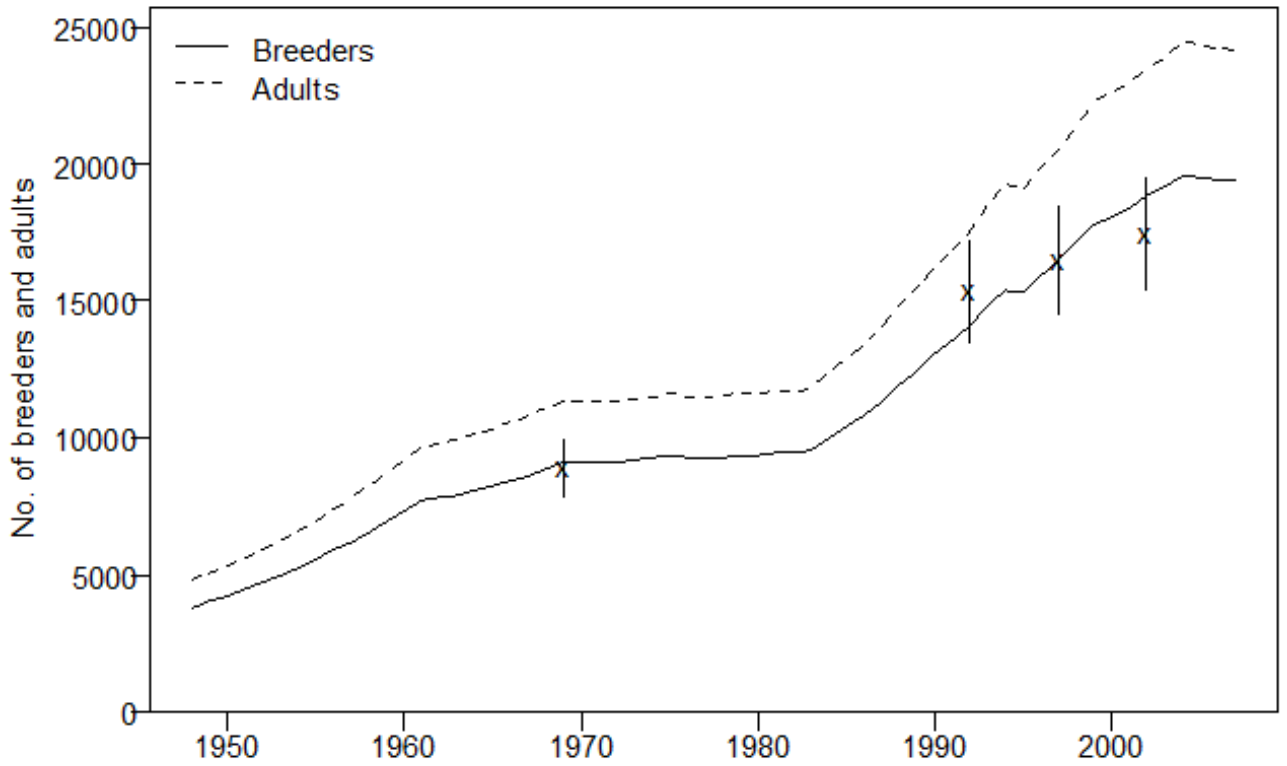


Figure 8.26: (from Francis et al. 2008). Estimates from model SBA21 of numbers of breeders (solid line) and adults (broken line) of southern Buller's albatross in each year. Also shown are the census observations (after Sagar & Stahl 2005) of numbers of breeders (crosses), with assumed 95% confidence intervals (vertical lines).

The modelling was repeated in 2015 including eight years of additional mark-recapture data and a new census estimate in 2014. Model SBA3 estimated no increase in the size of the breeding population between 2002 and 2014, and therefore concurred with Francis et al. (2008) that population growth may have stopped. The adult survival rate is likely to have declined since 1990, but for the most recent years (2008 onwards) has increased slightly. Since adult survival was the only year-varying demographic parameter, a decline in adult survival was the only possible demographic explanation for the changing population trajectory inferred by the latest census. The estimates of breeding rate (0.83 and 0.57 for breeders and non-breeders, respectively) and probability of breeding success (0.73) are similar to those of Francis et al. (2008). Forward projection assuming current demographic rates suggested that the population is likely to increase by 5.7% in the next 10 years.

There was considerable variability in some of the key demographic rates (e.g., breeding rate), and it is not known if this was due to noise in the data or natural variation in demographic processes. The ability to quantify these variabilities was hindered by a large reduction in resighting effort since 2006. Changes in resighting methods after 2006 precluded meaningful year-varying estimates of breeding parameters in recent years and so their potential effect on changing population was not properly investigated. Also, since 2008, there have been changes in the monitoring of breeding status and the cessation of monitoring of breeding success, precluding an assessment of temporal variation in reproductive rates. Also, the numbers of non-breeders may have been overstated from the mark-recapture field study during this period. These are likely to have biased the estimates of some of the demographic rates.

Fishery discards are an important component of the diet of chicks (James & Stahl 2000), but Francis et al. (2008) were not able to assess whether the associated positive effect on population growth (e.g., from increased breeding success) is greater or less than the negative effect of fishing-related mortality.

8.4.3.6 GIBSON'S ALBATROSS

Gibson's albatross is confined as a breeding taxon to the Auckland Islands: approximately 95% of the population breeds on Adams Island, with the remainder on Disappointment Island and main Auckland Island. The population on Adams Island has been the subject of research since 1991 (Walker & Elliott 2002), and the most recent report (Walker et al. 2017) provided summary data spanning most of the period 1991 to 2017.

Gibson's albatross nest numbers on Adams Island (Table 8.27) reveal a marked reduction in 2005 with only a slow increase in nest numbers subsequently.

Table 8.27: (from Walker et al. 2017). The number of Gibson's wandering albatross nests in late January in three census blocks on Adams Island in 1998–2017. Corrected total is the estimated number of nests in the three blocks taking account of the number of failed and un-laid nests at the time of counting. Estimated total population is the estimated number of nests on the island, based on the number of nests in the three counted blocks in 1997 when the last whole island count was undertaken.

Year	Rhys's Ridge (low density)	Amherst-Astrolabe (medium density)	Fly Square (high density)	Total no. of nests	Corrected total	Estimated total pop
1998	60	483	248	781	798	7875
1999	60	446	237	743	746	7367
2000	45	284	159	488	497	4904
2001	64	410	201	675	706	6969
2002	60	408	246	675	740	7303
2003	71	496	217	784	791	7809
2004	77	501	284	862	884	8728
2005	34	323	72	412	452	4467
2006	15	185	79	279	341	3363
2007	38	230	132	400	430	4245
2008	26	201	91	318	341	3371
2009	28	238	120	386	426	4211
2010	32	237	114	383	392	3872
2011	33	255	137	425	438	4323
2012	35	224	120	379	418	4131
2013	39	315	138	492	519	5120
2014	29	267	134	430	473	4669
2015	39	237	105	381	406	4010
2016	34	332	153	519	545	5385
2017	32	252	140	424	448	4423

Nesting success also declined sharply from approximately 50-75% from 1991 to 2004 to approximately 25% in 2006, and increased to 68% in 2016, the highest level recorded for 20 years (Figure 8.27, Walker et al. 2017).

Estimates of sex-specific adult survival also showed marked reductions, especially so for females, over the period 2006-2008 (Figure 8.28, Walker et al. 2017).

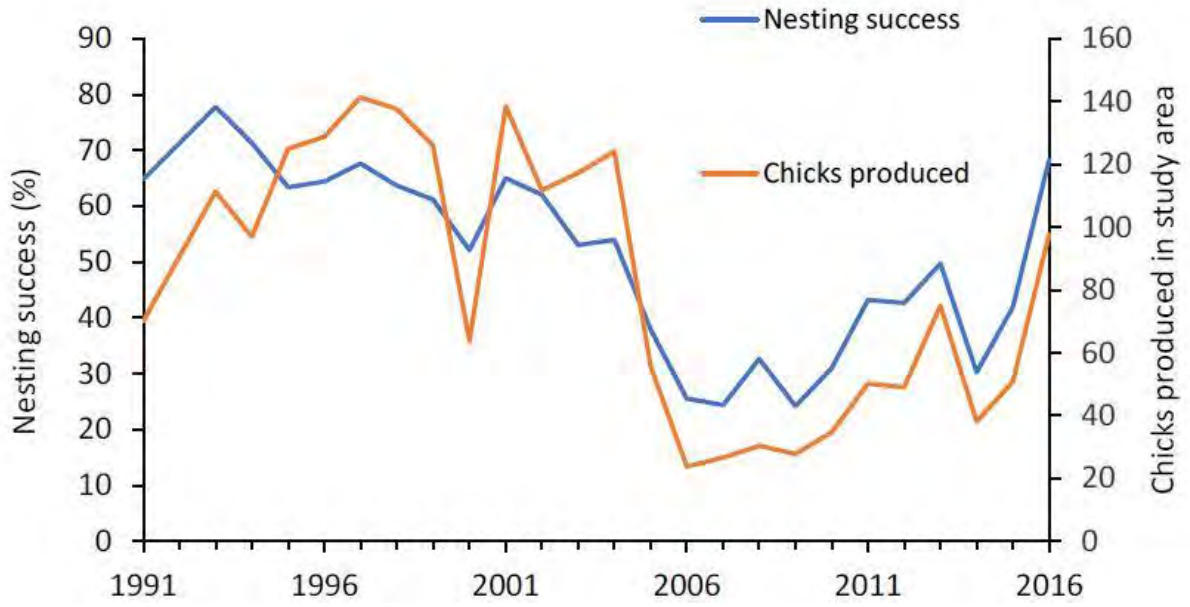


Figure 8.27: (from Walker et al. 2017). Nesting success and the number of chicks fledged from the study area on Adams Island.

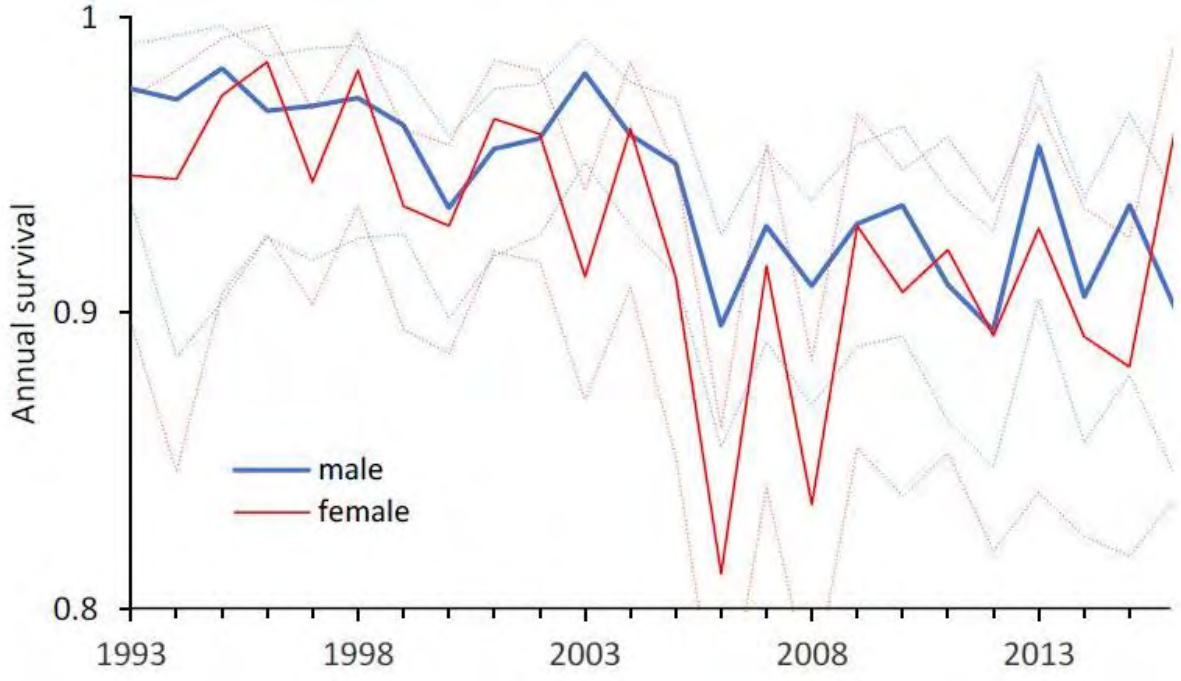


Figure 8.28: (from Walker et al. 2017). Annual survival of birds in the study area on Adams Island estimated by mark-recapture. The dotted lines are 95% confidence intervals.

Walker et al. (2017) also provided a summary of distribution data for Gibson's albatross, covering the periods 1996 to 2004 (satellite telemetry data acquired from 57 birds, also summarised in Walker & Elliott (2006)) and from 2009 (light-based geolocation data from 77 birds). Walker et al. (2017) concluded that the range of breeding birds had changed relatively little between the two periods but noted that distributions have extended further to the north post 2004 (Figure 8.29).

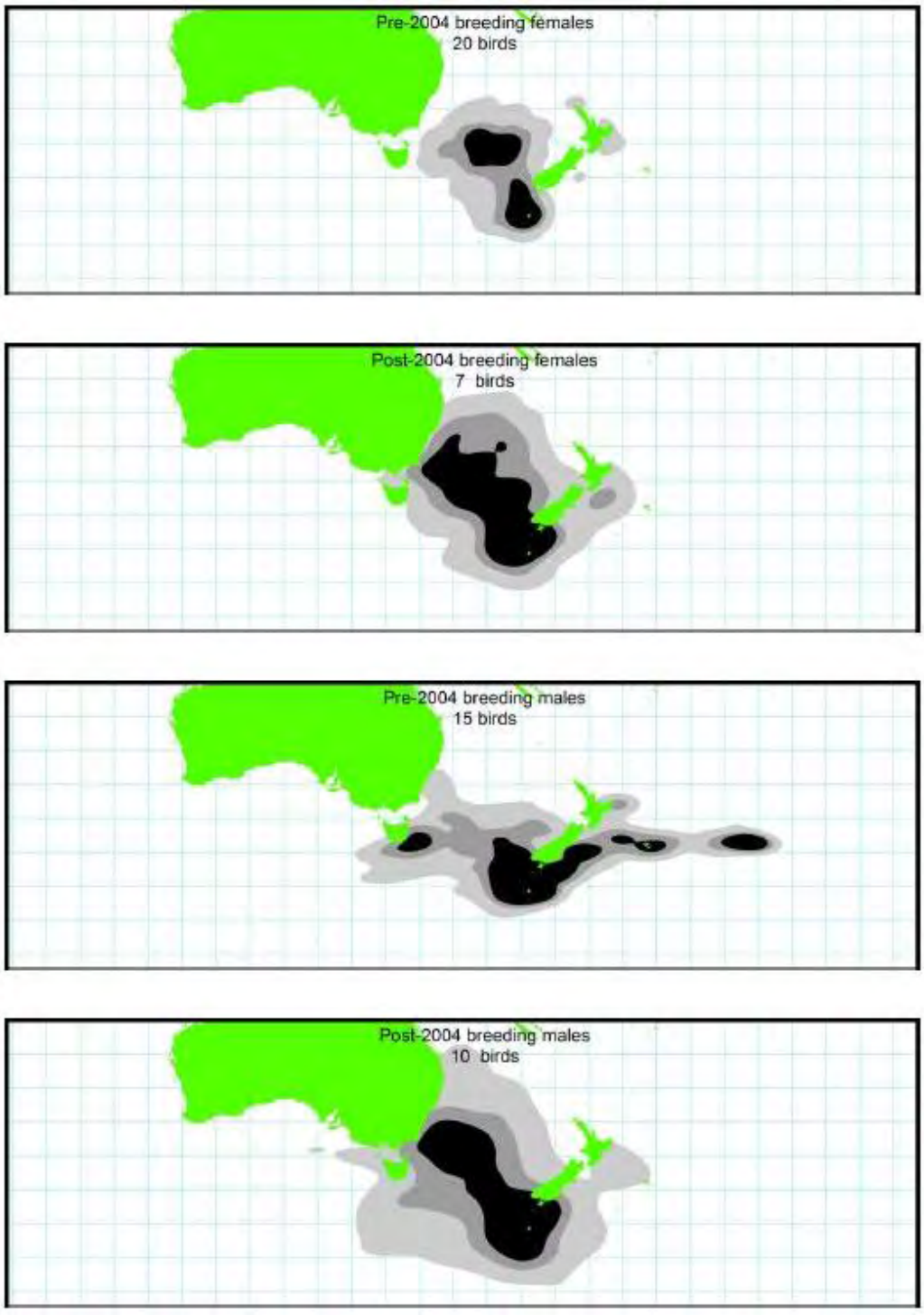


Figure 8.29: (taken from Walker et al. 2017). Kernel density plots of breeding Gibson's wandering albatrosses tracked in 1996-2004 and in 2009-17. Black indicates the 50% contour, dark grey the 75% contour and light grey the 95% contour.

For non-breeding birds, Walker et al. (2017) reported that the ranges of both sexes had increased between the two periods, with tracked birds from 2009 now utilising the South Australian Bight, a pattern that was particularly evident for females (Figure 8.30).

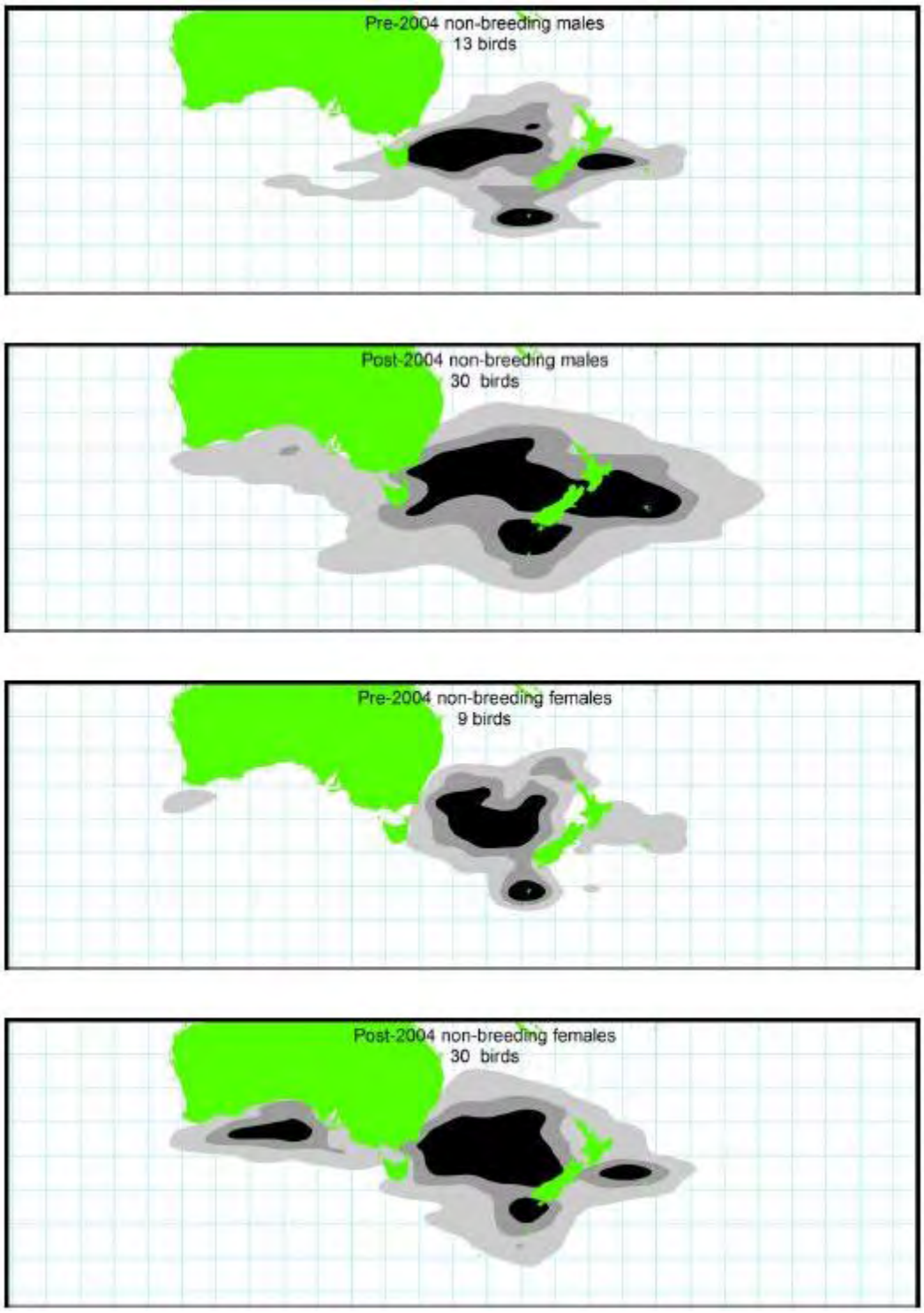


Figure 8.30: (taken from Walker et al. 2017). Kernel density plots of non-breeding Gibson's wandering albatrosses tracked in 1996-2004 and in 2009-17. Black indicates the 50% contour, dark grey the 75% contour and light grey the 95% contour.

8.4.5.6.6.1 QUANTITATIVE MODELS FOR GIBSON'S ALBATROSS

Francis et al. (2015) concluded that there is cause for concern about the status of the population of Gibson's albatross on the Auckland Islands. Since 2005, the adult population has been declining at 5.7%/yr (95% c.i.: 4.5–6.9%) because of sudden and substantial reductions in adult survival, the proportion of adults breeding, and the proportion of breeding attempts that are successful (Figure

8.31). Forward projections showed that the most important of these to the future status of this population is adult survival (Figure 8.32).

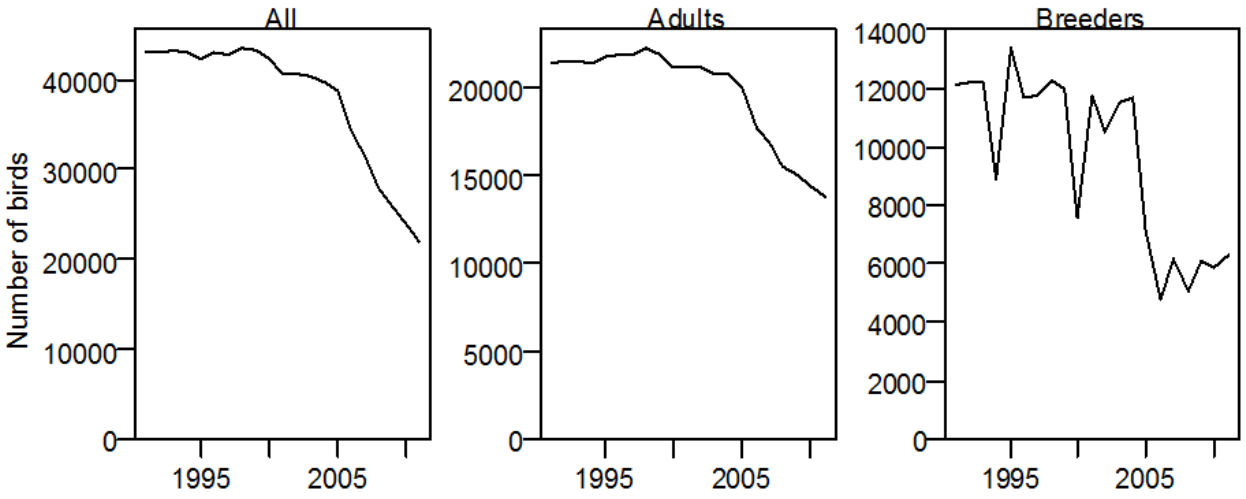


Figure 8.31: Estimated population trajectories for the whole Auckland Islands population of Gibson's wandering albatross. These were calculated by scaling up Francis et al.'s (2015) GIB5 trajectories to match the Walker & Elliott (1999) estimate for the whole population.

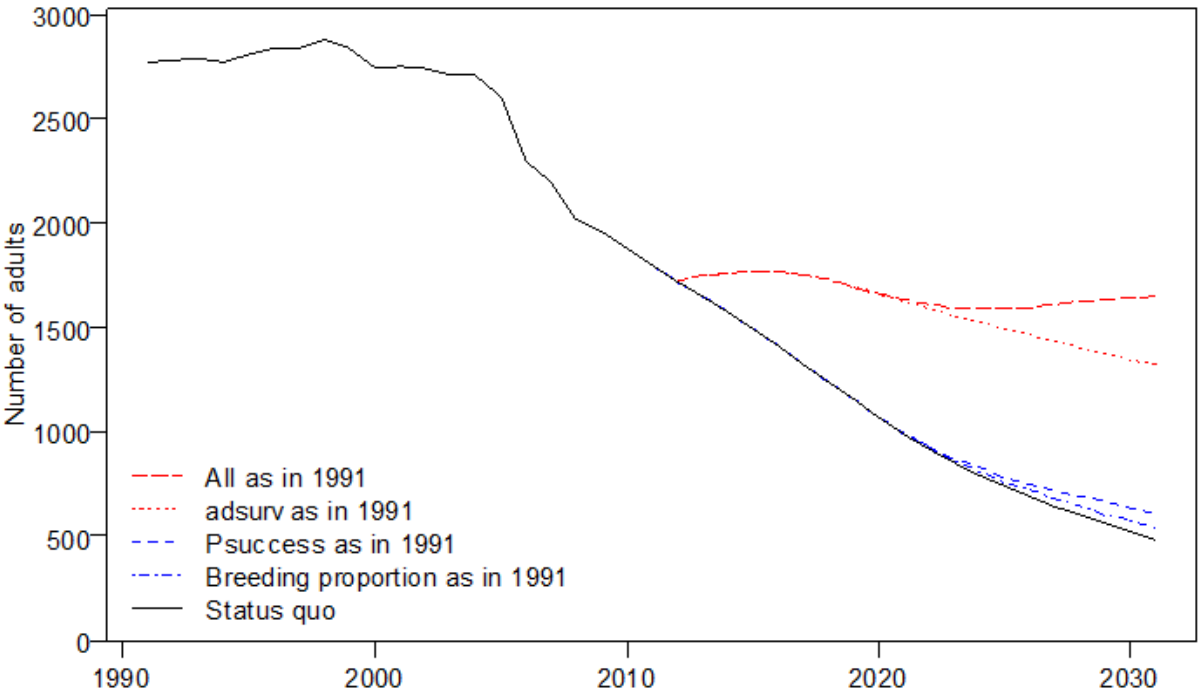


Figure 8.32: Estimated population trajectory for Gibson's albatross adults from Francis et al.'s (2015) model GIB5 with 20-year projections under five alternative scenarios about three demographic parameters: adult survival (adsurv); breeding success (Psuccess); and proportion of adults breeding. These scenarios differ according to whether each parameter remains at its status quo (i.e., 2011) level or recovers immediately to its 1991 level.

The population in 2011 was 64% (58–73%) of its estimated size in 1991. The breeding population dropped sharply in 2005, to 59% of its 1991 level, but has been increasing since 2005 at 4.2% per year (2.3–6.1%). The 2011 breeding population is estimated to be only 54% of the average of 5831 pairs estimated by Walker & Elliott (1999) for 1991–97.

Francis et al. (2015) found it difficult to assess the effect of fisheries mortality on the viability of this population because, although some information exists about captures in New Zealand and Australian waters, the effect of fisheries in international waters is unknown. Three conclusions are possible from the available data: most fisheries mortality of Gibson's is caused by surface longlines; mortality from fishing within the New Zealand EEZ is now probably lower than it was; and there is no indication that the sudden and substantial drops in adult survival, the proportion breeding, and breeding success were caused primarily by fishing.

8.4.3.7 CHATHAM ISLAND ALBATROSS

The Chatham Island albatross breeds only at The Pyramid, a small southern islet in the Chatham Island group (note that a translocation project began in early 2014 transferring chicks to the main Chatham Island with the hopes of establishing a second breeding site). In order to index the population size of the Chatham Islands albatross, nest counts are conducted on The Pyramid. The islet is divided into 19 areas and, within each, every accessible nest site is counted and its status recorded (Scofield et al. 2008a, Fraser et al. 2009b, 2010b).

Nest counts have been conducted when the birds are in the early stages of chick rearing. The total number of Chatham Island albatross nest sites counted in the most recent trip was 5296 (Bell et al. 2017). This result compared closely with previous counts (which have ranged from 5194 to 5407 in late November and early December (Fraser et al. 2011, Table 8.28) indicating a relatively stable number of occupied nests on The Pyramid. The average number of nest sites from 1999–2016 was 5294 (range 5194–5407) (Bell et al. 2017).Chatham Island albatross have been banded on The Pyramid since 1974 and, at each visit, the recaptures have added to the growing number of known-aged birds. This banding record enables an assessment of annual adult mortality. A total of 304 banded Chatham Island albatross were recaptured between 19 November and 2 December 2010 on The Pyramid and a further 50 new Chatham Island albatross were banded during the 2010 trip (Fraser et al. 2011).

To determine foraging movements and behaviour of Chatham Island albatross during the incubation and early chick rearing stages of the breeding season, GPS loggers were applied to breeding birds for the duration of one foraging trip. Where possible, birds were also tagged with a geolocator logger to record activity (i.e., salt water immersion) during foraging trips. The resulting distributional range of Chatham Islands albatross during incubation and early chick rearing from these tracking studies from November to December 2007–09 is given in Figure 8.33 (Fraser et al. 2010b). Deppe et al. (2014) provided a year-by-year analysis of these tracking data.

To track the birds on a longer time-scale during the non-breeding season, geolocation loggers (GLS) were used. These devices have a life span of up to about six years and are intended to remain on the birds for at least one year. They were applied to each banded bird's leg using a plastic band to which the loggers were attached with glue and a cable tie.

MPI has a current project to undertake population specific modelling of adult survival of the Chatham Island albatross (PRO2017-05B).

Table 8.28: (from Fraser et al. 2011 and Bell et al. 2017). Counts of Chatham Island albatross nest sites for the years: 2007 (19–29 November); 2008 (22 November – 7 December); 2009 (9–12 December); 2010 (24–30 December); and 2016 (9-14 November).

	2007	2008	2009	2010	2016
Total nests counted	5 247	5 407	5 194	5 245	5 296

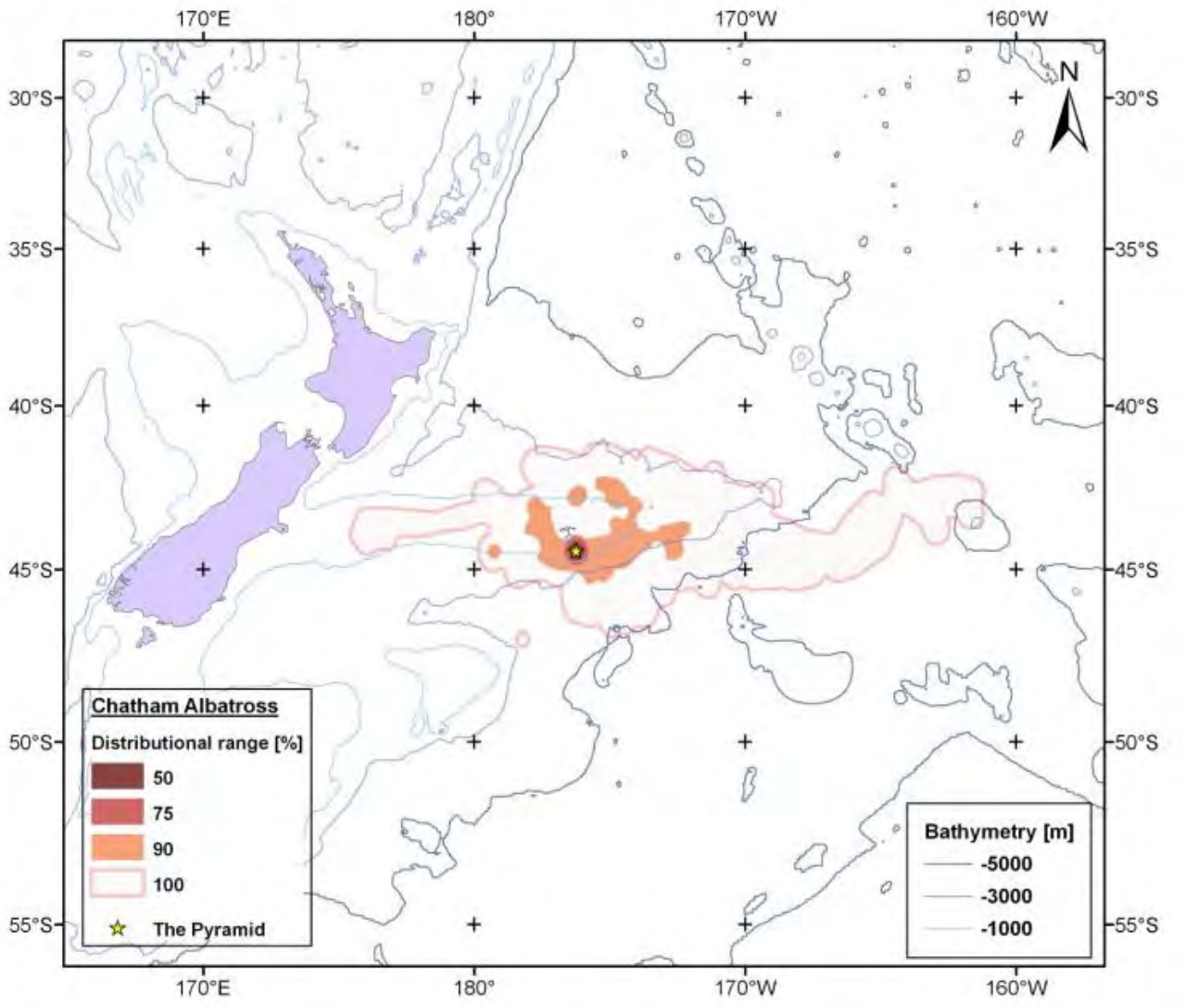


Figure 8.33: (from Fraser et al. 2010b). Distributional range of Chatham Island albatross during incubation and early chick rearing as derived from tracking studies in November/December 2007–09 (n=51 tracks).

8.4.3.8 WHITE-CAPPED ALBATROSS

Repeated population censuses of the white-capped albatrosses breeding in the Auckland Islands were conducted in the month of December between 2006 and 2010, and the month of January in 2012 and 2013, using aerial photography (Baker et al. 2007b, 2008a, 2009b, 2011b, 2013). These population censuses were carried out to estimate population size and track population trends. Photo montages were created from the aerial photography and counted by an observer. Counts of photo montages in all years except 2006 were undertaken by one observer only. Multiple counts of photo montages from the December 2006 census were undertaken to estimate counter variability associated with miscounting and misidentifying white spots on the ground as birds. Ground-truthing was conducted to determine the number of birds sitting or standing on nests, the number of pairs (partners accompanying an incubating bird), and the number of loafers present in the colony.

2006–10**:** In 2010, the total count of nesting white-capped albatrosses was estimated to be 72 635 (95% c.i.: 72 096–73 174), 4370 (4238–4502) and 117 (95–139) annual breeding pairs at Disappointment Island, South West Cape and Adams Island, respectively, giving a total for these sites of 77 122 (76 567–77 677) breeding pairs (Table 8.26). The counts of nesting white-capped albatross over the previous four years were significantly lower than the counts taken in 2006, when a total of 117 197 breeding pairs were present at the

Auckland Islands. These differences in counts may represent normal inter-annual variation in breeding rather than indicating a decline in numbers due to fisheries mortalities (Baker et al. 2011b).

2011–16**:** Surveys suggested 99 776 breeding pairs in 2011, 118 098 in 2012, 95 278 in 2013, 101 798 in 2014, 96 623 in 2015 and 91 117 in 2016. However, evidence from a series of close-up photographs taken each year over the entire series indicates that the number of non-breeding birds present in the colonies differed somewhat between December and January. The proportion was very low in December counts (1–2% of birds present) to 7 and 15% for the January counts taken in 2012 and 2013, respectively. Estimated annual counts for all three breeding sites in the Auckland Islands were adjusted to account for the presence of non-breeding birds (Table 8.29). These adjusted figures were used as inputs into models used for assessment of population trend. The population size estimates computed from a TRIM model indicate an average growth rate of -1.73% per year (λ = 0.9827 ± 0.001); assessed by TRIM as moderate decline. However, a simple linear trend analysis, as performed by TRIM is not well suited to a dataset with high inter-annual variability. Trend analysis using regression splines is more appropriate to such datasets and showed no evidence for systematic monotonic decline over the nine years of the study, therefore providing support to the null hypotheses of no trend (stability) in the total population. Full details are provided by Baker et al. (2013, 2014b, 2015a, *In Press*).

Further aerial surveys were conducted in 2016 (Baker & Jensz 2016) and 2017. The resulting aerial photographs are being analysed under MPI project SEA2016-29.

A marked population of breeding adult white-capped albatross has been established at Disappointment Island over 2015–18 in order to estimate their demographic parameters in the long term, including adult survival (Parker et al. 2016, 2018). To date, a total of 521 breeding white-capped albatrosses have been banded in the four annual visits to Disappointment Island (Parker et al. 2018).

Torres et al. (2011) tracked breeding white-capped albatrosses at South West Cape (main Auckland Island) during the guard stage using GPS tags and found that foraging effort was concentrated on an area to the east of the Auckland Islands over the edge of the shelf (approximately 250 m water depth).

Table 8.29: (after Baker et al. 2013, 2014b, 2015a, *In Press*). Aerial-photographic counts of breeding pairs of white-capped albatrosses on three islands in the Auckland Islands group in December 2006–14.

Year	Adams	Disappointment	SW Cape	Total	95% limits	Adjusted for loafers
2006	–	110 649	6 548	117 197	116 570–117 823	116 025
2007	79	86 080	4 786	90 945	90 342–91 548	90 036
2008	131	91 694	5 264	97 089	96 466–97 712	96 118
2009	132	70 569	4 161	74 862	74 315–75 409	73 838
2010	117	72 635	4 370	77 122	76 567–77 677	76 119
2011	178	93 752	5 846	99 776	99 144–100 408	92 692
2012	215	111 312	6 571	118 098	117 411–118 785	102 273
2013	184	89 552	5 542	95 278	94 661–95 895	74 031
2014	193	96 864	4 741	101 798	101 160–102 436	95 894
2015	176	91 554	4 893	96 623	90 949-92 159	87 057
2016	171	85 510	5 436	91 117	84 925-86 095	82 005

8.4.3.8.1 QUANTITATIVE MODELS FOR WHITE-CAPPED ALBATROSS

Francis (2012) described quantitative models for white-capped albatross, New Zealand's most numerous breeding albatross, and the most frequently captured, focusing on the population breeding at the Auckland Islands. After a correction for a probable bias introduced by sampling at different times of day in one of the surveys, aerial photographic counts by Baker et al. (2007b, 2008b, 2009b, and 2010b) suggested that the adult population declined at about 9.8% per year between 2006 and 2009. However, this estimate is imprecise and is not easily reconciled with the high adult survival rate (0.96) estimated from mark-recapture data. Francis (2012) also compared the trend with his estimate of the global fishing-related fatalities of white-capped albatross (slightly over 17 000 birds per year, about 30% of which is taken in New Zealand fisheries) and found that fishing-related fatalities were insufficient to account for the number of deaths implied by a decline of 9.8% per year (roughly 22 000 birds per year over the study period). The scarcity of information on cryptic mortality makes these estimates and conclusions uncertain, however. Since this modelling was conducted, further counts of white-capped albatross have been conducted (Figure 8.34 compiled from data presented in Baker et al. *In Press*), which showed considerable annual variation. Baker et al. (*In Press*) consider that the substantial year to year variation in counts is real, that trend analyses appropriate in this situation support the null hypothesis of no trend in the population and that the trend should be considered uncertain.

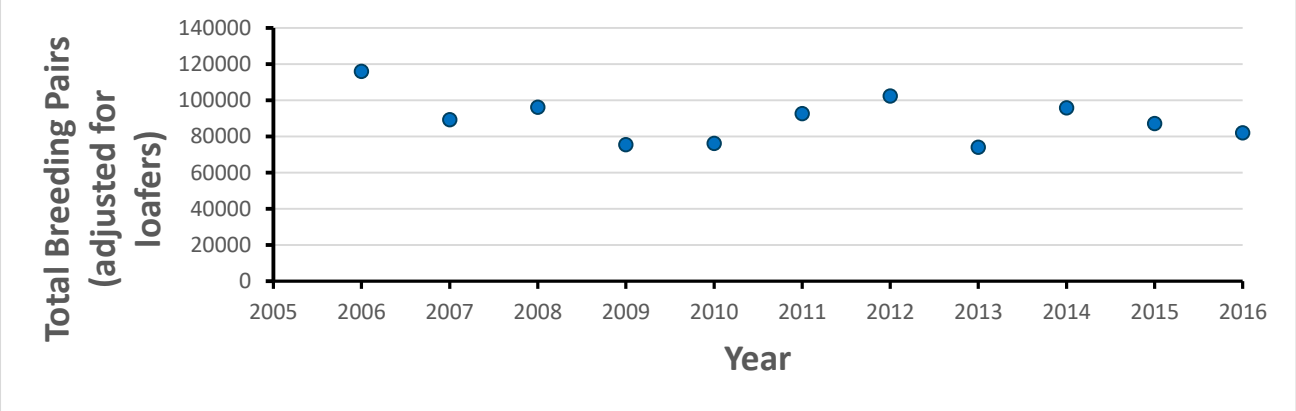


Figure 8.34: (Data from Baker et al. *In Press*). Total counts of white-capped albatross at the Auckland Islands (as adjusted for the presence of non-breeding birds).

8.4.3.9 NORTHERN BULLER'S ALBATROSS

The Forty-Fours, a small group of islands located about 35 km east of Chatham Island, are home to the main breeding population of northern Buller's albatross *Thalassarche bulleri platei*. The northern Buller's albatross nest estimate on the Forty-Fours for 2007 was 15 238 (Scofield et al. 2008b), for 2008 was 14 674 (Fraser et al. 2009a), and for 2009 was 14 185 (Fraser et al. 2010a). Fixed grids sampled each year also confirmed the consistent population count (Fraser et al. 2010a).

In 2016, aerial photography was compared with ground counts to determine the population size of northern Buller's albatross on the Forty-Fours and The Sisters. The aerial survey estimated a total of 17 969 breeding pairs (2 646 pairs at the Sisters and 15 322 pairs at the Forty-Fours) after correction with aerial close-ups, and 16 138 breeding pairs (2 366 pairs at the Sisters and 13 771 pairs at the Forty-Fours) after correction using ground counts (Baker et al. 2017). A ground survey of the Forty-Fours in December 2016 updated the population size for northern Buller's albatross with an estimate of 17 682 nest sites, although this total included empty nests and nests for which contents were not determined. The total for nests with either an egg or with signs of breeding was 16 492 (Bell et al. 2017).

In November/December 2017 a full census of northern Buller's albatross at the Sisters revealed a total of 3 158 nests with evidence of breeding (Bell et al. 2018). The overall Chatham Islands population of northern Buller's albatross is close to 20 000 pairs (Bell et al. 2018).

Additionally, a small population of northern Buller's albatross breeds at Rosemary Rock within the Three Kings group. Frost et al. (2018) provided a summary of all counts for this population and estimated 34 active nests in November 2017 using aerial photography.

8.4.3.10 ANTIPODEAN ALBATROSS

Antipodean albatross is an endemic taxon breeding primarily at Antipodes Island, with a very small number of pairs breeding at Campbell Island and the Chatham Islands. Monitoring of the Antipodes Island population began in 1994 and has been continued on a near-annual basis since inception. The most recent report of this work (Elliott & Walker 2017) provided summary information on the Antipodes Island population spanning 1994 to 2017 for most statistics.

The Antipodean albatross population at Antipodes Island mirrored the trajectory of Gibson's albatross at Adams Island in that following a period of population growth up to 2004 the population declined markedly from 2005, and although the rate of decline has slowed the population in 2017 was at the lowest level since the study began (Table 8.30, Figure 8.35).

Adult survival was generally high (approximately 0.96) and consistent between sexes up to 2004, from which point survival declined to 0.90-0.95 for males and to approximately 0.81-0.92 for females (Figure 8.36, Elliott & Walker 2017).

Table 8.30: (taken from Elliott & Walker 2017). Antipodean wandering albatross nests with eggs in February in three areas on Antipodes Island between 1994 and 2017. MCBA = Marked Census Block A.

Year	Study area	Block 32	Subtotal	MCBA	Total
1994	114	125	239	544*	783
1995	156	185	341	482*	823
1996	154	133	287	418*	705
1997	150			464*	
1998	160			534	
1999	142			479	
2000	119	130	249	462	
2001	160	141	301	443	711
2002	148	178	326	605	744
2003	214	187	401	608	931
2004	216	249	465	755	1009
2005	211	186	397	613	1220
2006					1010
2007	119	127	246		
2008	165	135	300		
2009	98	120	218		
2010	106	101	207		
2011	88	108	196		
2012	95	104	199	345	543
2013	88	93	181	297	478
2014	91	103	194	341	535
2015	73	86	159	291	450
2016	100	92	192	291	483
2017	57	82	139	230	369

* estimated (see Walker & Elliott 1998)

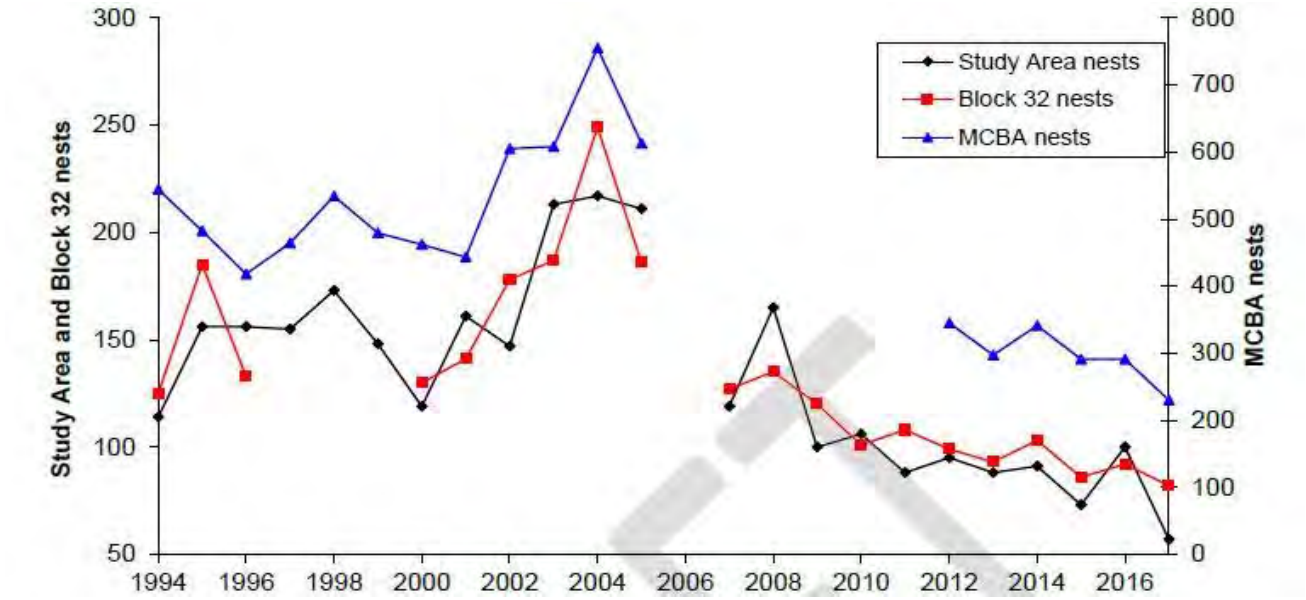


Figure 8.35: (taken from Elliott & Walker 2017). The number of Antipodean wandering albatross nests in three blocks on Antipodes Island since 1994. MCBA = Marked Census Block A

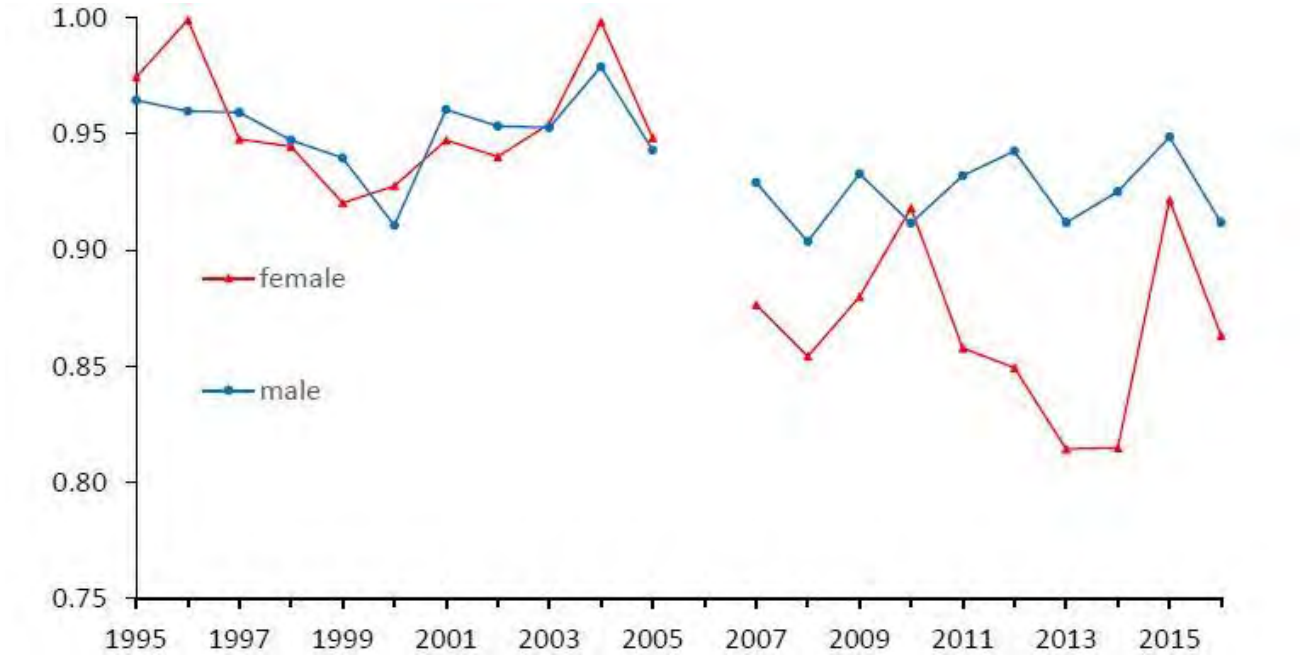


Figure 8.36: (taken from Elliott & Walker 2017). Estimated annual survival of Antipodean wandering albatross on Antipodes Island since 1994. Note that as the island wasn't visited in 2006, survival estimates for 2006 and 2007 were estimated from the survival over a 2-year period and then equally apportioned amongst the two years.

The at-sea distribution of Antipodean albatross has also shifted over the course of the study. A total of 65 birds were tracked using satellite telemetry between 1996 and 2004 (Walker & Elliott 2006) and the data from this earlier period have been compared with light-based geolocation data from 50 birds tracked from 2011 to 2017. For breeding birds, females have expanded their range to the north post-2004, while males appear to have their range in all directions post-2004 (Figure 8.37)

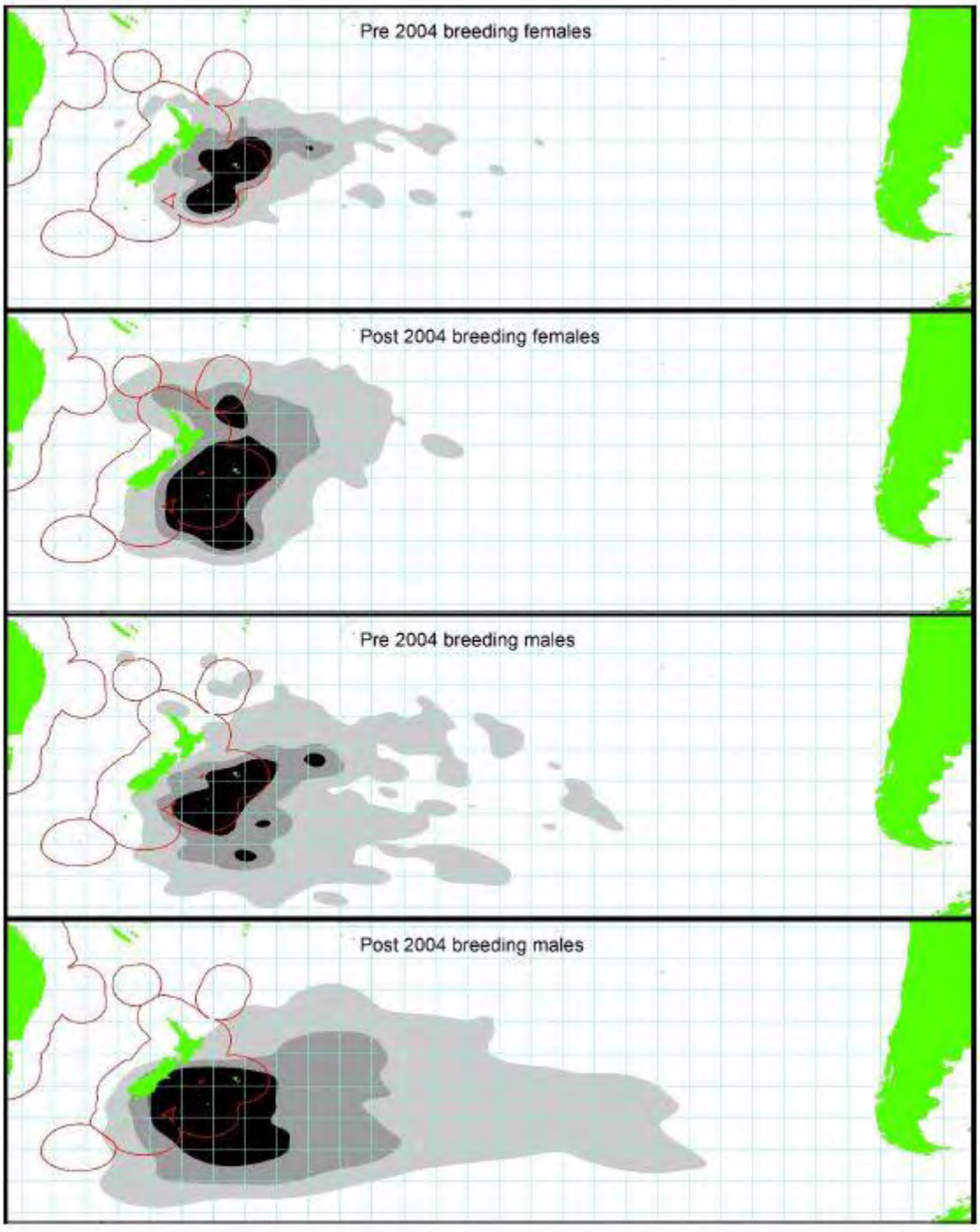


Figure 8.37: (taken from Elliott & Walker 2017). Kernel density plots of breeding Antipodean wandering albatrosses tracked in 1996-2004 and in 2011-17. Black indicates the 50% contour, dark grey the 75% contour, and light grey the 95% contour.

For non-breeding Antipodean albatrosses, the differences in distribution prior to 2004 and post-2004 were more marked. Post-2004 females travelled to waters off South America, a strategy that was not apparent prior to 2004, and males post-2004 utilised an area in the mid-pacific and a much larger area off South America compared to birds prior to 2004 (Figure 8.38, Elliott & Walker 2017).

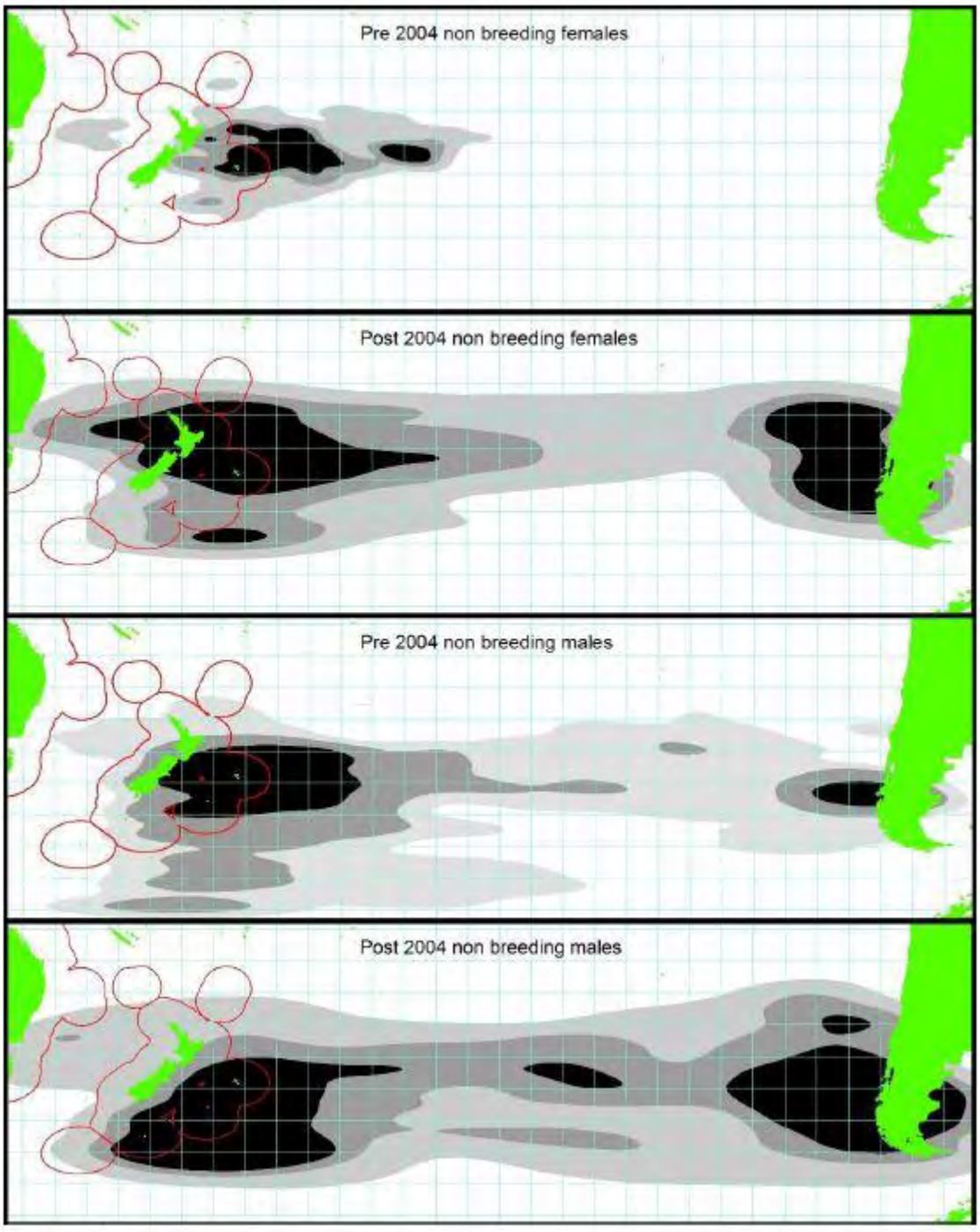


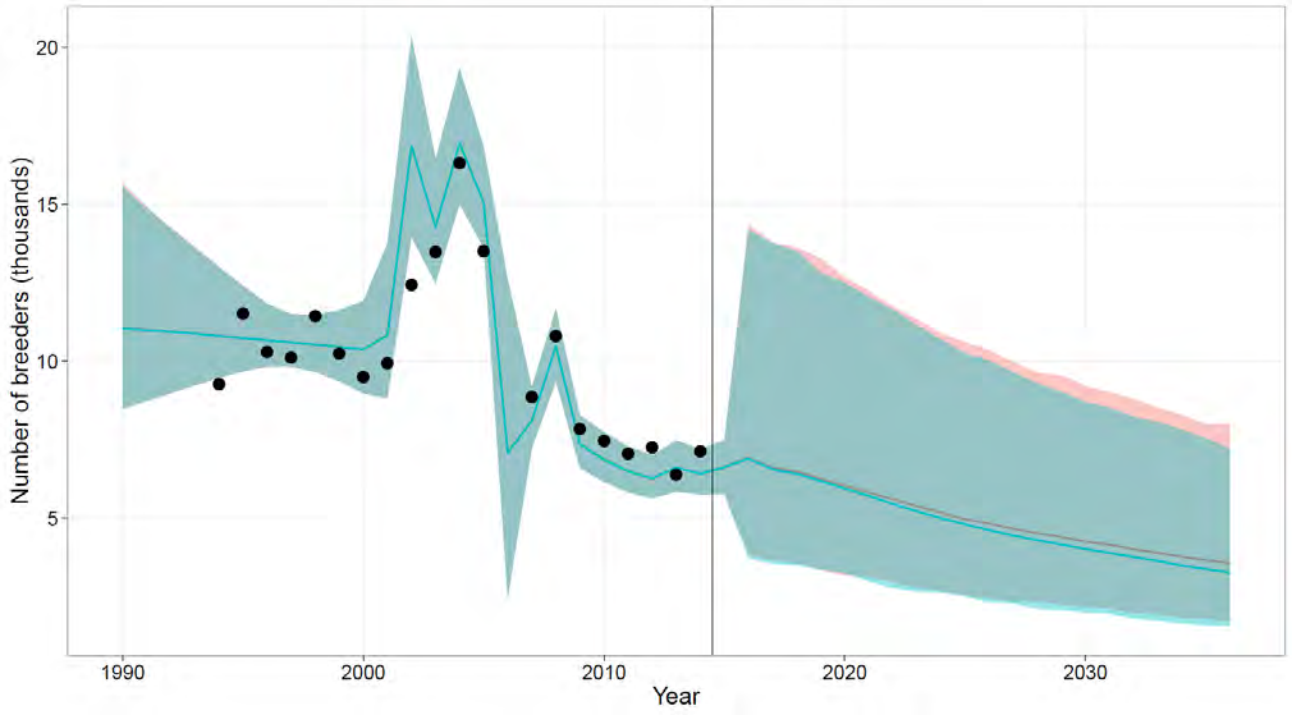
Figure 8.38: (taken from Elliott & Walker 2017). Kernel density plots of non-breeding Antipodean wandering albatrosses tracked in 1996-2004 and in 2011-17. Black indicates the 50% contour, dark grey the 75% contour, and light grey the 95% contour.

8.4.3.10.1 QUANTITATIVE MODELS FOR ANTIPODEAN ALBATROSS

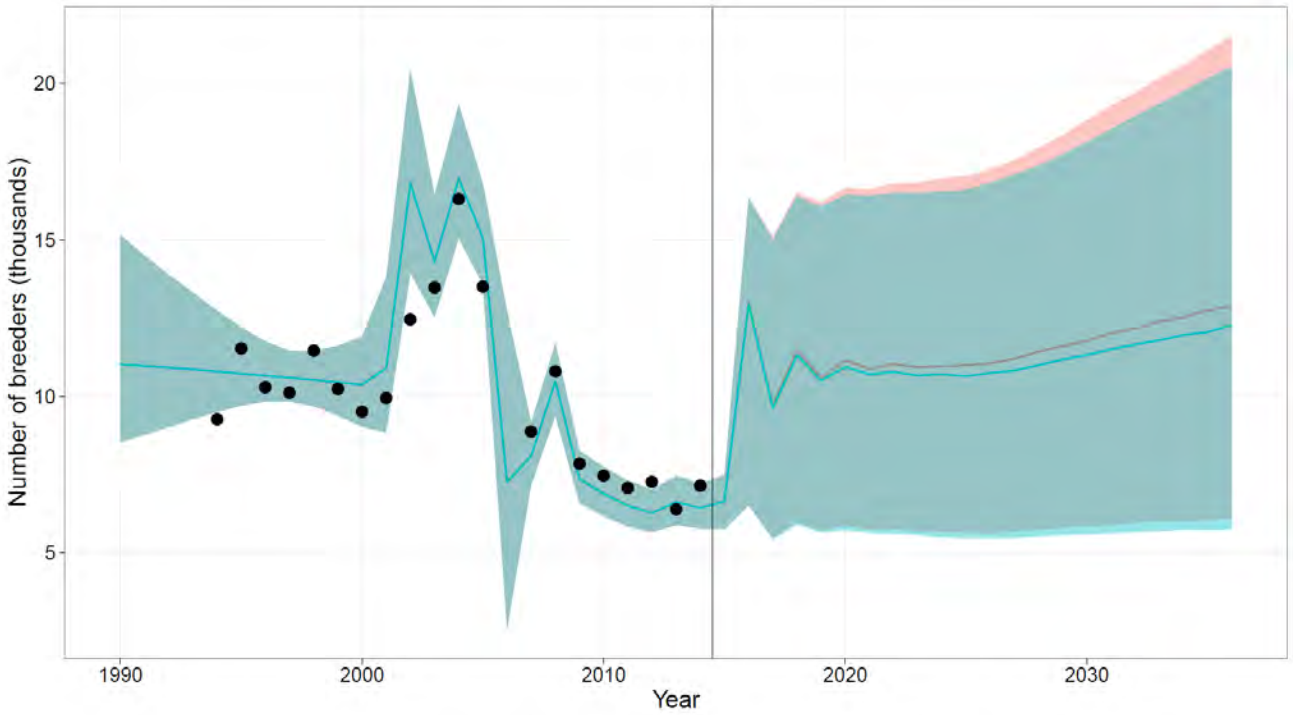
Edwards et al. (2017) developed a quantitative demographic model for the Antipodean albatross to estimate vital rates and predict population changes into the future given the observed declines in the population since 2005 (Elliott & Walker 2014). The model was parameterised using extensive mark-recapture and census data, which allowed the estimation of time-variant survivorship and breeding parameters.

Edwards et al. (2017) found that although the survivorship has changed over time, it was apparent from both the modelling and an empirical review of the data that changes in the breeding rate, success and age of first breeding are primarily responsible for temporal changes in the population abundance. The model predicted that the population has declined in numbers since 2007 and will continue to do so unless these demographic vital rates recover (Figure 8.39). Furthermore, it predicted that reduced adult survivorship as a result of fishing induced mortalities within New Zealand waters is likely to be having a negligible impact, although the impact of unquantified mortalities arising from potential species misidentification or captures outside of New Zealand waters could not be evaluated due to a lack of data (Edwards et al. 2017). The model demonstrates that domestic captures of Antipodean albatrosses are insufficient to be the cause of the population decline. The captures of Antipodean albatrosses outside of New Zealand is a key factor that needs to be addressed to reverse the current population trajectory.

(a) Basecase model with vital rates sampled from 2004–13



(b) Basecase model with vital rates from 1995–2004



(c) Female-only model with vital rates sampled from 2004–13

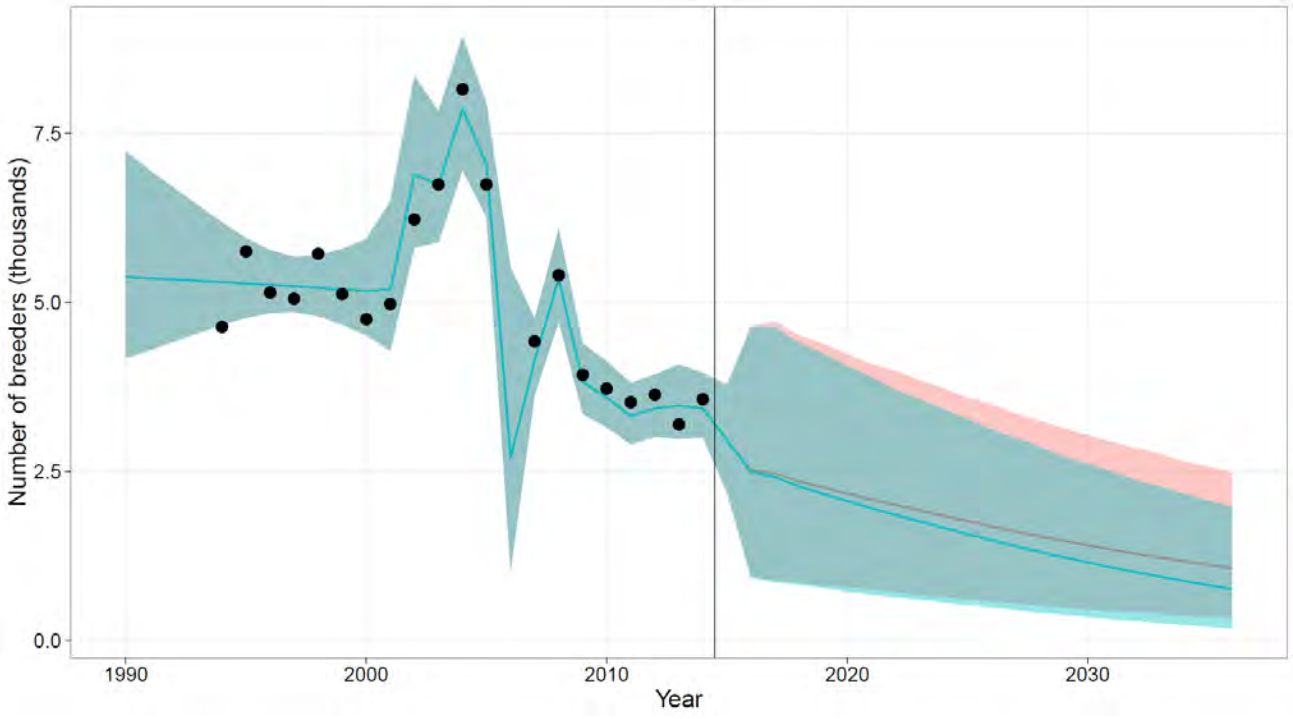


Figure 8.39: Predicted population dynamics for the whole island, showing the number of Antipodean albatross breeders from 2015–34. Predicted dynamics with and without fishing pressure are superimposed. The median and 95% confidence intervals are shown for each scenario. The vertical line indicates the limit of the empirical data, with subsequent dynamics representing an extrapolation. Empirical census data are also shown.

8.4.3.11 NORTHERN GIANT PETREL

The Forty-Fours in the Chatham Islands support the largest breeding colony of northern giant petrel in New Zealand. Northern giant petrels nest mainly in the north-eastern part of the island along the cliff tops. Estimates of nests with live chicks in two areas assessed over several years were: 431 in November 2007 (Scofield et al. 2008b); 335 in November 2008 (Fraser et al. 2009a), 270 in December 2009 (Fraser et al. 2010a) and 398 in December 2016 (Bell et al. 2017). It should be noted that because these counts were carried out during the chick-rearing phase of the breeding season they represent minimum estimates of breeding totals for the areas assessed, as some breeding attempts will have failed by the time the counts were made. Bell et al. (2017) conducted a complete count of nests with chicks at the Forty-Fours in December 2016 (1 235 nests), and corrected this total using a breeding success rate of 60.1% to calculate a total breeding population of 1 977 pairs of northern giant petrel.

Elsewhere in New Zealand, C.J.R. Robertson (cited in Imber 1994) estimated 336 nests at the Sisters (Chatham Islands) in 1976, and C.J.R. Robertson (cited in Hunter 1986) estimated a total of 2 150 breeding pairs for the Chatham Islands, no date specified. Wiltshire & Scofield (2000) and Wiltshire & Hamilton (2003), based on complete nest counts, estimated totals of 234 breeding pairs at Campbell Island in 1996-97 and 233 breeding pairs at Antipodes Island in 2000-01, respectively. Taylor (2000) concluded that at least 50 pairs breed at the Auckland Islands. However, this estimate did not include Adams Island where there were approximately 35 breeding pairs in 2003-04, centred on Fairchild's Garden towards the western end of the island (K. Walker pers. comm.). More recently, Parker et al. (2016) recorded 216 northern giant petrel chicks across eight of 15 islands in the Auckland Islands group in December 2015 and January 2016, with Enderby Island supporting the largest total of 96 chicks. Using a correction factor to account for breeding success, Parker et al. (2016) estimated an Auckland Islands breeding population of approximately 340 pairs.

8.4.3.12 YELLOW-EYED PENGUIN

The yellow-eyed penguin is an endemic species with a breeding range that extends from Banks Peninsula in the north, along the southeast coast of the South Island, Stewart Island and outlying islands, to the Auckland Islands and Campbell Island in the south. Yellow-eyed penguin population estimate data up to 2011 have been collated and summarised by Ellenberg et al. (2012) for the four main breeding areas: South Island, Stewart Island and outliers, the Auckland islands and Campbell Island.

Breeding on the South Island has been further partitioned into four distinct regions. In the north, six nests were found on Banks Peninsula in 2011-12 (Ellenberg et al. 2012). In North Otago, nest numbers peaked in 2008-09 at 77, and with intensive management at two North Otago colonies, nest numbers have increased regionally since 1992 (Ellenberg et al. 2012). In contrast and overall, nest numbers on the Otago Peninsula have declined significantly since 1992, with the strongest decline, of about -60% observed at Sandfly Bay. Nest numbers peaked at approximately 250 in 1996-97 and by 2001-12 were approximately 115 (Ellenberg et al. 2012). The Catlins region has been fully censused on four occasions: 1997-98, 2001-02, 2007-08 and 2011-12. These data suggest a stable yellow-eyed penguin population with between 190 and 240 breeding pairs.

Darby (2003) reported estimates of 170-230 pairs of yellow-eyed penguin on Stewart Island and its outliers and of 50-80 pairs on Codfish Island during the period 1984-1994. Massaro & Blair (2003) reported the numbers of yellow-eyed penguin pairs for the period 1999-2001, with totals of 79 pairs for Stewart island and 99 pairs on outlying islands, including 61 pairs on Codfish Island. In 2008-09 King et al. (2012) reported 77 breeding pairs throughout Stewart Island, but noted that along the northern coast of Stewart Island nest numbers declined by 27% between 1999-01 and 2008-09, and Ellenberg et al. (2012) noted that for Codfish Island yellow-eyed-penguin numbers declined from 61 pairs in 1999-01 (Massaro & Blair 2003) to 39 pairs in 2011-12.

At the Auckland Islands, Moore (1992) counted yellow-eyed penguins at landing sites at islands in the north of the archipelago and at Adams Island in the south in 1989 and estimated a population of 420-470 pairs. Moore (1992) further suggested that inclusion of breeding pairs along the eastern coast of main Auckland Island could take the overall population to 520-570 breeding pairs. Chilvers (2014) provided annual counts of yellow-eyed penguins at Sandy Bay, Enderby Island, which showed an increase in penguin numbers over the period 2001-12.

Moore et al. (2001) reported numbers of yellow-eyed penguins at Campbell Island as 2 277 individuals in 1988 and 1 347 birds in 1992, a decrease of 41%. Counts of penguins at 11 landing sites indicated that the Campbell Island population remained relatively low until 1994 and then began to increase through to 1998 (Moore et al. 2001).

Overall and based on the summaries above, Ellenberg et al. (2012) provided breeding pair totals of 400-600 for the South Island, 180 for Stewart Island and its outliers, 520-570 for the Auckland Islands and 350-540 for Campbell Island. On this basis, the sub-Antarctic yellow-eyed penguin population represents approximately 60% of the overall New Zealand population.

Mattern et al. (2007) described foraging routes and benthic foraging behaviour in yellow-eyed penguins tracked from Bushy Beach near Oamaru in 2003-05 and found that birds performed daily trips ranging in distance from 12 to 20 km from the coast, with shorter (less than 7 km) trips in the evening. Of all dives, 87% were benthic, and birds exhibited consistent foraging routes and destinations across consecutive foraging trips. Mattern et al. (2013) further explored foraging in yellow-eyed penguins, tracking birds from the Otago Peninsula, and found that some birds foraged benthically (>87% of dives) along straight trajectories (Figure 8.40).

Disease, sometimes aggravated by starvation or relatively poor dietary provisioning, has been demonstrated in mortality events and population declines in yellow eyed-penguins (Houston 2005, King et al. 2012, Argilla et al. 2013, Alley et al. 2017). Mattern et al. (2017) modelled climate effects on the fluctuations of a yellow-eyed penguin population at the Otago Peninsula and found that sea surface temperature accounted for 33% of the variation in penguin numbers, affecting both adult and fledgling survival: increasing sea surface temperatures since the mid-1990s was accompanied by a reduction in survival rates and a population decline.

8.4.3.12.1 QUANTITATIVE MODELS FOR YELLOW-EYED PENGUIN

Maunder et al. (2007) sought to assess the impact of commercial fisheries on the Otago Peninsula yellow-eyed penguins using mark-recapture data within a population dynamics model. They found the data available at that time inadequate to assess fisheries impacts but evaluated the likely utility of additional information on annual survival or an estimate of bycatch for a single year. Including auxiliary information on average survival in the absence of fishing allowed estimation of the fishery impact, but with poor precision. Including an estimate of fishery-related mortality for a single year improved the precision in the estimated fishery impact. The authors concluded that there was insufficient information to determine the impact of fisheries on yellow-eyed penguins and that quantifying fishing-related mortality over several years was required to undertake such an assessment using a population modelling approach.

Mattern et al. (2017) used a Bayesian modelling approach to assess the influence of climate change on population trends of yellow-eyed penguin over the last 30 years at the Otago Peninsula. They found that sea surface temperature (SST) was the key factor influencing survival of both adult birds and fledglings. Increasing SST since the mid-1990s was accompanied by a reduction in survival rates and a population decline. The population model showed that 33% of the variation in population numbers could be explained by SST alone. Mattern et al. (2017) suggested that the effect of SST significantly increased pressure on the penguin population such that the population becomes less resilient to non-climate related impacts, such as fisheries interactions, habitat degradation and human disturbance. Mattern et al. (2017) noted, however, that due to the absence of quantifiable data on fisheries impacts, habitat changes and human disturbance it was difficult to delineate the extent of the contribution of these factors to a declining population.

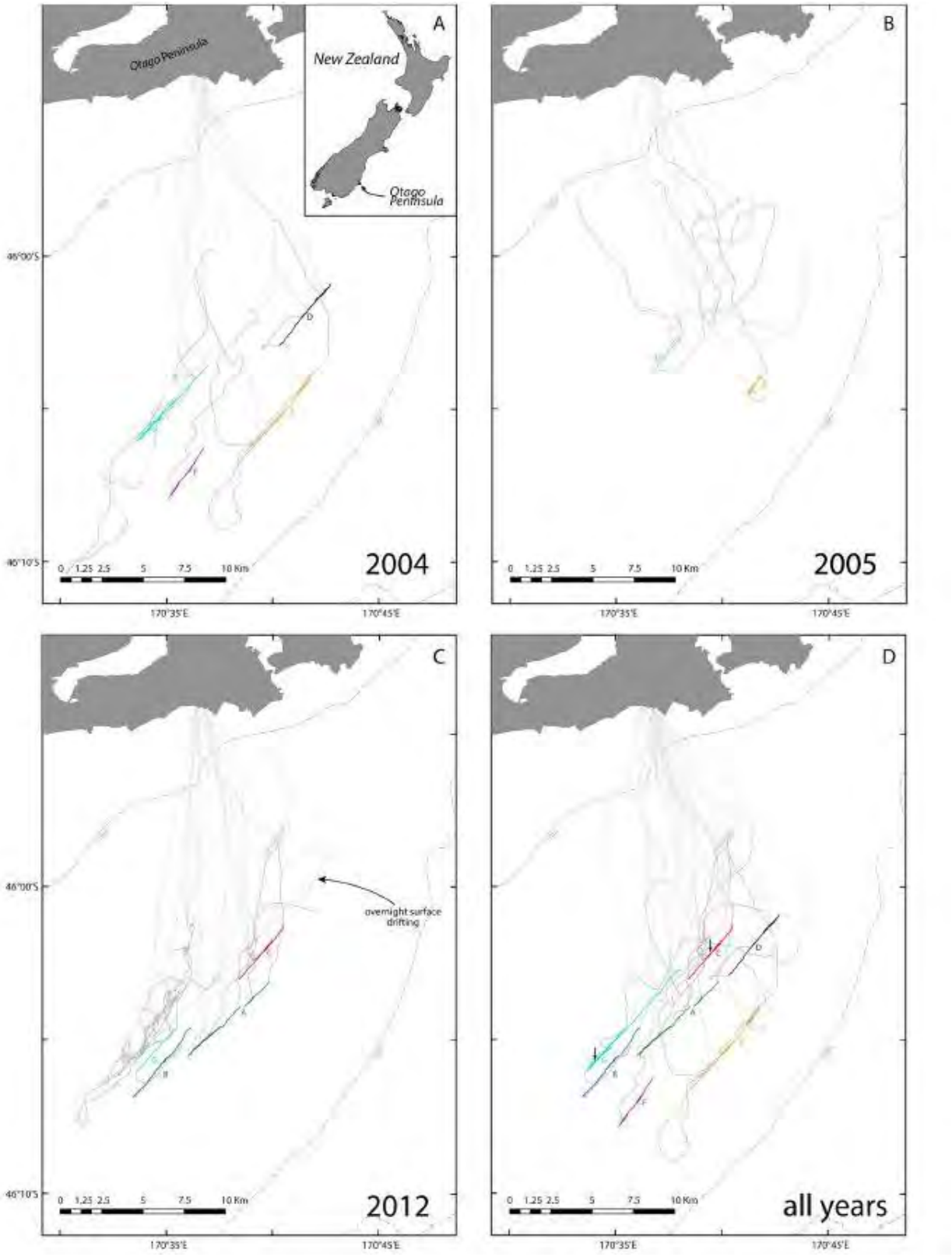


Figure 8.40: (taken from Mattern et al. 2013). Mid-shelf foraging tracks of yellow-eyed penguins recorded in 2004 (A), 2005 (B) and 2012 (C) that feature straight-line patterns. Foraging track segments in light grey represent outgoing and incoming stages of foraging trips; dark grey segments highlight the foraging stage. Dashed line segments indicate where linearity of the track is a result of interpolation. Track portions that met line criteria (see Methods) are highlighted in different colours; line identifiers shown in capital letters of the same colour. Small arrows in (D) indicate sites of ROV deployments in February 2013. Trips with lines from all three seasons are combined in (D).

8.4.3.13 OTAGO SHAG

Based on analyses of genetic and morphological evidence, Rawlence et al. (2016) argued that Stewart Island shag *Leucocarbo chalconotus* be split into two species, Foveaux shag *Leucocarbo stewartia* and Otago shag *Leucocarbo chalconotus*. Otago shag is distributed along the south-eastern coast of South Island, from The Sisters rocks just offshore on the north side of Teahimate Bay in the south to Oamaru in the north (Lalas & Perriman 2009), but with occasional sightings extending further north as far as Lake Ellesmere (Crossland 2012). Lalas & Perriman (2009) identified six breeding sites for Otago shag (north to south: Maukiekie Island, Taiaroa Head, Wharekakahu, Gull Rocks, Green Island and Kinakina Island), with a maximal overall population of approximately 1 900 nests in 1987, falling to approximately 1 150-1 300 nests in 2005-07, although these totals did not include the KinaKina Island breeding site, which Lalas & Perriman (2009) estimated could support 200-300 nests.

8.4.4 MANAGING FISHERIES INTERACTIONS

New Zealand had taken steps to reduce incidental captures of seabirds before the advent of the IPOA in 1999 and the NPOA in 2004. For example, regulations were put in place under the Fisheries Act to prohibit drift net fishing in 1991 and prohibit the use of netsonde monitoring cables ('third wires') in trawl fisheries in 1992. The use of tori lines (streamer lines designed to scare seabirds away from baited hooks) was made mandatory in all tuna longline fisheries in 1992.

.

The fishing industry also undertook several initiatives to reduce captures, including funding research into new or improved mitigation measures, and adopting voluntary codes of practice and best practice fishing methods. Codes of practice have been in place in the joint venture tuna longline fishery since 1997–98, requiring, among other things, longlines to be set at night and a voluntary upper limit on the incidental catch of seabirds. That limit was steadily reduced from 160 'at risk' seabirds in 1997–98, to In 2002, MFish, DOC, and stakeholders began working with other countries to reduce the incidental catch of seabirds. As a result, a group called Southern Seabird Solutions was formed and formally established as a Trust in 2003 (http://www.southernseabirds.org) and received royal patronage in 2012. Southern Seabird Solutions exists to promote responsible fishing practices that avoid the incidental capture of seabirds in New Zealand and the southern ocean. Membership includes representatives from the commercial fishing industry, environmental and conservation groups, and government departments. The 75 in 2003–04. Most vessels in the domestic longline tuna fishery had also voluntarily adopted night setting by 2004. A code of practice was in place for the ling auto-line fishery by 2002–03. Other early initiatives included reduced deck lighting, the use of thawed rather than frozen baits, sound deterrents, discharging of offal away from setting and hauling, weighted branch lines, different gear hauling techniques and line shooters. Current regulated and voluntary initiatives are summarised by fishery in Table 8.31.

Trust's vision is that: *All fishers in the Southern Hemisphere avoid the capture of seabirds*, and this is underpinned by the strategic goals on: Culture Change; Supporting Collaboration; Mitigation Development and Knowledge Transfer; Recognising Success; and Strengthening the Trust.

Building on these initiatives, New Zealand's 2004 NPOA established a more comprehensive framework to reducing incidental captures approach across all fisheries (because focusing on longline fisheries like the IPOA was considered neither equitable nor sufficient).

Table 8.31: (from MPI 2013, NPOA-seabirds). Summary of current mitigation measures applied to New Zealand vessels fishing in New Zealand waters to avoid incidental seabird captures. R, regulated; SM, required via a self-managed regime (non-regulatory, but required by industry organisation and audited

independently by government); V, voluntary with at least some use known; N/A, measure not relevant to the fishery; years in parentheses indicate year of implementation; *, part of a vessel management plan (VMP). Note, this table may not capture all voluntary measures adopted by fishers.

Mitigation measure	Surface longline	Bottom longline	Trawl ≥28 m	Trawl <28 m	Set net	Notes
Netsonde cable prohibition	N/A	N/A	R (1992)	R (1992)	N/A	Netsonde cables also called third wires
Streamer (tori) lines	R	R	N/A	N/A	N/A	
Additional streamer line	–	–	N/A	N/A	N/A	
Night setting	R (or line weighting)	R (or line weighting)	–	–	–	Longlines must use night setting if not line weighting, or *vice-versa*
Line weighting	R (or night setting)	R (or night setting)	N/A	N/A	N/A	
Seabird scaring device	N/A	N/A	R (2006)	R?	N/A	To prevent warp captures and collisions
Additional bird scaring device	N/A	N/A	SM (2008)*	–	N/A	
Dyed bait	V	–	N/A	N/A	N/A	
Offal management	V	R	SM (2008)*	–	–	
VMPs			SM (2008)	V	–	Some VMPs developed for vessels <28m
Code of Practice	V	–	VMP	–	–	

Note: A vessel management plan (VMP) is a vessel-specific seabird risk management plan that specifies seabird mitigation devices to be used, operational management requirements to minimise the attraction of seabirds to vessels, and incident response requirements and other techniques or processes in place to minimise risk to seabirds from fishing operations

This 2004 NPOA included two goals that set the overall direction:

1. To ensure that the long-term viability of protected seabird species is not threatened by their incidental catch in New Zealand fisheries waters or by New Zealand flagged vessels in high seas fisheries; and
2. To further reduce incidental catch of protected seabird species as far as possible, taking into account advances in technology, knowledge and financial implications.

Together the two goals established the NPOA as a long-term strategy. The second goal was designed to build on the first goal by promoting and encouraging the reduction of incidental catch beyond the level that is necessary to ensure long-term viability. The goals recognised that, although seabird deaths may be accidentally caused by fishing, most seabirds are absolutely protected under the Wildlife Act. The second goal balances the need to continue reducing incidental catch against the factors that influence how this can be achieved in practice (e.g., advances in technology and the costs of mitigation). The scope of the 2004 NPOA included:

- all seabird species absolutely or partially protected under the Wildlife Act;
- commercial and non-commercial fisheries;
- all New Zealand fisheries waters; and
- high seas fisheries in which New Zealand flagged vessels participate, or where foreign flagged vessels catch protected seabird species.

Specific objectives were established in the 2004 NPOA as follows:

- Implement efficient and effective management measures to achieve the goals of the NPOA, using best practice measures where possible.
- Ensure that appropriate incentives and penalties are in place so that fishers comply with management measures.
- Establish mandatory bycatch limits for seabird species where they are assessed to be an efficient and effective management measure and there is sufficient information to enable an appropriate limit to be set.
- Ensure that there is sufficient, reliable information available for the effective implementation and monitoring of management measures.

- Establish a transparent process for monitoring progress against management measures.
- Ensure that management measures are regularly reviewed and updated to reflect new information and developments, and to ensure the achievement of the goals of the NPOA.
- Encourage and facilitate research into affected seabird species and their interactions with fisheries.
- Encourage and facilitate research into new and innovative ways to reduce incidental catch.
- Provide mechanisms to enable all interested parties to be involved in the reduction of incidental catch.
- Promote education and awareness programmes to ensure that all fishers are aware of the need to reduce incidental catch and the measures available to achieve a reduction.

The 2004 NPOA-seabirds set out the mix of voluntary and mandatory measures that would be used to help reduce incidental captures of seabirds, noted research into the extent of the problem and the techniques for mitigating it, and outlined mechanisms to oversee, monitor and review the effectiveness of these measures. It was not within the scope of the NPOA to address threats to seabirds other than fishing. Such threats are identified in DOC's Action Plan for Seabird Conservation in New Zealand (Taylor 2000) and their management is undertaken by DOC.

Since publication of the NPOA in 2004, more progress has been made in the commercial fishing sector, including:

- in the deepwater fishing sector:
    - industry has implemented vessel specific risk management plans (VMPs) comprising non-mandatory seabird scaring devices, offal management, and other measures to reduce risks to seabirds,
    - the government has implemented mandatory measures to reduce risk to seabirds (e.g., use and deployment of seabird scaring devices), and
    - industry has taken a proactive stance in resourcing a 24/7 liaison officer to undertake incident response actions, mentoring, VMP and regime development and reviewing, and fleet-wide training;
- in the bottom- and surface-longline sectors, the government has implemented mandatory measures including tori lines, night setting, line weighting and offal management;
- a number of research projects have been or are currently being undertaken by government and industry into offal discharge, efficacy of seabird scaring devices, line weighting and longline setting devices; and
- workshops organised by both industry bodies and Southern Seabird Solutions are being held for the inshore trawl and longline sectors.

Mitigation has developed substantially since FAO's IPOA was published and a number of recent reviews consider the effectiveness of different methods (Bull 2007, 2009) and summarise currently accepted best practice (ACAP 2011). In December 2010, FAO held a Technical Consultation where International Guidelines on bycatch management and reduction of discards were adopted (FAO 2010). The text included an agreement that the guidelines should complement appropriate bycatch measures addressed in the IPOA-seabirds and its Best Practice Technical Guidelines (FAO 2009). The Guidelines were subsequently adopted by FAO in January 2011.

In 2013 the Ministry for Primary Industries released a revised and updated version of the NPOA-seabirds. This revision seeks to address recommendations from the IPOA/NPOA Seabirds Best Practice Technical Guidelines (FAO 2009). The scope of the revised New Zealand NPOA-seabirds 2013 is as follows:

- all seabird species absolutely or partially protected under the New Zealand Wildlife Act 1953;
- commercial, recreational and customary non-commercial fisheries in waters under New Zealand fisheries jurisdiction;
- all fishing methods that capture seabirds, including longlining, trawling, set netting, hand lining, trolling, purse seining and potting;

- all waters under New Zealand fisheries jurisdiction;
- high seas fisheries in which New Zealand flagged vessels participate, and, as appropriate and relevant, where foreign flagged vessels catch New Zealand seabirds; and
- other areas in which New Zealand seabirds are caught.

The long term objective of the 2013 NPOA-seabirds is: *'New Zealand seabirds thrive without pressure from fishing related mortalities, New Zealand fishers avoid or mitigate against seabird captures and New Zealand fisheries are globally recognised as seabird friendly.'*

The high-level subsidiary objectives of the NPOA-seabirds 2013 are:

i. Practical objective: All New Zealand fishers implement current best practice mitigation measures relevant to their fishery and aim through continuous improvement to reduce and where practicable eliminate the incidental mortality of seabirds.
ii. Biological risk objective: Incidental mortality of seabirds in New Zealand fisheries is at or below a level that allows for the maintenance at a favourable conservation status or recovery to a more favourable conservation status for all New Zealand seabird populations.
iii. Research and development objectives:
    a. the testing and refinement of existing mitigation measures and the development of new mitigation measures results in more practical and effective mitigation options that fishers readily employ;
    b. research and development of new observation and monitoring methods results in improved cost effective assurance that mitigation methods are being deployed effectively; and
    c. research outputs relating to seabird biology, demography and ecology provide a robust basis for understanding and mitigating seabird incidental mortality.
iv. International objective: In areas beyond the waters under New Zealand jurisdiction, fishing fleets that overlap with New Zealand breeding seabirds use internationally accepted current best practice mitigation measures relevant to their fishery.

Areas identified in the NPOA-seabirds 2013 that clearly require additional progress include:

i. mitigation measures for, and education, training and outreach in commercial set-net fisheries and inshore trawl fisheries;
ii. implementation of spatially and temporally representative at-sea data collection in inshore and some Highly Migratory Species (HMS) fisheries;
iii. mitigation measures for net captures for deepwater trawl fisheries;
iv. the extent of any cryptic mortality (seabird interactions that result in mortality but are unobserved or unobservable); and
v. mitigation measures for, education, training and outreach in, and risk assessment of non-commercial fisheries (in particular the set-net and hook and line fisheries).

The most important factor influencing contacts between seabirds and trawl warp cables is the discharge of offal (Wienecke & Robertson 2002; Sullivan et al. 2006b, ACAP 2011). Offal management methods used to reduce the attraction of seabirds to vessels include mealing, mincing and batching. ACAP recommends (ACAP 2011) full retention of all waste material where practicable because this significantly reduces the number of seabirds feeding behind vessels compared with the discharge of unprocessed fish waste (Wienecke & Robertson 2002, Abraham 2009, Favero et al. 2010) or minced waste (Melvin et al. 2010). Offal management has been found to be a key driver of seabird bycatch in New Zealand trawl fisheries (Abraham 2007, 2010b, Abraham & Thompson 2009b, Abraham et al. 2009, Pierre et al. 2010, 2012a, 2012b). Other best practice recommendations (ACAP 2011) are the use of bird scaring lines to deter birds from foraging near the trawl warps, use of snatch blocks to reduce the aerial extent of trawl warps, cleaning fish and benthic material from nets before shooting, minimising the time the trawl net is on the surface during hauling, and binding of large meshes in pelagic trawl before shooting.

In New Zealand, the three legally permitted devices used for mitigation by trawlers are tori lines (e.g., Sullivan et al. 2006a), bird bafflers (Crysel 2002), and warp scarers (Carey

2005). Middleton & Abraham (2007) reported experimental trials of mitigation devices designed to reduce the frequency of collisions between seabirds and trawl warps on 18 observed vessels in the squid trawl fishery in 2006. The frequencies of birds striking either warps or one of three mitigation devices (tori lines, 4-boom bird bafflers, and warp scarers) were assessed using standardised protocols during commercial fishing. Different warp strike mitigation treatments were used on different tows according to a randomised experimental design. Middleton & Abraham (2007) confirmed that the discharge of offal was the main factor influencing seabird strikes; almost no strikes were recorded when there was no discharge, and strike rates were low when only sump water was discharged (see also Abraham et al. 2009). In addition to this effect, tori lines were shown to be most effective mitigation approach and reduced warp strikes by 80–95% of their frequency without mitigation. Other mitigation approaches were only 10–65% effective. Seabirds struck tori lines about as frequently as they did the trawl warps in the absence of mitigation, but the consequences are unknown.

Recommended best practice for surface (pelagic) longline fisheries and bottom (demersal) longlines (ACAP 2011) includes weighting of lines to ensure rapid sinking of baits (including integrated weighted line for bottom longlines), setting lines at night when most vulnerable birds are less active, and the proper deployment of bird scaring lines (tori lines) over baits being set, and offal management (especially for bottom longlines). A range of other measures are offered for consideration.

In 2016, ACAP revised its best practice recommendations for surface-longline fishing, to modify the line weighting configurations and add approved hook shielding devices as stand-alone measures (ACAP 2016).

A review of the implementation of the 2013 NPOA-seabirds was scheduled to occur after four years. MPI commenced this review in April 2017 with significant input from a multi-stakeholder Seabird Advisory Group (SAG) established under the NPOA-seabirds and administered by MPI. The SAG is reflective of the multi-sector interests in seabirds and those that were involved in the development of the NPOA-seabirds, including the Crown's Treaty partner in fisheries matters, commercial industry, recreational sector, NGOs and the relevant government departments (DOC, MPI and MFAT).

The review examines the extent to which the five-year objectives of the NPOA-seabirds have been achieved and identifies key actions as priority for the current/next NPOA-seabirds. The review will also consider whether longer-term objectives need modification, and the effectiveness of the NPOA-seabirds implementation processes.

A new NPOA-seabirds is expected to be released in 2018. The current NPOA-seabirds will remain in effect until the new NPOA-seabirds is in effect.

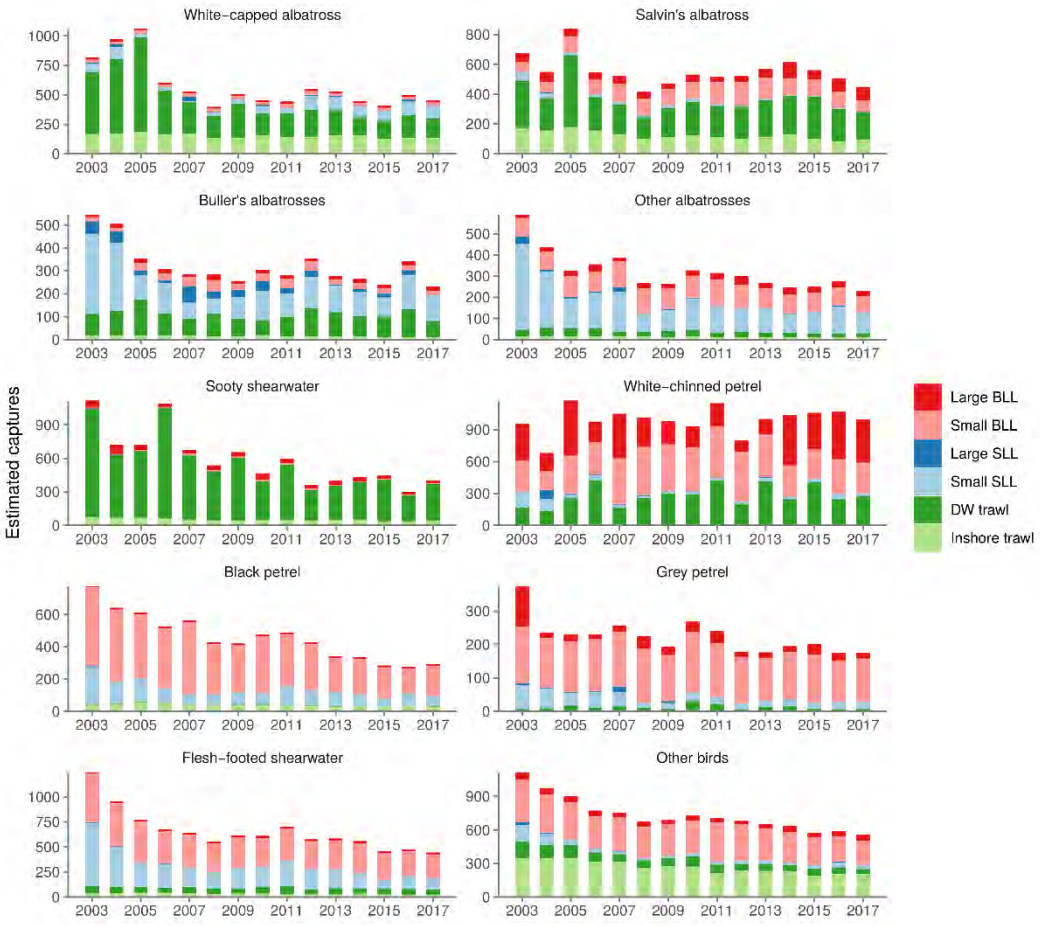
8.5 INDICATORS AND TRENDS

Population size	Multiple species and populations: see Taylor (2000)
Population trend	Multiple species and populations: see Taylor (2000)
Threat status	Multiple species and populations: see Robertson et al. (2017)
Number of interactions[1]	In the 2016–17 October fishing year, there were an estimated 4186 seabird captures (excluding cryptic mortalities) across all trawl and longline fisheries (http://data.dragonfly.co.nz/psc, data version v2016001). About 42% of the estimated captures across these fisheries (other fisheries such as set net are excluded) were in trawl fisheries, 14% in surface-longline fisheries (there are no large-SLL vessels operating in New Zealand's EEZ), and 44% in bottom-longline fisheries:

Bird group	DW Trawl	Inshore Trawl	Large BLL	Small BLL	Large SLL	Small SLL	All these methods
White-capped albatross	168	141	2	30	0	105	446
Salvin's albatross	184	103	79	75	0	4	445
Buller's albatross	69	15	10	19	0	114	227
Other albatrosses	19	9	19	78	0	97	222
Sooty shearwater	326	51	12	6	0	1	396
White-chinned petrel	272	11	389	293	0	26	991
Black petrel	1	35	0	188	0	64	288
Grey petrel	7	3	12	129	0	23	174
Flesh footed shearwater	56	30	3	239	0	108	436
Other birds	43	213	40	222	0	37	555
All birds combined	1 145	361	567	1279	0	579	3 931

[1] For more information, see: http://data.dragonfly.co.nz/psc.

Trends in interactions	Captures of all birds combined show a decreasing trend between 2002–03 and 2016–17 (http://data.dragonfly.co.nz/psc, data version v2018001) but there are substantial differences in trends between species and fisheries. Captures of several species have decreased following high total estimates in the early portion of the period, including white-capped albatross, Buller's albatross, sooty shearwater, black petrel and flesh-footed shearwater:



Capture rate trends (excluding cryptic mortalities) are described for the four fisheries estimated to account for most captures of a species (usually accounting for 70–80% of the total). For white-capped albatross, captures rates in the trawl fisheries for squid declined after 2004–05 but showed no trend for inshore and middle depth trawlers, and increasing capture rates to a peak in 2015–16 for surface longliners targeting southern bluefin tuna. For Salvin's albatross, captures rates have fluctuated without trend or increased in all fisheries taking substantial numbers of this species. Capture rates were unusually high in the hoki trawl fishery in 2004–05. For Buller's albatross, estimated capture rates fluctuated in bigeye and southern bluefin tuna target fisheries. The squid fishery shows some signs of an increasing trend, although had three relatively low capture rates in years 2012–13, 2014-15 and 2016–17 with a peak in 2015–16. The hoki trawl fishery in recent years has had higher capture rates. For white-chinned petrel, captures rates increased in squid trawlers but showed little trend for bottom longliners targeting ling, bluenose and hapuka. For sooty shearwaters, captures rates fluctuated without apparent trend in squid, middle-depth, hoki and inshore trawlers.

8.6 REFERENCES

Abraham, E R (2007) Summary of data collected during the southern squid fishery mincing trial. Research report prepared by Dragonfly for the Department of Conservation, Wellington, New Zealand. 39 p.

Abraham, E R (2009) Batching of waste to reduce seabird numbers behind trawl vessels. Unpublished report prepared for the Department of Conservation, Wellington, New Zealand.

Abraham, E R (2010a) Warp strike in New Zealand trawl fisheries, 2004–05 to 2008–00. *New Zealand Aquatic Environment and Biodiversity Report No. 60*. 29 p.

Abraham, E R (2010b) Mincing offal to reduce the attendance of seabirds at trawlers. Report prepared by Dragonfly for Department of Conservation, Wellington, New Zealand. 28 p.

Abraham, E R; Berkenbusch, K N; Richard, Y (2010a) The capture of seabirds and marine mammals in New Zealand non-commercial fisheries *New Zealand Aquatic Environment and Biodiversity Report No. 64*. 52 p.

Abraham, E R; Berkenbusch, K; Richard, Y; Thompson, F (2016) Summary of the capture of seabirds, mammals, and turtles in New Zealand commercial fisheries, 2002–03 to 2012–13. *New Zealand Aquatic Environment and Biodiversity Report No. 169*. 205 p.

Abraham, E R; Pierre, J P; Middleton, D A J; Cleal, J; Walker, N A; Waugh, S M (2009) Effectiveness of fish waste management strategies in reducing seabird attendance at a trawl vessel. *Fisheries Research 95*: 210–219.

Abraham, E R; Thompson, F N (2009a) Capture of protected species in New Zealand trawl and longline fisheries, 1998–99 to 2006–07. *New Zealand Aquatic Environment and Biodiversity Report No. 32*. 197 p.

Abraham, E R; Thompson, F N (2009b) Warp strike in New Zealand trawl fisheries, 2004–05 to 2006–07. *New Zealand Aquatic Environment and Biodiversity Report No. 33*. 21 p.

Abraham, E R; Thompson, F N (2010) Estimated capture of seabirds in New Zealand trawl and longline fisheries, 2002–03 to 2006–07. Final Research Report for research project PRO2007-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Abraham, E R; Thompson, F N (2011a) Estimated capture of seabirds in New Zealand trawl and longline fisheries, 2002–03 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 79*. 74 p.

Abraham, E R; Thompson, F N (2011b) Summary of the capture of seabirds, marine mammals, and turtles in New Zealand commercial fisheries, 1998–99 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 80*. 170 p.

Abraham, E R; Thompson, F N; Berkenbusch, K (2013) Estimated capture of seabirds in New Zealand trawl and longline fisheries, 2002–03 to 2010–11. Final Research Report for research project PRO2010-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Abraham, E R; Thompson, F N; Oliver, M D (2010b) Summary of the capture of seabirds, marine mammals and turtles in New Zealand commercial fisheries, 1998–99 to 2007–08. *New Zealand Aquatic Environment and Biodiversity Report No. 45*. 148 p.

ACAP (2010) ACAP species assessment. Retrieved from http://www.acap.aq.

ACAP (2011) Report of the Fourth Meeting of the Seabird Bycatch Working Group, Guayaquil, Ecuador, 22–24 August 2011. 94 p. AC6 Doc 14 Rev4.

ACAP (2012) ACAP Conservation Priorities. Fourth Meeting of the Parties, Lima, Peru, 23–27 April 2012. 13 p. Retrieved from http://www.acap.aq/docman/english/meeting-of-the-parties/mop4/mop4-final-report.

ACAP (2016) Report or the Ninth Meeting of the Advisory Committee, La Serena, Chile, 9–13 May 2016. Retrieved from http://acap.aq/en/advisory-committee/ac9/2845-ac9-report/file.

Amey, J; Sagar, P (2013) Salvin's albatross population trend at the Bounty Islands, 1997-2011. Report prepared for PRO2012-06 for the Department of Conservation, Wellington. 31 p.

Arcos, J M; Oro, D (1996) Changes in foraging range of Audouin's gulls Larus audouinii in relation to a trawler moratorium in the western Mediterranean. *Waterbirds 19*: 128–131.

Astles, K L; Holloway, M G; Steffe, A; Green, M; Ganassin, C; Gibbs, P J (2006) An ecological method for qualitative risk assessment and its use in the management of fisheries in New South Wales, Australia. *Fisheries Research 82*: 290–303.

Ayers, D; Francis, M P; Griggs, L H; Baird, S J (2004) Fish bycatch in New Zealand tuna longline fisheries, 2000/01 and 2001/02. *New Zealand Fisheries Assessment Report 2004/46*. 47 p.

Baird, S J (1994) Nonfish Species and Fisheries Interactions Working Group Report. *New Zealand Fisheries Assessment Working Group Report 94/1*. Ministry of Agriculture and Fisheries, Wellington. 54 p.

Baird, S J (1995) Nonfish Species and Fisheries Interactions Working Group Report — April 1995. *New Zealand Fisheries Assessment Working Group Report 95/1*. Ministry of Agriculture and Fisheries, Wellington. 24 p.

Baird, S J (1996) Nonfish Species and Fisheries Interactions Working Group Report — May 1996. *New Zealand Fisheries Assessment Working Group Report 96/1*. Ministry of Fisheries, Wellington. 34 p.

Baird, S J (1997) Report on the incidental capture of nonfish species during fishing operations in New Zealand waters. Report completed as part of the Ministry of Fisheries SANF01 contract. 15 p plus appendices on New Zealand fur seal-trawl fishery interaction (54 p) and seabird-tuna longline fishery interaction (34 p). (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baird, S J (1998) Estimation of nonfish bycatch in commercial fisheries in New Zealand waters, 1990–91 to 1993–94. Final Research Report for Ministry of Fisheries research Project ENV9701. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baird, S J (1999) Estimation of nonfish bycatch in commercial fisheries in New Zealand waters, 1997–98. Report completed for Ministry of Fisheries project ENV9801 Objective 1. 57 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baird, S J (2000) Incidental capture of seabirds in New Zealand tuna longline fisheries. *In*: Flint, E; Swift, K (Eds.), Second Albatross Conference on the Biology and Conservation of Albatrosses and other Petrels, Honolulu, Hawaii, USA, 8–12 May 2000. p 126. Abstracts and Oral Presentations. *Marine Ornithology 28*: 125–152.

Baird, S J (2001a) Estimation of the incidental capture of seabird and marine mammal species in commercial fisheries in New Zealand waters, 1998–99. *New Zealand Fisheries Assessment Report 2001/14*. 43 p.

Baird, S J (Comp. & Ed.) (2001b) Report on the International Fishers' Forum on Solving the Incidental Capture of Seabirds in Longline Fisheries, Auckland, New Zealand, 6–9 November 2000. Department of Conservation, Wellington. 63 p.

Baird, S J (2003) New Zealand breeding seabirds: human-induced mortality — a review. Draft report prepared for the Ministry of Fisheries project ENV2000/09. 74 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baird, S J (2004a) Estimation of the incidental capture of seabird and marine mammal species in commercial fisheries in New Zealand waters, 1999–2000. *New Zealand Fisheries Assessment Report 2004/41*. 56 p.

Baird, S J (2004b) Incidental capture of seabird species in commercial fisheries in New Zealand waters, 2000–01. *New Zealand Fisheries Assessment Report 2004/58*. 63 p.

Baird, S J (2004c) Incidental capture of seabird species in commercial fisheries in New Zealand waters, 2001–02. *New Zealand Fisheries Assessment Report 2004/60*. 51 p.

Baird, S J (2005) Incidental capture of seabird species in commercial fisheries in New Zealand waters, 2002–03. *New Zealand Fisheries Assessment Report 2005/2*. 50 p.

Baird, S J; Francis, M; Griggs, L H; Dean, H A (1998) Annual review of bycatch in southern bluefin and related tuna longline fisheries in the New Zealand 200 n. mile Exclusive Economic Zone. CCSBT-ERS/9806/31. Third Meeting of the Ecologically Related Species Working Group, Tokyo, 9–13 June 1998.

Baird, S J; Gilbert, D J (2010) Initial assessment of risk posed by commercial fisheries to seabirds breeding in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 50*. 99 p.

Baird, S J; Gilbert, D J; Smith, M H (2006) Review of environmental risk assessment methodologies with relevance to seabirds and fisheries within New Zealand waters. Final Research Report for Ministry of Fisheries project ENV2005/01 Objective 3. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baird, S J; Griggs, L H (2004) Estimation of within-season chartered southern bluefin tuna (*Thunnus maccoyii*) longline seabird incidental captures, 2002. *New Zealand Fisheries Assessment Report 2004/42*. 15 p.

Baird, S J; Sanders, B M; Dean, H A; Griggs, L H (1999) Estimation of nonfish bycatch in commercial fisheries in New Zealand waters, 1990/91 to 1993/94. Final Research Report for Ministry of Fisheries project ENV9701 Objective 1. 63 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baird, S J; Smith, M H (2008) Incidental capture of seabird species in commercial fisheries in New Zealand waters, 2005–06. *New Zealand Aquatic Environment and Biodiversity Report No. 18*. 124 p.

Baker, B; Cunningham, R; Hedley, G; King, S (2008a) Data collection of demographic, distributional and trophic information on the Westland petrel to allow estimation of effects of fishing on population viability. Final Research Report prepared for the Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, B; Cunningham, R; Hedley, G; King, S (2011a) Data collection of demographic, distributional and trophic information on the Westland petrel to allow estimation of effects of fishing on population viability. Final Research Report prepared for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, B; Double, M (2007) Data collection of demographic, distributional and trophic information on selected seabird species to allow estimation of effects of fishing on population viability. Final Research Report prepared for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, G B; Double, M C; Gales, R; Tuck, G N; Abbott, C L; Ryan, P G; Petersen, S L; Robertson, C J R; Alderman, R (2007a) A global assessment of the impact of fisheries related mortality on shy and white-capped albatrosses: conservation implications. *Biological Conservation 137*: 319–333.

Baker, B; Hedley, G; Cunningham, R (2010a) Data collection of demographic, distributional and trophic information on the

flesh-footed shearwater to allow estimation of effects of fishing on population viability: 2009–10 Field Season. Final Research Report prepared for Ministry for Primary Industries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington).

Baker, G B; Hedley, G; Cunningham, R (2011) Data collection of demographic, distributional and trophic information on the flesh-footed shearwater to allow estimation of effects of fishing on population viability. 2010/11 Field Season. Final Research Report prepared for the Ministry for Primary Industries project PRO2006-01.

Baker, B; Hedley, G; Waugh, S; Cunningham, R (2008b) Data collection of demographic, distributional and trophic information on selected seabird species to allow estimation of effects of fishing on population viability. Final Research Report prepared for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, B; Hedley, G; Waugh, S; Cunningham, R (2009a) Data collection of demographic, distributional and trophic information on the flesh-footed shearwater to allow estimation of effects of fishing on population viability: 2008–09 Field Season. Final Research Report prepared for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, G B; Jensz, K (2016) White-capped albatross aerial photographic survey 2016. Report prepared by Latitude 42 for the Department of Conservation, Wellington, New Zealand. 3 p.

Baker, G B; Jensz, K; Bell, M; Fretwell, P T; Phillips, R A (2017) Seabird Population Research, Chatham Islands 2016/17 aerial photographic survey. Report prepared by Latitude 42 for the Department of Conservation, Wellington, New Zealand. 20 p.

Baker, G B; Jensz, K; Cunningham, R (2009b) Data collection of demographic, distributional and trophic information on the white-capped albatross to allow estimation of effects of fishing on population viability – 2008 field season. Final Research Report for Ministry of Fisheries project PRO2006-01H. 13 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, G B; Jensz, K; Cunningham, R (2010b) Data collection of demographic, distributional and trophic information on the white-capped albatross to allow estimation of effects of fishing on population viability – 2009 field season. Final Research Report for Ministry of Fisheries project PRO2006-01I. 13 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, G B; Jensz, K; Cunningham, R (2011b) Data collection of demographic, distributional and trophic information on the white-capped albatross to allow estimation of effects of fishing on population viability – 2010 Field Season. Final Research Report for Ministry of Fisheries project PRO2006-01J. 15 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, G B; Jensz, K; Cunningham, R (2013) White-capped albatross population estimate – 2011/12 and 2012/13. Final Report for Department of Conservation Contract 4431 & Project POP2012-05. 22 p.

Baker, B; Jensz, K; Cunningham, R (2014b) White-capped albatross aerial survey 2014. Final Research Report for the Department of Conservation Contract 4423/4524. (Unpublished report held by the Department of Conservation, Wellington.) Retrieved from http://www.doc.govt.nz/Documents/conservation/marine-and-coastal/marine-conservation-services/reports/pop2013-02-white-capped-albatross-survey-2014-final-report.pdf.

Baker, G B; Jensz, K; Cunningham, R; Holdsworth, M; Chilvers, B L (2015a) White-capped albatross aerial survey 2015. Report prepared by Latitude 42 for the Department of Conservation, Wellington, New Zealand. 31 p.

Baker, G B; Jensz, K; Double, M; Cunningham, R (2007b) Data collection of demographic, distributional and trophic information on selected seabird species to allow estimation of effects of fishing on population viability. Final Research Report for Ministry of Fisheries project PRO2006-01. 12 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, G B; Jensz, K; Double, M; Cunningham, R (2008c) Data collection of demographic, distributional and trophic information on selected seabird species to allow estimation of effects of fishing on population viability. Final Research Report for Ministry of Fisheries project PRO2006-01. 11 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, G B; Jensz, K; Sagar, P (2012) Data collection of demographic, distributional and trophic information on Salvin's albatross to allow estimation of effects of fishing on population viability. Final Research Report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Baker, G B; Jensz, K; Sagar, P (2014a) 2013 Aerial survey of Salvin's albatross at the Bounty Islands. Final Report for the Department of Conservation Contract 4521.

Baker, B G; Jensz, K; Sagar, P (2015b) 2014 Aerial survey of Salvin's albatross at The Snares, Western Chain. Report prepared by Latitude 42 for the Department of Conservation, Wellington, New Zealand. 9 p.

Barbraud, C; Booth, A; Taylor, G A; Waugh, S M (2014) Survivorship in Flesh-footed Shearwater *Puffinus carneipes* at two sites in northern New Zealand. *Marine Ornithology 42*: 91–97

Bartle, J A (1991) Incidental capture of seabirds in the New Zealand subantarctic squid trawl fishery, 1990. *Bird Conservation International 1*: 351–359.

Bartumeus, F; Giuggioli, L; Louzao, M; Bretagnolle, V; Oro, D; Levin, S A (2010). Fishery discards impact on seabird movement patterns at regional scales. *Current Biology 20*: 215–222.

Bell, E A (2002) (cited in Sommer et al. 2010): Grey petrels (*Procellaria cinerea*) on Antipodes Island, New Zealand: research feasibility, April to June 2001. *Department of Conservation Science Internal Series 60*. Department of Conservation, Wellington. 31 p.

Bell, M D; Bell, D J; Boyle, D P (2017) Chatham Island Mollymawk research on Te Tara Koi Koia: November 2016. Technical report to the Department of Conservation.

Bell, E A; Mischler, C P; MacArthur, N; Sim, J L (2016) Black petrels (*Procellaria parkinsoni*) population study on Hauturu-o-Toi/Little Barrier Island, 2015/16. (Unpublished report prepared for the Department of Conservation).

Bell, E A; Sim, J L; Scofield, P; Francis, R I C C (2012) Population parameters of the black petrels (*Procellaria parkinsoni*) on Great Barrier Island (Aotea Island), 2009/10. Report for Conservation Services Programme Project POP2009-01, near-final (post-review) draft. 82 p. Department of Conservation, Wellington.

Bell, E A; Sim, J L; Scofield, P; Francis, R I C C; Landers, T (2013) At-sea distribution and population parameters of the black petrels (*Procellaria parkinsoni*) on Great Barrier Island (Aotea Island), 2012/13. Draft report on project POP2012/03 for DOC-CSP working group, August 2013. Department of Conservation, Wellington.

Bell, E A; Burgin, D; Sim, J; Dunleavy, K; Fleishman, A; Scofield, R P (2018) Population trends, breeding distribution and habitat use of black petrels (*Procellaria parkinsoni*) – 2016/2017 operational report. *New Zealand Aquatic Environment and Biodiversity Report No. 198.*

Brothers, N P (1991) Albatross mortality and associated bait loss in the Japanese longline fishery in the Southern Ocean. *Biological Conservation 55*: 255–268.

Brothers, N P; Cooper, J; Lokkeborg, S (1999) The incidental catch of seabirds by longline fisheries: worldwide review and technical guidelines for mitigation. *FAO Fisheries Circular No. 937*. 107 p.

Brothers, N P; Duckworth, A R; Safina, C; Gilman, E L (2010) Seabird bycatch in pelagic longline fisheries is grossly underestimated when using only haul data. *PLoS ONE 5*: e12491.

Bull, L S (2007) A review of methodologies for mitigating incidental catch of seabirds in New Zealand Fisheries. DOC Research & Development Series 263. Department of Conservation, Wellington. 57 p.

Bull, L S (2009) New mitigation measures reducing seabird by-catch in trawl fisheries. *Fish and Fisheries 8*: 31–56.

Campbell, J L; Gallagher, C (2007) Assessing the relative effects of fishing on the New Zealand marine environment through risk analysis. *ICES Journal of Marine Science 64*: 256–270.

Carey, C (2005) Carefree's cunning contraption. *Seafood New Zealand 13 (6)*: 44–45.

Carroll, J; Charteris, M; Sagar P (2010) Population assessment of Salvin's albatrosses at the Snares Western Chain, 29 September – 14 October 2009. Final research report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Charteris, M; Carroll, J; Sagar P (2009) Population assessment of Salvin's albatrosses at the Snares Western Chain, 29 September – 17 October 2008. Final research report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Croxall, J P (2008) The role of science and advocacy in the conservation of Southern Ocean albatrosses at sea. *Bird Conservation International 18*: S13–29.

Croxall, J P; Butchart, S H M; Lascelles, B; Stattersfield, A J; Sullivan, B; Symes, A; Taylor, P (2012) Seabird conservation status, threats and priority actions: a global assessment. *Bird Conservation International 22*: 1–34.

Croxall, J P; Prince, P A; Rothery, P; Wood, A G (1998) Population changes in albatrosses at South Georgia. pp 68–83. *In*: Robertson, G; Gales, R (Eds.), Albatross biology and conservation. Surrey Beatty & Sons, Chipping Norton, Australia.

Crysel, S (2002) Baffling birds brings benefits. *Seafood New Zealand 10 (10)*: 60–61.

Darby, J T; Dawson, S M (2000) Bycatch of yellow-eyed penguin (*Megadyptes antipodes*) in gillnets in New Zealand waters 1979–1997. *Biological Conservation 93*: 327–332.

Debski, I; Hjörvarsdóttir, F (2017) Salvin's albatross Bounty Islands: methodology development. Report of Workshop, 28 November 2016.

Dillingham, P W; Fletcher, D (2011) Potential biological removal of albatrosses and petrels with minimal demographic information. *Biological Conservation 144*: 1885–1894.

Edwards, C T T; Roberts, J O; Walker, K; Elliot, G (2017) Quantitative modelling of Antipodean wandering albatross. *New Zealand Aquatic Environment and Biodiversity Report No. 180*. 32 p.

Elliott, G; Walker, K (2014) Gibson's wandering albatross research, Adams Island 2014. Unpublished report prepared for the Department of Conservation, Wellington, New Zealand. Retrieved from http://www.doc.govt.nz/Documents/conservation/marine-and-coastal/marine-conservation-services/meetings/pop2013-03-gibsons-albatross-population-study-draft-final-report.pdf.

FAO (1999) International Plan of Action for reducing incidental catch of seabirds in longline fisheries. International Plan of Action for the conservation and management of sharks. International Plan of Action for the management of fishing capacity. Food & Agriculture Organisation of the United Nations, Rome. 26 p.

FAO (2009) Fishing operations. 2. Best practices to reduce incidental catch of seabirds in capture fisheries. FAO Technical Guidelines for

Responsible Fisheries. No. 1, Suppl. 2. Food & Agriculture Organisation of the United Nations, Rome. 49 p.

FAO (2010) Report of the Technical Consultation to Develop International Guidelines on Bycatch Management and Reduction of Discards, Rome, 6–10 December 2010. *FAO Fisheries and Aquaculture Report No. 957*. 41 p.

Favero, M; Blanco, G; Garcia, G; Copello, S; Seco Pon, J P; Frere, E; Quintana, F; Yorio, P; Rabuffetti, F; Canete, G; Gandini, P (2010) Seabird mortality associated with ice trawlers in the Patagonian shelf: effect of discards on the occurrence of interactions with fishing gear. *Animal Conservation 14*: 131–139.

Filippi, D; Waugh, S M; Nicol, S (2010) Revised spatial risk indicators for seabird interactions with longline fisheries in the western and central Pacific. Western and Central Pacific Fisheries Commission, Scientific Committee (2010) Paper EB-IP-01.

Fletcher, D; MacKenzie, D; Dillingham, P (2008) Modelling of impacts of fishing-related mortality on New Zealand seabird populations. Final Research Report for Ministry of Fisheries project ENV2005-08. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Fletcher, W J (2005) The application of qualitative risk assessment methodology to prioritize issues for fisheries management. *ICES Journal of Marine Science 62*: 1576–1587.

Francis, R I C C (2012) Fisheries risks to the population viability of white-capped albatross *Thalassarche steadi*. *New Zealand Aquatic Environment and Biodiversity Report No. 104*. 24 p.

Francis, R I C C; Bell, E A (2010) Fisheries risks to population viability of black petrel (Procellaria parkinsoni). New Zealand Aquatic Environment and Biodiversity Report No. 51. 57 p.

Francis, R I C C; Elliott, G; Walker, K (2015) Fisheries risks to the viability of Gibson's wandering albatross Diomedea gibsoni. New Zealand Aquatic Environment and Biodiversity Report No. 152. 48 p.

Francis, R I C C; Sagar, P M (2012) Modelling the effect of fishing on southern Buller's albatross using a 60-year dataset. New Zealand Journal of Zoology 39: 1, 3–17.

Francis, R I C C; Sagar, P M; Fu, D (2008) Status of the Snares Islands population of southern Buller's albatross. Final Research Report for Year 2 of Ministry of Fisheries research project PRO200602. 138 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Fraser, M J; Bell, M; Scofield, P; Robertson, C J R (2009a) Population assessment of Northern Buller's Albatross, Northern Royal Albatross, and Northern Giant Petrels at the Forty-Fours, 09–18 November 2008. Research Progress Report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Fraser, M J; Cameron, N; Scofield, P; Robertson, C J R (2010a) Population assessment of Northern Buller's Albatross and Northern Giant Petrels at the Forty-Fours, Chatham Islands, 1–8 December 2009. Final Research Report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Fraser, M J; Henderson, G; Robertson, C J R; Scofield, P (2011) Population dynamics of the Chatham Mollymawk at The Pyramid, 19 November–2 December 2010. Final Research Report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Fraser, M J; Hunt, S; Scofield, P; Robertson, C J R (2009b) Population Dynamics of the Chatham Mollymawk at The Pyramid, 22 November to 07 December 2008. Final Research Report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Fraser, M J; Palmer, D; Deppe, L; Scofield, P; Robertson, C J R (2010b) Population dynamics of the Chatham Mollymawk at The Pyramid, 20 November–14 December 2009. Final Research Report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Furness, R W (2003) Impacts of fisheries on seabird communities. Scientia Marina 67: 33–45.

Furness, R W; Ensor, K; Hudson, A V (1992) The use of fishery waste by gull populations around the British Isles. Ardea 80: 105–113.

Gilbert, D (2009) Calculating the population ratio of total seabirds to adults. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Grémillet, D; Pichegru, L; Kuntz, G; Woakes, A G; Wilkinson, S; Crawford, R J M; Ryan, P G (2008) A junk-food hypothesis for gannets feeding on fishery waste. Proceedings of the Royal Society of London B Biological Sciences 275: 1149–1156.

Hartill, B; Bian, R; Armiger, H; Vaughan, M; Rush, N (2007) Recreational marine harvest estimates of snapper, kahawai, and kingfish in QMA 1 in 2004–05. *New Zealand Fisheries Assessment Report 2007/26*. 44 p.

Hobday, A J; Smith, A; Webb, H; Daley, R; Wayte, S; Bulman, C; Dowdney, J; Williams, A; Sporcic, M; Dambacher, J; Fuller, M; Walker, T (2007) Ecological risk assessment for the effects of fishing: methodology. Report R04/1072 for the Australian Fisheries Management Authority, Canberra.

Kirby, J; Hobday, A (2007) Ecological risk assessment for the effects of fishing the western and central Pacific Ocean: productivity-susceptibility analysis. Paper presented to the Scientific Committee of the Third Regular Session of the Western and Central Pacific Fisheries Commission, WCPFC-SC3-EB SWG/WP-1.

Lalas, C (1991) Assessment of bird kills in set nets in Otago Harbour over a period of eight years (1977–1985). (Unpublished report held by Department of Conservation, Dunedin.)

Lewison, R L; Crowder, L B; Read, A J; Freeman, S A (2004) Understanding impacts of fisheries bycatch on marine megafauna. *Trends in Ecology and Evolution 19*: 598–604.

Lewison, R L; Oro, D; Godley, B J; Underhill, L; Bearhop, S; Wilson, R P; Ainley, D; Arcos, J M; Boersma, P D; Borboroglu, P G; Boulinier, T; Frederiksen, M; Genovart, M; González-Solís, J; Green, J A; Grémillet, D; Hamer, K C; Hilton, G M; Hyrenbach, K D; Martínez-Abraín, A; Montevecchi, W A; Phillips, R A; Ryan, P G; Sagar, P M; Sydeman, W J; Wanless, S; Watanuki, Y; Weimerskirch, H; Yorio, P (2012) Review: Research priorities for seabirds: improving conservation and management in the 21st century. *Endangered Species Research 17*: 93–121.

Lonergan, M (2011) Potential biological removal and other currently used management rules for marine mammal populations: a comparison. *Marine Policy 35*: 584–589.

Louzao, M; Arcos, J M; Guijarro, B; Valls, M; Oro, D (2011) Seabird-trawling interactions: factors affecting species-specific to regional community utilisation of fisheries waste. *Fisheries Oceanography 20*: 263–277.

Maunder, M N; Houston, D M; Dunn, A; Seddon, P J; Kendrick, T H (2007) Assessment to risk of yellow-eyed penguins *Megadyptes antipodes* from fisheries incidental mortality in New Zealand fisheries and definition of information requirements for managing fisheries related risk. Final Research Report for Ministry of Fisheries project ENV2005-13. 28 p. (Unpublished report held by the Ministry for Primary Industries, Wellington).

Melvin, E F; Dietrich, K S; Fitzgerald, S; Cordoza, T (2010) Reducing seabird strikes with trawl cables in the pollock catcher-processor fleet in the eastern Bering Sea. Agreement on the Conservation of Albatrosses and Petrels, SBWG-3 Doc 14 Rev1, Hobart, Australia. 18 p.

MFish-DOC (2004) National plan of action to reduce the incidental catch of seabirds in New Zealand fisheries. Ministry of Fisheries and Department of Conservation, Wellington. 62 p.

Middleton, D A J; Abraham, E R (2007) The efficacy of warp strike mitigation devices: trials in the 2006 squid fishery. Final Research Report for Ministry of Fisheries research project IPA2006/02. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Mischler, C P (2016) Conservation Services Programme, Flesh-footed Shearwater Project 4653, Demographic Component, April–May 2016 Report. Unpublished technical report to the Department of Conservation.

Miskelly, C M; Dowding, J E; Elliott, G P; Hitchmough, R A; Powlesland, R G; Robertson, H A; Sagar, P M; Scofield, R P; Taylor, G A (2008) Conservation status of New Zealand birds, 2008. *Notornis 55*: 117–135.

Moore, J E; Zydelis, R (2008) Quantifying seabird bycatch: where do we go from here? *Animal Conservation 11*: 257–259.

MPI (2013) National Plan of Action – 2013 to reduce the incidental catch of seabirds in New Zealand Fisheries. Ministry for Primary Industries. 63 p.

Newman, J; Scott, D; Bragg, C; McKechnie, S; Moller, H; Fletcher, D (2009) Estimating regional population size and annual harvest intensity of the sooty shearwater in New Zealand. *New Zealand Journal of Zoology 36*: 307–323.

Niel, C; Lebreton, J (2005) Using demographic invariants to detect overharvested bird populations from incomplete data. *Conservation Biology 19*: 826–835.

Oro, D; Cam, E; Pradel, R; Martinez-Abrain, A (2004) Influence of food availability on demography and local population dynamics in a long-live seabird. *Proceedings of the Royal Society of London B Biological Sciences 271*: 387–396.

Oro, D; Furness, R W (2002) Influences of food availability and predation on survival of kittiwakes. *Ecology 83*: 2516–2528.

Oro, D; Pradel, R; Lebreton, J D (1999) Food availability and nest predation influence life history traits in Audouin's gull, *Larus audouinii*. *Oecologia 118*: 438–445.

Oro, D; Ruiz, X; Jover, L; Pedrocchi, V; Gonzalez-Solis, J (1997) Audouin's gull diet and adult time budget responses on changes in food availability induced by commercial fisheries. *Ibis 139*: 631–637.

OSNZ (2010) Checklist of the birds of New Zealand and the Ross Dependency, Antarctica. Fourth Edition. Te Papa Press in association with the Ornithological Society of New Zealand Inc., Wellington.

Parker, G C; Rexer-Huber, K; Thompson, D (2015) Are grey petrels returning to Campbell Island? Survey and census 14 years after rodent eradication. Unpublished report to the Department of Conservation. Parker Conservation, Dunedin.

Parker, G C; Sagar, P; Thompson, D; Rexer-Huber, K (2016) White-capped albatross – adult survival & other demographic parameters, Auckland Islands 2017. Report to the Conservation Services Programme, Department of Conservation. Parker Conservation, Dunedin.

Petersen, S L; Honig, M B; Ryan, P G; Underhill, L G (2009) Seabird bycatch in the pelagic longline fishery off southern Africa. *African Journal of Marine Science 31*: 191–204.

Phillips, R A; Silk, J R D; Croxall, J P; Afanasyev, V (2006) Year-round distribution of white-chinned petrels from South Georgia: relationships with oceanography and fisheries. *Biological Conservation 129*: 336–347.

Pierre, J P; Abraham, E R; Cleal, J; Middleton, D A J (2012a) Reducing effects of trawl fishing on seabirds by limiting foraging opportunities provided by fishery waste. *Emu 112*: 244–254.

Pierre, J P; Abraham, E R; Middleton, D A J; Cleal, J; Bird, R; Walker, N A; Waugh, S M (2010) Reducing interactions between trawl fisheries and seabirds: responses to foraging patches provided

by fish waste batches. *Biological Conservation 143*: 2779–2788.

Pierre, J P; Abraham, E R; Richard, Y (2012b) Concluding six years of research on seabird bycatch reduction through modified discharge management regimes: Is batch discharge better than ad-hoc discharge from trawl vessels? SBWG-4 Doc 14, Progress report on Agreement on the Conservation of Albatrosses and Petrels, Project Application 2010-04.

Poncet, S; Robertson, G; Phillips, R A; Lawton, K; Phalan, B; Trathan, P N; Croxall, J P (2006) Status and distribution of wandering, black-browed and grey-headed albatrosses breeding at South Georgia. *Polar Biology 29*: 772–781.

Priddel, D; Carlile, N; Fullagar, P; Hutton, I; O'Neill, L (2006) Decline in the distribution and abundance of flesh-footed shearwaters (*Puffinus carneipes*) on Lord Howe Island, Australia. *Biological Conservation 128 (3)*: 412–424.

Ramm, K C (2011) Conservation Services Programme Observer Report for the period 1 July 2008 to 30 June 2009. Retrieved from www.doc.govt.nz/documents/science-and-technical/2008–09-csp-observer-report.pdf.

Ramm, K C (2012) Conservation Services Programme Observer Report for 1 July 2009 to 30 June 2010. Retrieved from www.doc.govt.nz/documents/conservation/marine-and-coastal/marine-conservation-services/csp-observer-report-2009–10.pdf.

Rexer-Huber, K; Parker, G C; Sagar, P; Thompson, D (2015) White-chinned petrel population estimate, Disappointment Island (Auckland Islands). Report prepared by Parker Conservation for the Agreement for the Conservation of Albatross and Petrels. 14 p.

Rice, J (2006) Impacts of Mobile Bottom Gears on Seafloor Habitats, Species, and Communities: A Review and Synthesis of Selected International Reviews. Canadian Science Advisory Secretariat Research Document 2006/057. 35 p. Retrieved from http://www.dfo-mpo.gc.ca/CSAS/Csas/DocREC/2006/RES2006_057_e.pdf.

Richard, Y; Abraham, E R (2013a) Application of Potential Biological Removal methods to seabird populations. *New Zealand Aquatic Environment and Biodiversity Report No. 108*. 30 p.

Richard, Y; Abraham, E R (2013b) Risk of commercial fisheries to New Zealand seabird populations. *New Zealand Aquatic Environment and Biodiversity Report No. 109*. 58 p.

Richard, Y; Abraham, E R (2015) Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006–07 to 2012–13. *New Zealand Aquatic Environment and Biodiversity Report. No. 162*. 85 p.

Richard, Y; Abraham, E R; Filippi, D (2011) Assessment of the risk to seabird populations from New Zealand commercial fisheries. Final Research Report for Ministry of Fisheries projects IPA2009-19 and IPA 2009-20. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Richard, Y; Pierre, J P; Abraham, E R (2014) Seasonality and temporal trends in counts of seabirds from pelagic tours off Kaikoura, New Zealand. (Unpublished report held by the Encounter Foundation, Kaikoura, New Zealand.)

Richard, Y; Abraham, E A; Berkenbusch, K (2017) Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006–07 to 2014–15. *New Zealand Aquatic Environment and Biodiversity Report 191*. 133 p.

Robertson, C J R (1998) (cited in Scofield 2011): Factors influencing the breeding performance of the Northern Royal Albatross. *In*: Robertson, G; Gales, R (Eds.), Albatross Biology and Conservation. pp 20–45. Surrey Beatty & Sons, Chipping Norton.

Robertson, C J R; Bell, E A; Sinclair, N; Bell, B D (2003) Distribution of seabirds from New Zealand that overlap with fisheries worldwide. *Science for Conservation 233*. Department of Conservation, Wellington. 102 p.

Robertson, H A; Baird, K; Dowding J E; Elliott, G P; Hitchmough, R A; Miskelly, C M; McArthur, N; O'Donnell, C F J; Sagar, P M; Scofield, R P; Taylor, G A (2017) Conservation status of New Zealand birds, 2016. *New Zealand Threat Classification Series 19*. Department of Conservation, Wellington. 23 p.

Romano, M D; Piatt, J F; Roby, D D (2006) Testing the junk-food hypothesis on marine birds: effects of prey type on growth and development. *Waterbirds 29*: 407–414.

Rowe, S J (2009) Conservation Services Programme Observer Report for the period 1 July 2004 to 30 June 2007. DOC Marine Conservation Services Series 1. Department of Conservation, Wellington. 97 p.

Rowe, S J (2010) Conservation Services Programme Observer Report for the period 1 July 2007 to 30 June 2008. DOC Marine Conservation Services Series 4. Department of Conservation, Wellington. 97 p.

Rowe, S J (2013) Level 1 risk assessment for incidental seabird mortality associated with fisheries in New Zealand's Exclusive Economic Zone. DOC Marine Conservation Services Series 10. Department of Conservation, Wellington. 58 p.

Sagar, P (2014) Population studies of southern Buller's albatrosses on The Snares – population study of Buller's albatrosses. Report prepared for Department of Conservation, Ministry for Primary Industries, and Deepwater Group Limited.

Sagar, P M; Carroll, J; Charteris, M; Thompson, D; Scofield, P (2011) Population assessment of Salvin's albatrosses at the Snares Western Chain, 25 September – 14 October 2010. Final Research Report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Sagar, P M; Stahl, J C (2005) Increases in the numbers of breeding pairs in two populations of Buller's albatross (*Thalassarche bulleri bulleri*). *Emu 105*: 49–55.

Sagar, P M; Torres, L; Thompson, D (2010) Demography and distribution of Buller's Albatrosses *Thalassarche bulleri bulleri*: Final research report of the 2010 field season. NIWA Client Report CHC2010-156, prepared for Department of Conservation.

Scofield, P (2011) Aerial photography of Northern Royal Albatrosses at the Chatham Islands, 2006–2010. Final Research Report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington).

Scofield, P; Fraser, M J; Robertson, C J R (2008a) Population Dynamics of the Chatham Mollymawk at The Pyramid, 19–29 November 2007. Final Research Report for Ministry of Fisheries project PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington).

Scofield, P; Fraser, M J; Robertson, C J R (2008b) Population assessment of Northern Buller's Albatross, Northern Royal Albatross, and Northern Giant Petrels at the Forty-Fours, 13–19 November 2007. Final Research Report for Ministry of Fisheries project, PRO2006-01. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Sharp, B (2009) A risk assessment framework for incidental seabird mortality associated with New Zealand fishing in the New Zealand EEZ. (Unpublished report held by the Ministry for Primary Industries.)

Sharp, B; Parker, S; Smith, N (2009) An impact assessment framework for bottom fishing methods in the CCAMLR area. CAMMLR Science, 15 June 2009.

Sharp, B R; Waugh, S M; Walker, N A (2011). A risk assessment framework for incidental seabird mortality associated with commercial fishing in the New Zealand EEZ. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Sommer, E; Bell, B; Boyle, D (2008) Antipodes Island white-chinned petrel trip report, 2008. Final Research Report for Ministry of Fisheries project PRO2006-01. 15 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Sommer, E; Boyle, D; Baer, J; Fraser, M; Palmer, D; Sagar, P (2010) Antipodes Island white-chinned petrel and grey petrel field work report, 2009–10. Final Research Report for Ministry of Fisheries project PRO2006-01. 13 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Sommer, E; Boyle, D; Fraser, M (2009) Antipodes white-chinned petrel and grey petrel trip report, 2009. Final Research Report for Ministry of Fisheries project PRO2006-01. 19 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Soykan, C U; Moore, J E; Zydelis, R; Crowder, L B; Safina, C; Lewison, R L (2008) Why study bycatch? An introduction to the Theme Section on fisheries bycatch. *Endangered Species Research 5*: 91–102.

Sullivan, B J; Brickle, P; Reid, T A; Bone, D G; Middleton, D A J (2006a) Mitigation of seabird mortality on factory trawlers: Trials of three devices to reduce warp cable strikes. *Polar Biology 29*: 745–753.

Sullivan, B J; Reid, T A; Bugoni, L (2006b) Seabird mortality on factory trawlers in the Falkland Islands and beyond. *Biological Conservation 131*: 495–504.

Taylor, G A (2000) Action plan for seabird conservation in New Zealand. Part A, threatened seabirds. Threatened Species Occasional Publication No. 16. Department of Conservation, Wellington. 234 p.

Thompson, F N; Abraham, E R (2009) Six monthly summary of the capture of protected species in New Zealand commercial fisheries, summer 2007–08. *New Zealand Aquatic Environment and Biodiversity Report No. 35*. 22 p.

Tuck, G N; Phillips, R A; Small, C; Thomson, R B; Klaer, N; Taylor, F; Wanless, R; Arrizabalaga, H (2011) An assessment of seabird–fishery interactions in the Atlantic Ocean. *ICES Journal of Marine Science 68*: 1628–1637.

Tuck, G N; Polacheck, T; Croxall, J P; Weimerskirch, H; Ryan, P; Nel, D; Wayte, S; Bulman, C M (2004) Modelling the impact of fishery incidental mortality on three populations of wandering albatross. pp 149–241. *In*: Tuck, G N (Ed.), A comprehensive study of the ecological impacts of the worldwide pelagic longline industry: Southern Hemisphere studies. CSIRO, Melbourne, Australia.

Vié, J-C; Hilton-Taylor, C; Stuart, S N (Eds.) (2009) Wildlife in a changing world – an analysis of the 2008 IUCN Red List of Threatened Species. IUCN, Gland, Switzerland. 180 p.

Wade, P R (1998) Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. *Marine Mammal Science 14*: 1–37.

Wagner, E L; Boersma, P D (2011) Effects of fisheries on seabird community ecology. *Reviews in Fisheries Science 19*: 157–167.

Walker, K; Elliott, G (1999) Population changes and ecology of the wandering albatross *Diomedia exulans gibsoni* at the Auckland Islands. *Emu 99*: 239–247.

Walker, K; Elliott, G (2009) Gibson's wandering albatross research Adams Island 2009. 11 p. Unpublished report produced for the Department of Conservation.

Walker, N; Smith, N; Sharp, B; Cryer, M (2015) A qualitative review of New Zealand's 2013 level two risk assessment for seabirds. *New Zealand Fisheries Science Review 2015/1*. 58 p.

Watkins, B P; Petersen, S L; Ryan, P G (2008) Interactions between seabirds and deep-water hake trawl gear: an assessment of impacts in South African waters. *Animal Conservation 11*: 247–254.

Waugh, S M; Baker, G B; Gales, R; Croxall, J P (2008a) CCAMLR process of risk assessment to minimise the effects of longline fishing mortality on seabirds. *Marine Policy 32*: 442–454.

Waugh, S M; Filippi D (2009) Ecological risk assessment for seabirds in New Zealand fisheries: primary components of ecological risk assessment methodology and calculating vulnerability and productivity for species. Research Progress Report for Ministry of Fisheries research project IPA2009-19. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Waugh S M; Filippi, D; Walker, N A; Kirby, D S (2008b) Updated preliminary results of an ecological risk assessment for seabirds and marine mammals with risk of fisheries interactions. Western Central Pacific Fisheries Commission Scientific Committee 4th Regular Session, August 2008, Port Moresby, Papua New Guinea. WCPFC-SC4-2008/EB-WP-2.

Waugh, S M; Jamieson, S E; Stahl, J-C; Filippi, D P; Taylor, G A; Booth, A (2014) Flesh-footed shearwater – population study and foraging areas. Project POP2011-02. Report to the Department of Conservation, Wellington.

Waugh, S M; Taylor, G (2012) Annual Report on Project POP2011-02 Flesh-footed Shearwaters - population study trial and at-sea distribution. Project POP2011-02. Report to the Department of Conservation, Wellington.

Waugh, S M; Tennyson, A J D; Taylor, G A; Wilson, K (2013) Population sizes of shearwaters (*Puffinus* spp.) breeding in New Zealand, with recommendations for monitoring. *Tuhinga 24*: 159–204.

Webber, D (2014) Independent review of the seabird risk assessment. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

WGSE (2011) Report of the International Council for the Exploration of the Sea (ICES) Working Group on Seabird Ecology (WGSE), 1–4 November 2011, Madeira, Portugal. ICES CM 2011/SSGEF:07. 73 p.

Wienecke, B; Robertson, G (2002) Seabird and seal-fisheries interactions in the Australian Patagonian toothfish *Dissostichus eleginoides* trawl fishery. *Fisheries Research 54*: 253–265.

Wood, C (2006) (cited in Baker et al. 2011a). Westland petrel (*Procellaria westlandica*) colony and burrow distribution survey 2003–2005. (Unpublished report held by Department of Conservation, Wellington, New Zealand).

THEME 2: NON-TARGET FISH AND INVERTEBRATE CATCH

9 NON-TARGET FISH AND INVERTEBRATE CATCH

Status of chapter	This chapter has been fully updated for AEBAR 2018.
Scope of chapter	This chapter outlines the main fish and invertebrate species caught as non-target species catch in New Zealand's major offshore fisheries, with summaries of the amounts caught and discarded. Much of the research in this field has been conducted fishery by fishery with no spatial breakdown of annual catch totals, but this summary incorporates a re-assessment of historical analyses for deepwater fisheries, with stratification aligned to standardised areas. This provides estimates of non-target species catch and discarding across all offshore fisheries within separate regions of New Zealand fisheries. Research begun in 2013 to analyse individual non-target species catch over time for each of the Tier 1 Deepwater fisheries has continued, with updated estimates for some fisheries and for the 2016–17 fishing year. This chapter presents the latest available information, however the last date of detailed analysis differs between fisheries, e.g., the Jack Mackerel fishery was updated to 2013–14, and the Hoki/Hake/Ling fishery is currently being updated to 2016–17 (see Table 9.1 for more details). The fisheries summarised are as follows:
Area	Total annual non-target species catch and discard levels are summarised for 10 of the 11 fishery areas shown in Figure 9.1. Non-target species catch from the small amount of fishing in the Kermadec area is not addressed.

Trawl fisheries	Longline fisheries	Other fisheries
Arrow squid	Ling (bottom)	Albacore tuna troll
Hoki/hake/ling	Tuna (surface)	Skipjack tuna purse seine
Jack mackerel		
Southern blue whiting		
Orange roughy		
Oreo		
Scampi		

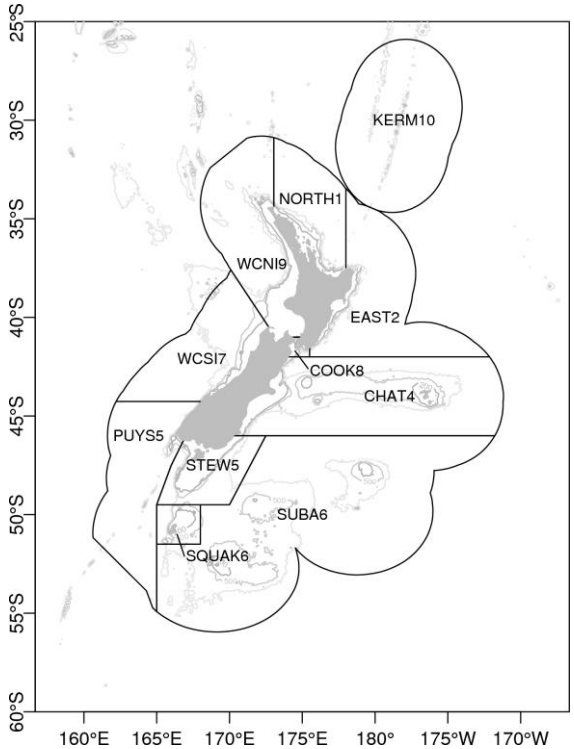


Figure 9.1: Standardised assessment areas for estimation of total non-target fish and invertebrate catch in offshore fisheries.

Focal localities	**Trawl fisheries** *Arrow squid*: Auckland Islands and Stewart/Snares Shelf (80–300 m). *Hoki/hake/ling*: Chatham Rise, west coast South Island, Campbell Plateau, Puysegur Bank, and Cook Strait (200–800 m). *Jack mackerel*: West coast of the North and South Islands, Chatham Rise, and Stewart-Snares Shelf (0–300 m). *Southern blue whiting*: Campbell Plateau and Bounty Plateau (250–600 m). *Orange roughy*: The entire New Zealand region (700–1200 m). *Oreos*: South Chatham Rise, Pukaki Rise, Bounty Plateau, and Southland (700–1200 m). *Scampi*: East coasts of the North and South Islands, Chatham Rise, and Auckland Islands (300–450 m). **Longline fisheries** *Ling (bottom)*: Chatham Rise, Bounty Plateau, and Campbell Plateau (150–600 m). *Tuna (surface)*: East coast of the North Island and west coast of the South Island. **Other fisheries** *Albacore tuna troll*: West coasts of the North and South Islands. *Skipjack tuna purse seine*: Northern North Island
Key issues	• Lack of data on non-target species catch and discards for most inshore (0–200 m) fisheries because of low observer coverage, and simpler reporting requirements prior to 1 October 2007, which saw most catch and effort data aggregated per day and by statistical area (Catch Effort and Landing Return). Collection of more detailed fishing event catch and effort data for smaller trawl (6–28 m), longline, and setnet vessels began on 1 October 2007. Implementation of electronic monitoring tools is planned on inshore vessels for data collection which may help address this issue.
Emerging issues	• Trends showing increased rates and levels of catch and discarding of several non-target species or species categories, especially some non-QMS fish species and invertebrates.
Fisheries New Zealand (FNZ) research (current)	ENV2015-04 *Estimation of incidental captures, fish bycatch and discards using electronic monitoring*; DAE2017-01 *Bycatch monitoring and quantification in the HOK/HAK/LIN fishery*; DAE2018-01 *Catch composition in deepwater fisheries (LIN longline, JMA trawl, ORH/OEO trawl)* (upcoming); DAE2017-02 *Taxonomic identification of benthic samples*; DAE2017-04 *Quantification of key bycatch groups across fisheries*.
NZ government research (current)	DOC19301-POP2018-01 *Improved habitat suitability modelling for protected corals in New Zealand waters*
Related chapters/issues	Chondrichthyans (sharks, rays, and chimaeras)

9.1 CONTEXT

For this chapter *non-target species catch* is equivalent to *bycatch* and refers to all fish and invertebrates caught that were not the stated target species for that fishing event whether or not they were discarded (McCaughran 1992). *Discarded catch* (or *discards*) is defined as 'all the fish, both target and non-target species, which are returned to the sea whole as a result of economic, legal, or personal considerations' (McCaughran 1992). *Discarded catch* in this report includes discards of invertebrate species and estimates of any fish lost from the net at the surface.

DEEPWATER TRAWL AND BOTTOM-LONGLINE FISHERIES

The management of non-target fish and invertebrate catch in the deepwater and middle-depth fisheries is described in the National Fisheries Plan for Deepwater and Middle-depth Fisheries (the National Deepwater Plan). Under the National Deepwater Plan, the objective most relevant for management of non-target fish and invertebrate catch is Management Objective 2.4: *Identify and avoid or minimise adverse effects of deepwater and middle-depth fisheries on incidental bycatch species*. Specific objectives for the management of non-target species catch are outlined in the fishery-specific chapters of the National Deepwater Plan. Estimation of non-target species catch is carried out

for each of the Tier-1 Deepwater fisheries on a rotational basis, with each of the following fisheries updated about every 4–5 years:

- arrow squid
- ling bottom longline
- hoki/hake/ling trawl
- jack mackerel trawl
- southern blue whiting trawl
- orange roughy/oreo trawl
- scampi trawl.

SURFACE LONGLINE, TROLL AND PURSE-SEINE FISHERIES

Non-target fish species catch in the fisheries for Highly Migratory Species (HMS) is addressed in the HMS fish plan. Tuna fisheries incidental bycatch is examined, with updates every 1–3 years as required by the relevant commissions. Some data on bycatch in the Albacore tuna troll fishery and the skipjack tuna purse seine fishery are also available.

INSHORE FISHERIES

The three National Fisheries Plans for Inshore species (finfish, shellfish and freshwater fisheries) also include objectives that address non-target species catch, but research on these objectives has yet to be conducted. However, summaries of the main bycatch species have occasionally been included in reports from fisheries characterisation projects, for example school shark, red gurnard, and elephantfish (Starr et al. 2010a, 2010b, 2010c, Starr & Kendrick 2012, Starr & Kendrick 2013).

9.2 GLOBAL UNDERSTANDING

Bycatch of unwanted zero- or low-value species and discarding of these and of target species that are damaged or too small to process are significant issues in many fisheries worldwide. Few, if any, fisheries are completely without bycatch and this issue has been the subject of many studies and international meetings. Saila (1983) made the first comprehensive global assessment and estimated, albeit with very poor information, that at least 6.7 million tonnes was discarded each year. Alverson et al. (1994) extended that work and estimated the global bycatch at 27.0 (range 17.9–39.5) million tonnes each year. An update by Kelleher (2005) suggested global bycatch of about 8% of the global catch, or 7.3 million tonnes, in 1999–2001.

Tropical shrimp trawl fisheries typically have the highest levels of unwanted bycatch, with an average discard rate of 62% (Kelleher 2005), accounting for about one-quarter to one-third of global bycatch. Discard rates in demersal trawl fisheries targeting finfish are much lower but, because they are so widespread, make a considerable contribution to total global discards. Tuna longline fisheries have the next largest contribution and tend to have greater unwanted bycatch than other line fisheries (Kelleher 2005).

The estimated global level of discards reduced considerably since the Alverson et al. (1994) estimate, but differences in the methodology and definition of bycatch used (see Kelleher 2005, Davies et al. 2009) make it difficult to quantify the decline. The main reasons for the estimated decline in bycatch may be due to a combination of higher retention rates, better fisheries management, and more selective/targeted fishing methods.

Bycatch and discard estimation is frequently very coarse, and estimates of rates based on occasional surveys are often scaled up to represent entire fisheries and applied across years, or even to other fisheries (e.g., Bellido et al. 2011). Data from dedicated fisheries observers are also frequently used for individual fisheries, and these are considered to provide the most accurate results, providing that discarding is not illegal (leading to bias due to 'observer effects'; Fernandes et al. 2011). Ratio estimators similar to those historically applied in some New Zealand fisheries are frequently used to scale observed bycatch and discard rates to the wider fishery, and the methods used in New Zealand fisheries are broadly similar to those used elsewhere (e.g., Fernandes et al. 2011, Borges et al. 2005). A new methodology has recently been developed for New Zealand fisheries, which is now replacing the ratio method. This method uses multiple predictor variables in a model-based estimation process fitted using Bayesian methods and has shown in simulation studies to provide estimates with less bias and improved precision (Edwards et al. 2015). This modeling approach has been used alongside the ratio method in two assessments (Anderson et al. 2017a, 2017b), and as the sole method in the most recent assessments (Anderson & Edwards 2018, Anderson et al. in prep).

Discard data are increasingly incorporated into fisheries stock assessments and management decision-making, especially with the move towards an Ecosystem Approach to Fisheries (EAF) (Bellido et al. 2011), and as third party fishery certification schemes more closely examine the

effects of fishing on the ecosystem. These data have also been used to assess impacts on non-target species overseas (e.g., Pope et al. 2000, Casini et al. 2003).

9.3 STATE OF KNOWLEDGE IN NEW ZEALAND

9.3.1 OVERVIEW

Estimation of annual catch and discard levels of non-target species in selected New Zealand fisheries have been undertaken at regular intervals since 1998 (Table 9.1).

Table 9.1: Summary of research into bycatch and discards in New Zealand fisheries.

Trawl fisheries	Report
Arrow squid trawl (SQU)	Anderson et al. (2000) Anderson (2004b) Ballara & Anderson (2009) Anderson (2013a) Anderson (2013b) Anderson (2014b) Ballara (2015) Edwards et al. (2015) Anderson (2017) Anderson & Edwards (2018) Finucci et al. (in prep)
Hoki trawl (HOK)	Clark et al. (2000) Anderson et al. (2001) Anderson & Smith (2005) Ballara et al. (2010) Anderson (2013b) Anderson (2014b) Ballara (2015) Ballara & O'Driscoll (2015) Anderson (2017) Anderson et al. (in prep) Finucci et al. (in prep)
Hake trawl (HAK)	Ballara et al. (2010) Anderson (2013b) Anderson (2014b) Ballara (2015) Ballara & O'Driscoll (2015) Anderson (2017) Anderson et al. (in prep) Finucci et al. (in prep)
Ling trawl (LIN)	Ballara et al. (2010) Anderson (2013b) Anderson (2014b) Ballara (2015) Ballara & O'Driscoll (2015) Anderson (2017) Anderson et al. (in prep) Finucci et al. (in prep)
Ling longline (LLL)	Anderson et al. (2000) Anderson (2008) Anderson (2013a) Anderson (2013b) Anderson (2014a) Anderson (2014b) Ballara (2015) Edwards et al. (2015) Anderson (2017) Finucci et al. (in prep)
Jack mackerel trawl (JMA)	Anderson et al. (2000) Anderson (2004b) Anderson (2007) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson et al. (2017b) Anderson (2017) Finucci et al. (in prep)
Southern blue whiting trawl (SBW)	Clark et al. (2000) Anderson (2004a) Anderson (2009b) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson (2017) Finucci et al. (in prep)
Orange roughy trawl (ORH)	Clark et al. (2000) Anderson et al. (2001) Anderson & Clark (2003) Anderson (2009a) Anderson (2011) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson (2017) Anderson et al. (2017a) Finucci et al. (in prep)
Oreo trawl (OEO)	Clark et al. (2000) Anderson (2004a) Anderson (2011) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson (2017) Anderson et al. (2017a) Finucci et al. (in prep)
Scampi trawl (SCI)	Clark et al. (2000) Anderson (2004a) Ballara & Anderson (2009) Anderson (2012) Anderson (2013b) Anderson (2014b) Edwards et al. (2015) Anderson (2017) Anderson & Edwards (2018) Finucci et al. (in prep)

Other fisheries	Report
Albacore tuna troll	Griggs et al. (2014)
Skipjack tuna purse seine	Anon (2013)
Skipjack tuna purse seine	Anon (2017)
Tuna longline	Griggs et al. (2018)

TRAWL AND BOTTOM-LONGLINE FISHERIES

The estimation process for the trawl and bottom-longline fisheries used rates of bycatch and discards in various categories, i.e., in recent analyses 'all QMS species combined (QMS)', 'all non-QMS fish species combined (non-QMS)', and 'all non-QMS invertebrate species combined (INV)'. It also used fishery strata in the observed fraction of the fishery, and effort statistics from the wider fishery, to calculate annual bycatch and discard levels. The ratio-based approach estimates precision by incorporating a multi-step bootstrap algorithm, which considers the effect of correlation between trawls in the same observed trip and stratum, while the statistical model method estimates uncertainty from the 95% credibility interval of the posterior distribution of model estimates. For this report, additional estimates of annual bycatch and discards within standardised areas (Figure 9.1) were re-calculated from archived data where possible (and where necessary), but without estimates of precision. The original analyses were based on a stratification using different sets of areas and in some cases additional strata such as depth or gear-type. For this re-calculation, the estimated values for each area were scaled so as to have the same annual total as the published values. To enable totals to be calculated across all fisheries within each area, bycatch and discard estimates for years/fisheries where data has yet to become available were assumed to be equal to that of the last year for which an estimate has been published.

Estimates of the annual bycatch of a wide range of individual species were also made in the most recent analysis of the ling longline fishery (Anderson 2014a), hoki/hake/ling fishery (Anderson et al. in prep), jack mackerel fishery (Anderson et al. 2017b), orange roughy and oreo fisheries (Anderson et al. 2017a), and the scampi and arrow squid fisheries (Anderson & Edwards, 2018) as well as in a more simplified manner for the remaining Deepwater Tier 1 fisheries (Finucci et al. in prep).

In some cases the apparent increase or decrease in bycatch of a species is likely to be the result of external factors including the introduction of new species to the QMS, new species-specific 3-letter codes to replace generic codes, and improvements in species identification over time, e.g., the increase in recorded bycatch of floppy tubular sponge in the hoki/hake/ling trawl fishery reflects the improved identification of sponges in more recent years, and use of the species specific code for giant spider crab (GSC) instead of unspecified crabs (CRB) in the hoki/hake/ling trawl fishery. Some codes may also have been misused, e.g., among paddle crab species in the arrow squid fishery where the increase in recorded bycatch of the smooth red swimming crab (*Nectocarcinus bennetti*, NCB) appears to be at the expense of bycatch of the similar-looking *Ovalipes catharus*, PAD), which has a code which may be mistaken for a generic species code for paddle crabs.

The approach used in these analyses has relied heavily on an appropriate level and spread of observer effort being achieved, and this was examined in detail in each published report. Although details of bycatch and discards were recorded directly by vessel skippers for all fishing events through catch-effort forms, these data were generally inadequate for precise measurement of annual totals as the forms only require the top five or eight catch species to be reported, discard information is often not required and they generally lacked the accuracy of identification and precision of observer data. Despite these inadequacies, annual bycatch totals were usually derived from catch effort data, but presented only as secondary estimates.

SURFACE-LONGLINE FISHERIES

The estimation process used for surface-longline fisheries up until the 2014–15 year was similar to that used for trawl and bottom-longline fisheries, with each species assessed separately. In this case CPUE was calculated as the number of fish observed caught per 1000 hooks set stratified by fishing year, fleet (Foreign Licenced, Foreign Chartered and Domestic), and area. CPUE was expressed using a ratio of means estimator (see Bradford 2002, Ayers et al. 2004). The total number of each species caught in each stratum was estimated by scaling up the CPUE to the total number of hooks set. These numbers were then summed across strata to give total annual catch estimates. An analytical estimator was used to calculate variance, using an adjustment to account for correlation between variance and the mean of the effort variable (after Thompson 1992). Additional estimates of annual bycatch within the standardised areas used for the deepwater trawl and bottom-longline fisheries are currently not available.

TROLL AND PURSE SEINE FISHERIES

Fish bycatch research in these fisheries is limited to annual summaries of observer recorded species catches, without any attempts to scale/apply observed catch rates to the total commercial fishery.

INSHORE FISHERIES

Some bycatch information is available from fishery characterisation studies (see Section 9.1) but there were no detailed analyses of bycatch and discards from inshore fishing principally because of the lack of observer data. Most of the analyses of bycatch and discards for offshore fisheries were reliant on observer data, e.g., Anderson 2012, 2013a, and similar analyses for inshore fisheries are not currently possible. Past observer coverage of inshore fisheries has been low (e.g., fewer than 2% of tows observed in 2009–10; Ramm 2012) and coverage has often been issue focused – e.g., monitoring of Hector's and Māui dolphin interactions and abundance for the Threat Management Plan – rather than representative. There are also practical and logistical problems with placing observers on smaller inshore vessels, and other options are being explored for the monitoring of these fisheries.

Detailed fishing event data for inshore fishing, e.g., tow-by-tow catch and effort, were not collected by all vessels before 1 October 2007 using the statutory reporting system. Before 1 October 2007, smaller trawl (6–28 m), longline, and set net vessels used the Catch Effort and Landing Return (CELR) to collect daily summary catch-effort and landings data by general statistical area. From 1 October 2007 onwards, detailed data for each fishing event were collected using the new Trawl Catch and Effort Return (TCER), and this will be used to support analyses of bycatch in inshore fisheries.

Electronic reporting and monitoring is being implemented in a phased manner across all New Zealand fisheries (including for the inshore). Some progress has been made with estimating the bycatch of undersized fish such as sub-legal-sized snapper (SNA), but there remain some issues to surmount before electronic monitoring can provide all the information required to estimate fish and invertebrate bycatch.

9.3.2 CHATHAM RISE (CHAT)

The Chatham Rise is an important region for all the major deepwater fisheries except for southern blue whiting. Total bycatch from deepwater fisheries has ranged from about 15 000 t to about 37 000 t, with generally decreasing amounts after 2000–01, then rising again after 2010–11 (Figure 9.2). In each year since 1995–96 the combined trawl fishery targeting hoki, hake, ling, silver warehou and white warehou has been the main contributor to total bycatch in this area. Prior to that most of the bycatch from deepwater fisheries was attributed to the orange roughy fishery. The arrow squid and scampi fisheries also contributed substantially in some years.

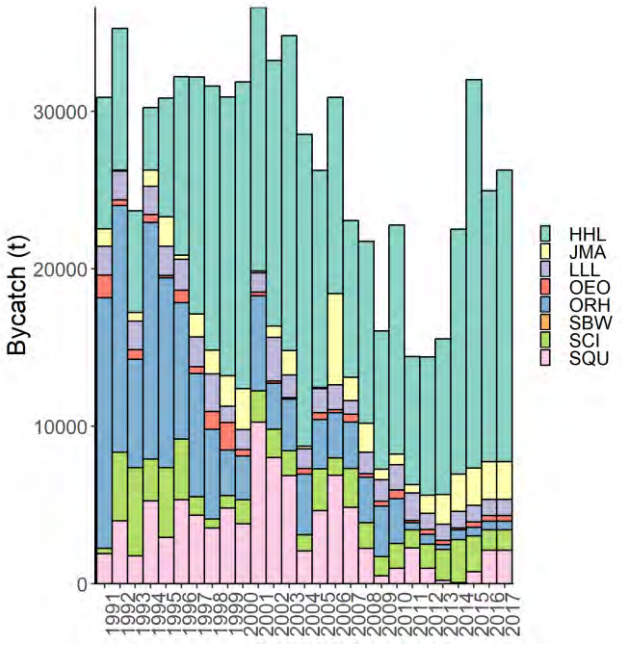


Figure 9.2: Estimated total annual bycatch (source fisheries shown by bar colouration) in Chatham Rise deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards on the Chatham Rise from deepwater fisheries has ranged from about 4000 t to about 25 000 t, with generally decreasing amounts after 2002–03 (Figure 9.3). In most years the largest contributor to discards by volume in this area was the hoki/hake/ling trawl fishery. Discards were relatively low in the orange roughy fishery compared to bycatch, as a large part of the bycatch in that fishery, especially in earlier years, is likely to have been commercial species – oreos in particular.

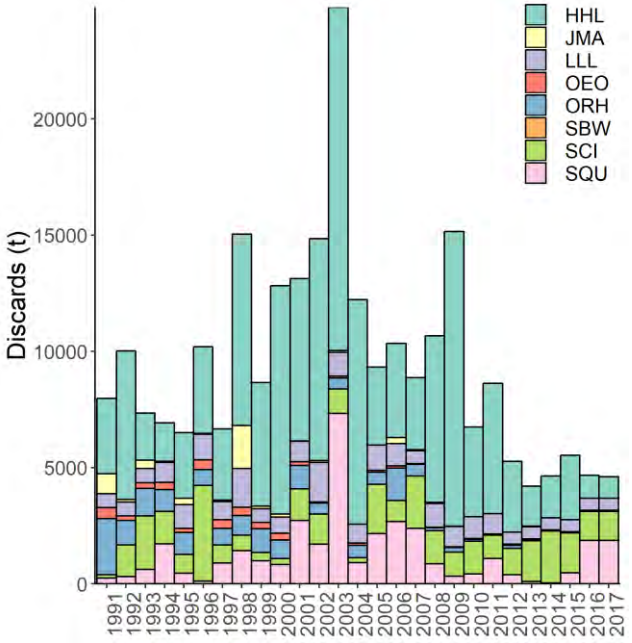


Figure 9.3: Estimated total annual discards (source fisheries shown by bar colouration) in Chatham Rise deepwater fisheries. For fisheries abbreviations see Table 9.1.

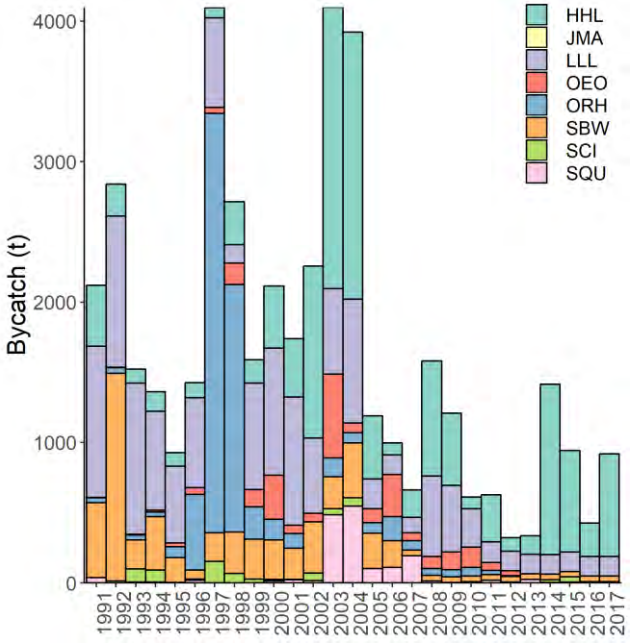


Figure 9.4: Estimated total annual bycatch (source fisheries shown by bar colouration) in Sub-Antarctic deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.3 SUBANTARCTIC (SUBA)

The Sub-Antarctic is an important region for all the major deepwater fisheries except for jack mackerel. Total bycatch from deepwater fisheries in the area has ranged from about 300 t to about 4100 t, with variable levels but generally lower since 2004–05 (Figure 9.4). In the past, major contributors have been the ling longline, southern blue whiting, orange roughy, and hoki/hake/ling fisheries. Most recently the hoki/hake/ling (including longline) fisheries have been the greatest contributors.

Total discards in the Sub-Antarctic from deepwater fisheries have ranged from about 200 t to about 2200 t, with generally decreasing levels over time, especially after 2002–03 (Figure 9.5). Discards in the southern blue whiting fishery are high relative to bycatch due to the discarding of target species. Currently total discards are mostly split between the southern blue whiting and hoki, hake, ling trawl fisheries, and the ling longline fishery.

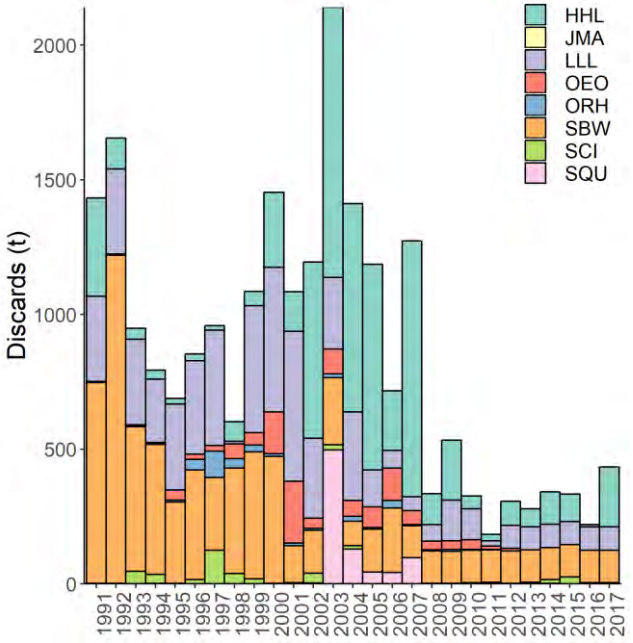


Figure 9.5: Estimated total annual discards (source fisheries shown by bar colouration) in Sub-Antarctic deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.4 STEWART-SNARES SHELF (STEW)

The Stewart-Snares Shelf is an important region for the jack mackerel, hoki/hake/ling and arrow squid trawl fisheries, with smaller fisheries also operating for oreo and orange roughy and ling (longline). Total bycatch in the Stewart-Snares Shelf area from deepwater fisheries has ranged from about 3000 t to about 32 000 t per year, with the lowest values in the mid-1990s, but lower levels also after 2005–06 (Figure 9.6). The majority of this bycatch, in all years except for 1994–95, has been in the arrow squid fishery, with most of the remainder coming from the hoki/hake/ling fishery. In several years there has also been a notable contribution from the jack mackerel fishery.

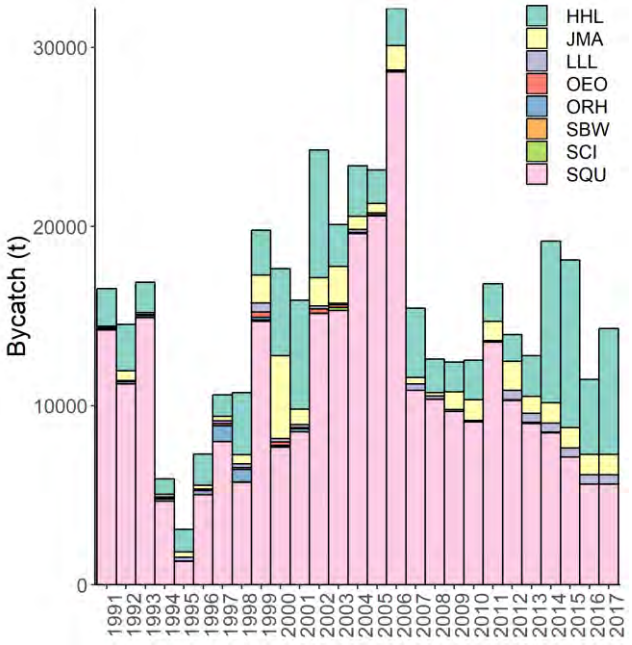


Figure 9.6: Estimated total annual bycatch (source fisheries shown by bar colouration) in Stewart-Snares Shelf deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the Stewart-Snares Shelf area from deepwater fisheries has ranged from about 500 t to about 7000 t, with lower values in the mid-1990s (Figure 9.7). Currently discarding in this area is mostly attributed to the arrow squid and hoki/hake/ling fisheries.

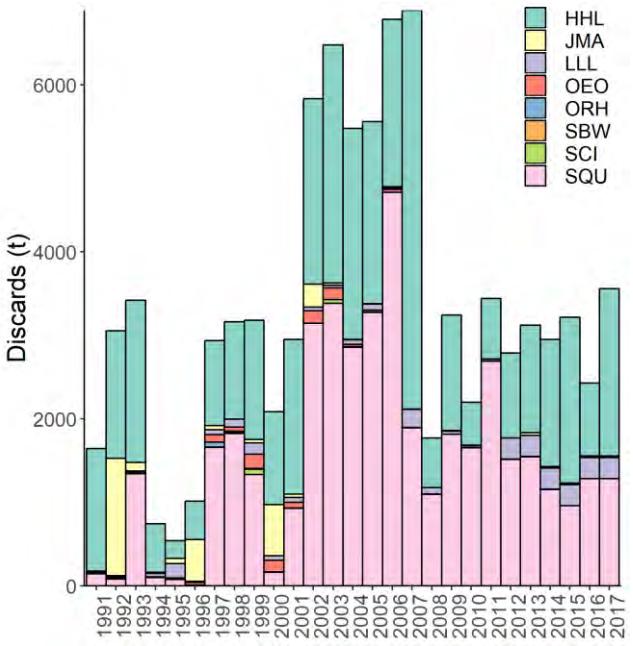


Figure 9.7: Estimated total annual discards (source fisheries shown by bar colouration) in Stewart-Snares Shelf deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.5 AUCKLAND ISLANDS (AK)

The main fisheries currently operating in the Auckland Islands region are the scampi and arrow squid trawl fisheries, with smaller fisheries for hoki/hake/ling also present. An orange roughy fishery operated in the region from the mid-1990s to the early 2000s, but has been very minor in recent years. Total bycatch in the Auckland Islands area from deepwater fisheries has ranged from about 750 t to about 7500 t per year, but fluctuating between about 1800 t and 4200 t since 1997–98 (Figure 9.8). The main contributors to bycatch in this area have been the scampi and arrow squid fisheries, as well as the orange roughy fishery during the 1990s. Currently the main contributing fishery is the arrow squid fishery.

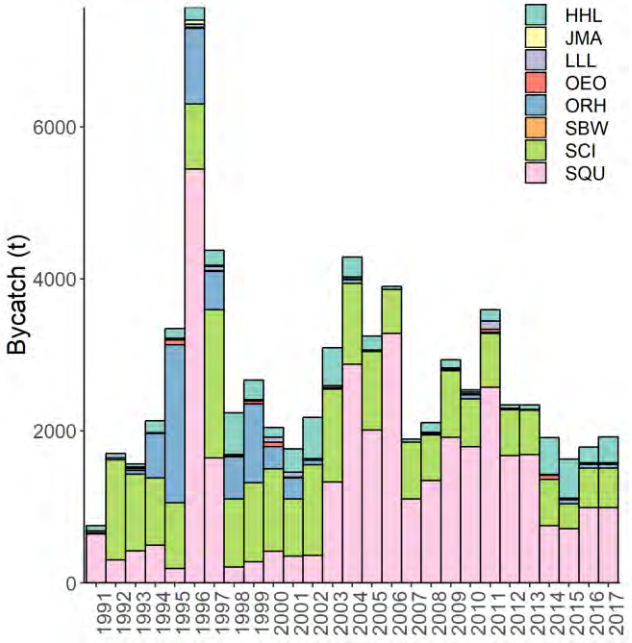


Figure 9.8: Estimated total annual bycatch (source fisheries shown by bar colouration) in Auckland Islands deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the Auckland Islands area from deepwater fisheries have ranged widely, from about 100 t to about 3000 t per year, resulting mostly from the scampi and arrow squid fisheries (Figure 9.9). Although variable, current levels are typical of the last 15 years.

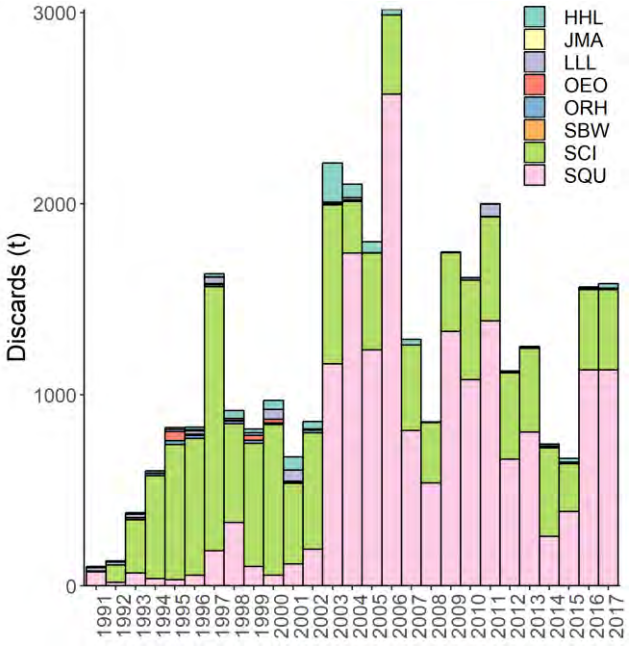


Figure 9.9: Estimated total annual discards (source fisheries shown by bar colouration) in Auckland Islands deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.6 PUYSEGUR (PUYS)

Most deepwater fisheries have operated at some time in the Puysegur area, with bycatch mainly attributed to the orange roughy fishery in the early 1990s, to the arrow squid fishery in the early 2000s, and to the hoki/hake/ling fishery since the mid-2000s. Total bycatch in the area from deepwater fisheries has ranged from about 200 t to about 4500 t per year, with generally decreasing amounts between about 2000–01 and 2009–10 followed by higher levels after 2012–13 (Figure 9.10). Annual bycatch is currently relatively low, and attributed mostly to the hoki/hake/ling trawl fishery, with smaller contributions from the arrow squid and orange roughy fisheries.

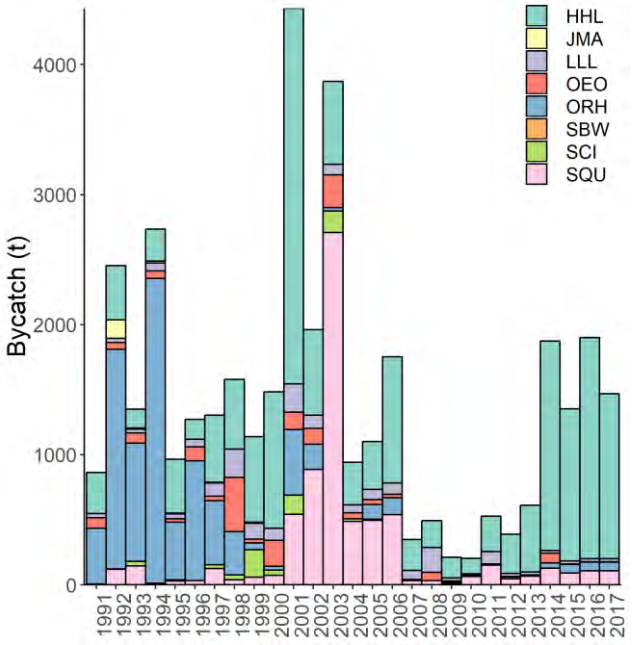


Figure 9.10: Estimated total annual bycatch (source fisheries shown by bar colouration) in Puysegur deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the Puysegur area from deepwater fisheries has ranged from about 100 t to about 4200 t per year with mostly lower amounts after 2002–03, a year in which increased effort in the arrow squid fishery coupled with some large discards resulted in a high estimate for that fishery (Figure 9.11). Discards were mostly attributable to the hoki/hake/ling fishery except for a few years in the early 2000s when the arrow squid fishery was operating more in this area.

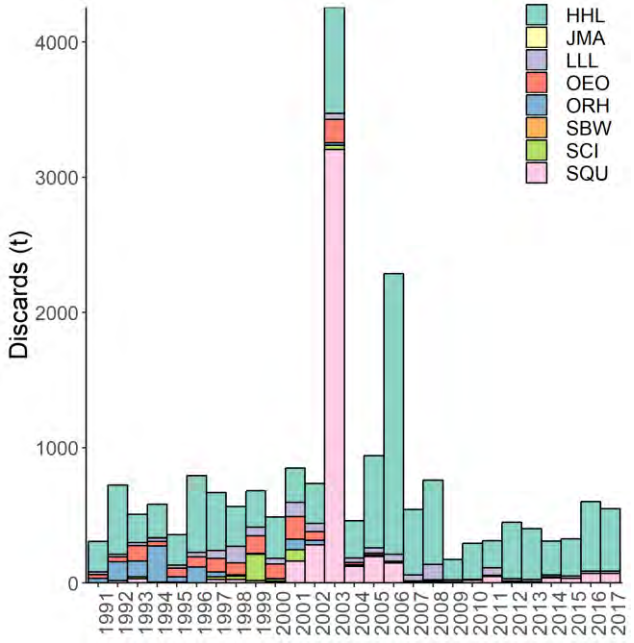


Figure 9.11: Estimated total annual discards (source fisheries shown by bar colouration) in Puysegur deepwater fisheries. For fisheries abbreviations see Table 9.1.

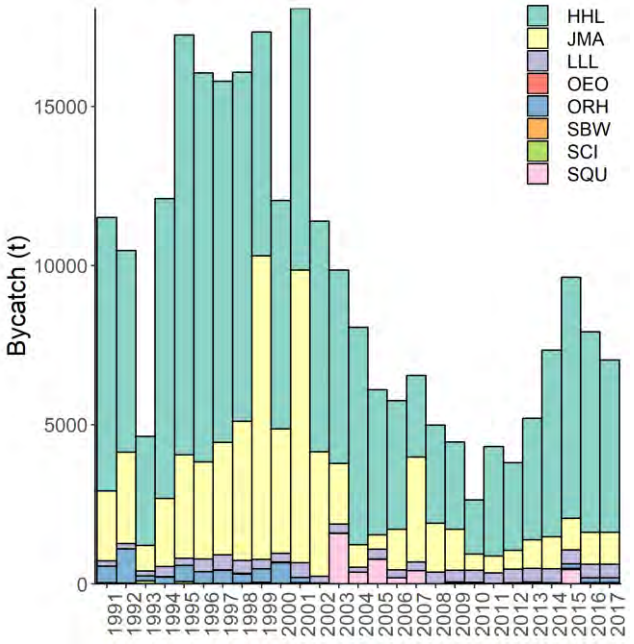


Figure 9.12: Estimated total annual bycatch (source fisheries shown by bar colouration) in west coast South Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.7 WEST COAST SOUTH ISLAND (WCSI)

The main fisheries in this area are the trawl fisheries for hoki/hake/ling and jack mackerel, as well as a small ling longline fishery and, at times, orange roughy and arrow squid trawl fisheries. Currently most bycatch can be attributed to the hoki/hake/ling fishery. Total bycatch in the west coast South Island area from deepwater fisheries has ranged from about 2500 t to about 18 000 t per year, with generally decreasing amounts between about 2000–01 and 2009–10, then increasing levels over the last few years (Figure 9.12).

Total discards in the west coast South Island area from deepwater fisheries has ranged from about 700 t to about 9000 t per year, with generally decreasing amounts since about 1994–95 and a relatively low contribution from the jack mackerel fishery compared to bycatch (Figure 9.13). Total discards have been below 4000 t per year since 2003–04, attributed mostly to the hoki/hake/ling fishery, with a small contribution from the ling longline fishery.

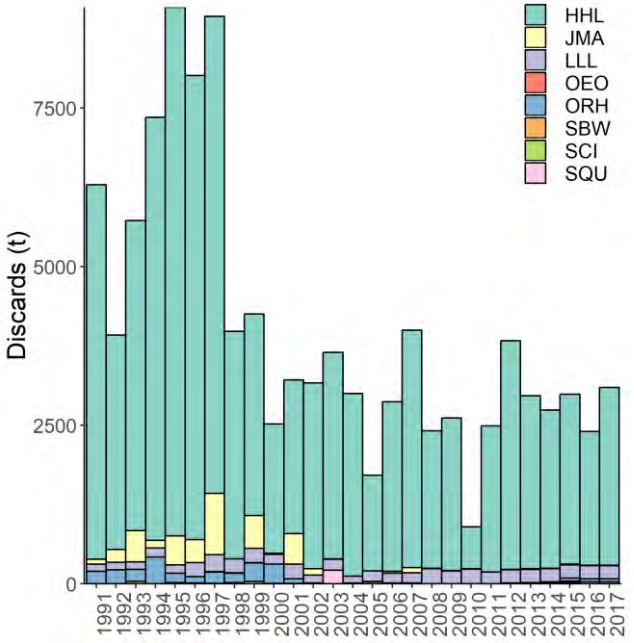


Figure 9.13: Estimated total annual discards (source fisheries shown by bar colouration) in west coast South Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.8 WEST COAST NORTH ISLAND (WCNI)

The dominant deepwater fishery on the west coast North Island region is currently the jack mackerel trawl fishery, with fisheries for orange roughy and arrow squid operating mainly before 2003–04. Total bycatch in the west coast North Island area from deepwater fisheries has ranged from about 1100 t to about 13 000 t per year, with generally decreasing amounts since 2003–04 (Figure 9.14). In most years almost all of the bycatch can be attributed to the jack mackerel fishery, but with moderate contributions from the orange roughy and arrow squid fisheries in some earlier years.

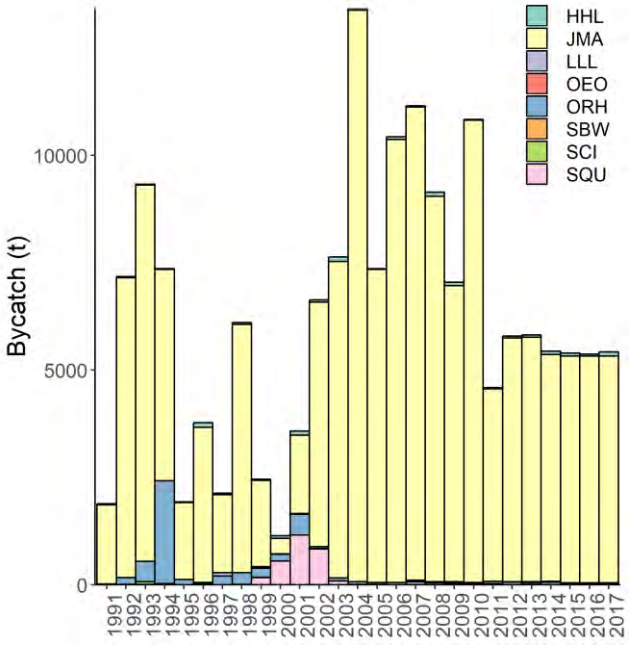


Figure 9.14: Estimated total annual bycatch (source fisheries shown by bar colouration) in west coast North Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the west coast North Island area from deepwater fisheries has ranged from about 20 t to about 1400 t per year, with generally stable levels of 100–300 t per year since 2003–04 (Figure 9.15). The jack mackerel fishery contributes relatively less to total discards than it does to bycatch but still dominates in most years.

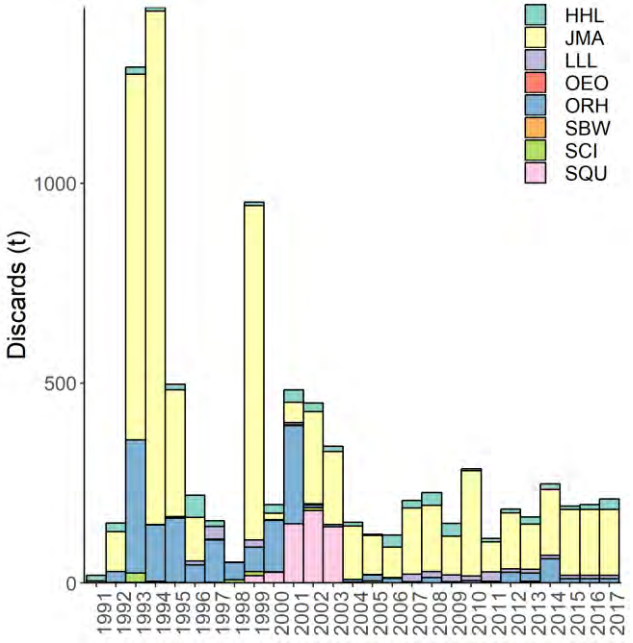


Figure 9.15: Estimated total annual discards (source fisheries shown by bar colouration) in west coast North Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.9 NORTHLAND (NRTH)

Deepwater fisheries in the Northland region are mainly limited to a trawl fishery for scampi, and smaller fisheries for orange roughy and hoki/hake/ling. Total bycatch in the area from deepwater fisheries has ranged from about 500 t to about 5000 t per year, but with generally stable levels of less than 1400 t per year since about 1998–99 (Figure 9.16). In most years bycatch was mainly associated with the scampi fishery, with smaller amounts from the hoki/hake/ling fishery and a large contribution from the orange roughy fishery in 1996–97. Other deepwater fisheries are minor in this area and currently total annual bycatch is less than 1000 t, split mostly between the scampi and hoki/hake/ling fisheries.

Total discards in the Northland area from deepwater fisheries have ranged from about 300 t to about 1000 t per year, with levels of about 300–500 t per year over the last several years (Figure 9.17). Discards in this area are dominated by the scampi fishery in all but two years in which the hoki/hake/ling and orange roughy fisheries contributed more.

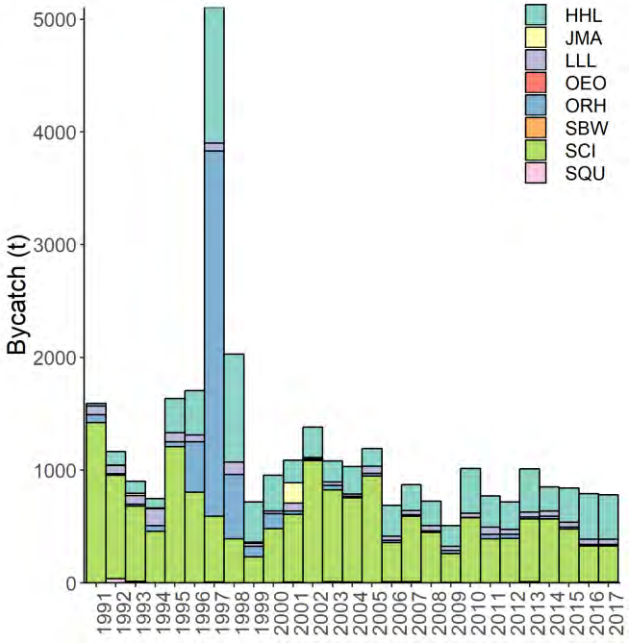


Figure 9.16: Estimated total annual bycatch (source fisheries shown by bar colouration) in Northland deepwater fisheries. For fisheries abbreviations see Table 9.1.

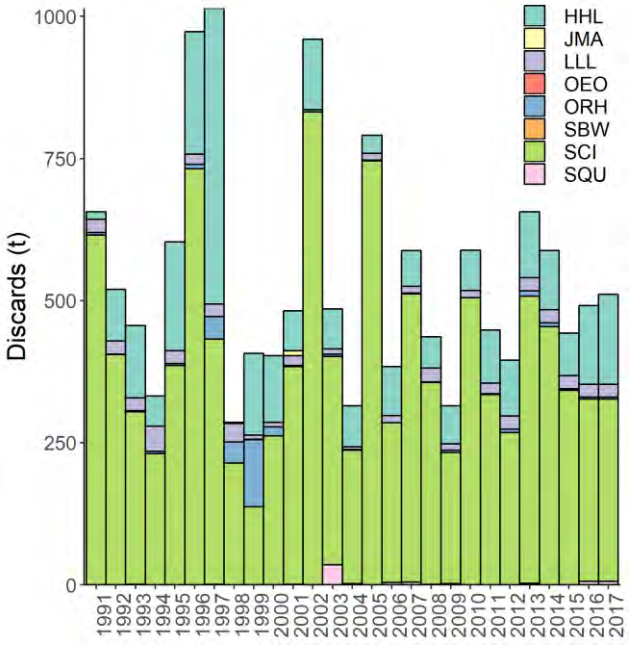


Figure 9.17: Estimated total annual discards (source fisheries shown by bar colouration) in Northland deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.10 EAST COAST NORTH ISLAND (EAST)

The main deepwater fisheries operating in the East Coast North Island have been the scampi, hoki/hake/ling, and orange roughy fisheries, and the ling longline fishery. Total bycatch in the area from deepwater fisheries has ranged from about 1000 t to about 7500 t per year, with generally decreasing levels since about 1997–98 (Figure 9.18). Most of the bycatch comes from the hoki/hake/ling and scampi fisheries, with larger contributions from the orange roughy fishery before 2004–05, and lower contributions from the ling longline fishery.

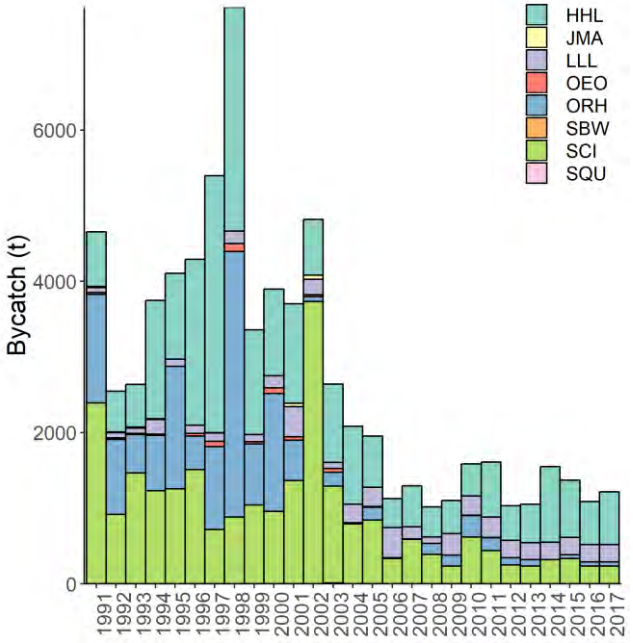


Figure 9.18: Estimated total annual bycatch (source fisheries shown by bar colouration) in East Coast North Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the East Coast North Island area from deepwater fisheries has ranged from about 300 t to about 2700 t per year, with generally lower levels after 2001–02 (Figure 9.19). The scampi and orange roughy fisheries contributed more to discards than to bycatch in this area, and in most years only a small proportion of total discards was attributable to the hoki/hake/ling fishery. Current annual discards are about 400–500 t, mostly associated with the scampi fishery.

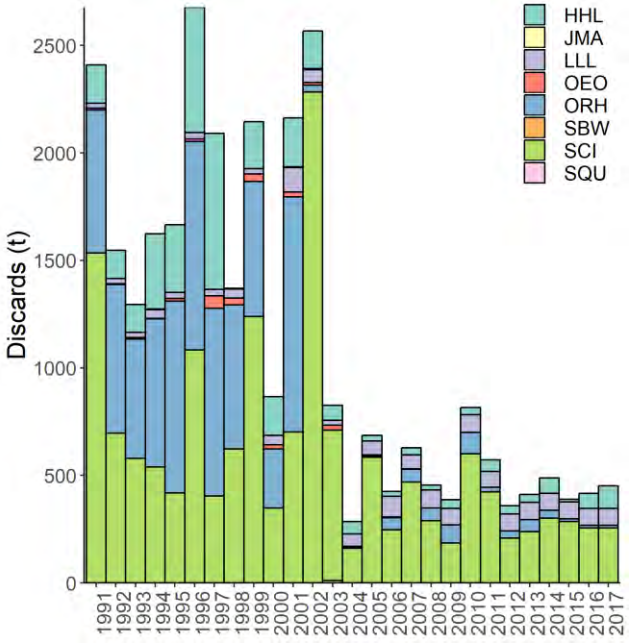


Figure 9.19: Estimated total annual discards (source fisheries shown by bar colouration) in East Coast North Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

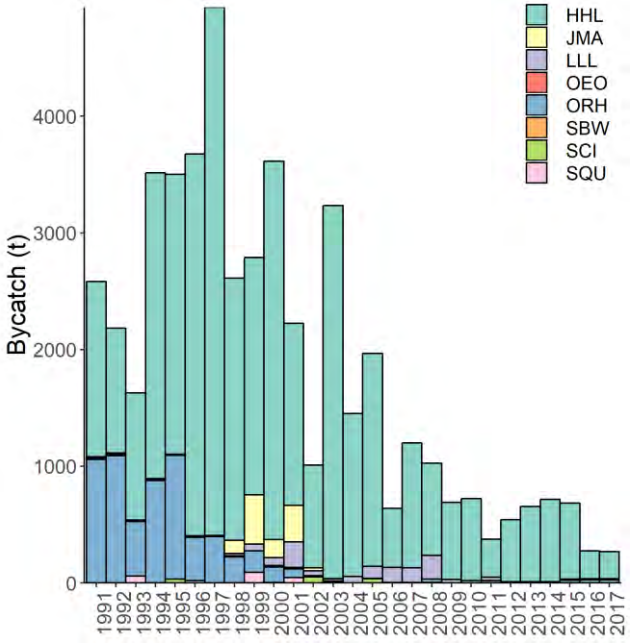


Figure 9.20: Estimated total annual bycatch (source fisheries shown by bar colouration) in Cook Strait deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.11 COOK STRAIT (COOK)

The main fishery in the Cook Strait area has been the hoki/hake/ling trawl fishery, with this fishery contributing the great majority of bycatch in most years. Total bycatch in the Cook Strait area from deepwater fisheries has ranged from about 200 t to about 5000 t per year, with generally decreasing levels since about 1996–97 (Figure 9.20). The orange roughy fishery operating on the fringes of this area also contributed substantially to total annual bycatch through the early 1990s. Currently total annual bycatch is less than 800 t, almost all from the hoki/ hake/ling fishery.

Total discards in the Cook Strait area from deepwater fisheries has ranged from about 200 t to about 4000 t per year, with generally decreasing levels since about 1995–96 (Figure 9.21). Discards in this area have virtually all been associated with the hoki/ hake/ling fishery. Current discard levels are about 200–300 t per year.

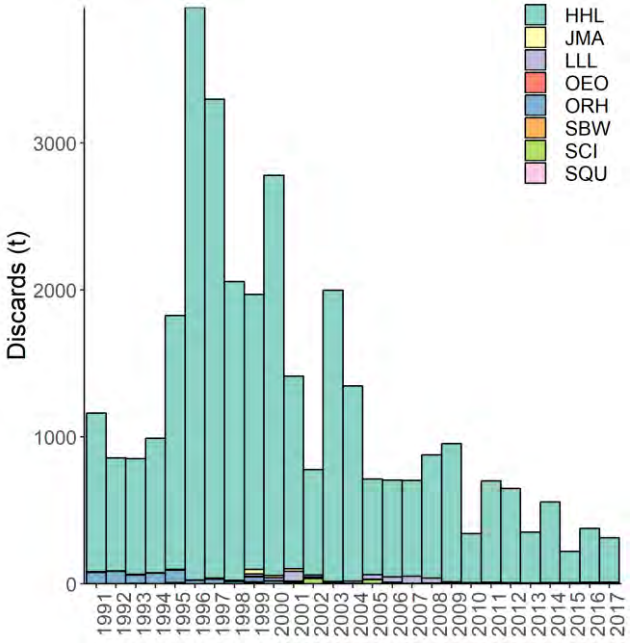


Figure 9.21: Estimated total annual discards (source fisheries shown by bar colouration) in Cook Strait deepwater fisheries. For fisheries abbreviations see Table 9.1.

9.3.12 ARROW SQUID TRAWL FISHERY

Since 1990–91 the level of observer coverage in this fishery was 6–97% of the total annual catch and was relatively high, 28–40%, from 2006–07 to 2010–11 due to the management measures imposed for the protection of New Zealand sea lions (*Phocarctos hookeri*) (Ministry for Primary Industries, 2012) and higher still after 2011–12, 90–97%, due to 100% coverage requirements for Foreign Charter Vessels (FCVs). This coverage was well spread across the fleet and annually 10–71% of all vessels targeting arrow squid were observed, with this fraction increasing over time. Observers covered the full size range of vessels operating in the fishery, although the smallest vessels were slightly undersampled and the largest oversampled.

The observer effort was mostly focused on the main arrow squid fisheries around the Auckland Islands and Stewart-Snares Shelf, but the smaller fisheries on the Puysegur Bank and off Banks Peninsula were also covered, although less consistently. Observer coverage was more focused on the central period of the arrow squid season, February to April, than the fleet was in general – with fishing in January and May slightly undersampled.

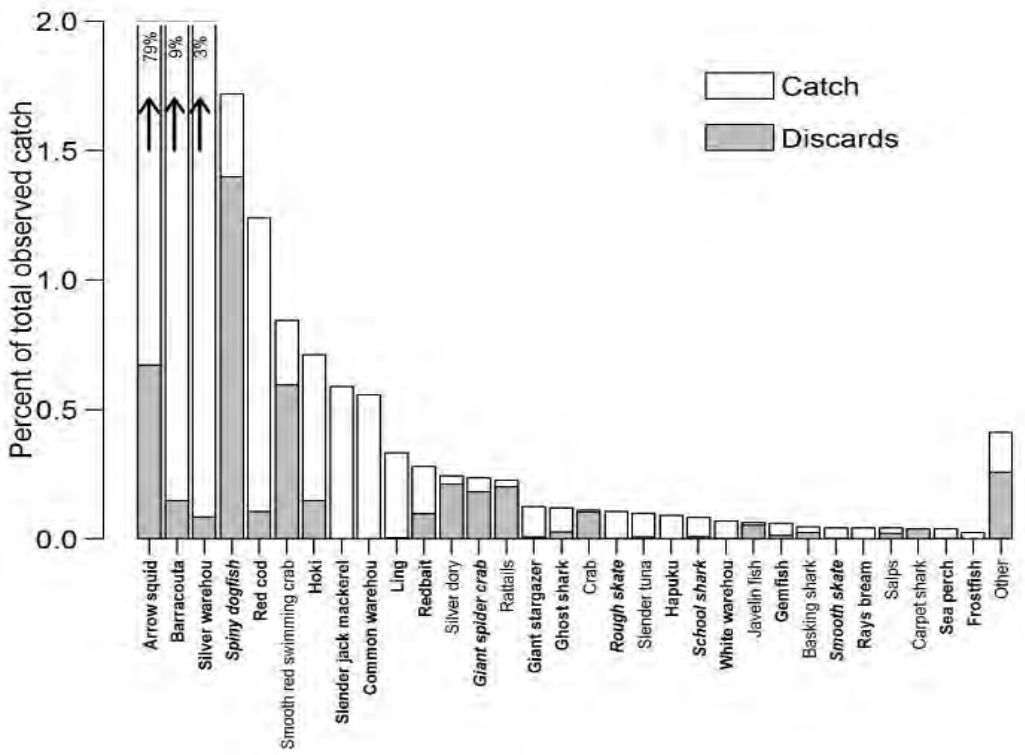


Figure 9.22: Percentage of the total catch contributed by the main bycatch species (those representing 0.02% or more of the total catch; white) in the observed portion of the arrow squid fishery, and the percentage discarded (grey), 1 October 2001 to 30 September 2016 (Anderson & Edwards, 2018). The 'Other' category is the sum of all bycatch species representing less than 0.02% of the total catch. QMS species are shown in bold, Schedule 6 species are in italics.

The most recent assessment of bycatch and discards in this fishery (Anderson & Edwards, 2018) was based on a statistical model approach using a combination of standard areas, fishing years, net type, and meal plant usage as model covariates, and covered the period from 2002–03 to 2015–16. The key categories of catch/discards examined were; all QMS species combined, all non-QMS species combined, and all invertebrate species combined, with membership of these categories adjusted from year to year as species were added to the QMS. Total annual bycatch in the arrow squid fishery for 2002–03 to 2015–16 was about 9000–40 000 t, with a significant downward trend (Figure 9.23). The large majority of the bycatch comprised QMS species, with less than 1000 t of non-QMS species and invertebrate species bycatch in most years.

Since 1990–91, nearly 600 bycatch species or species groups were identified by observers in this fishery, most being non-commercial species (including invertebrate species) caught in low numbers. Arrow squid accounted for about 79% of the total estimated catch recorded by

observers. The main bycatch species or species groups were the QMS species barracouta (9.1%), silver warehou (3.3%), spiny dogfish (1.7%), and red cod (1.2%); and of these only spiny dogfish were generally discarded (Figure 9.22), which is legally allowed under Schedule 6 of the Fisheries Act.

Of the other (non-squid) invertebrate groups, crustaceans (1.2%), in particular smooth red swimming crab (*Nectocarcinus bennetti*) (0.8%), were caught in the greatest amounts and were mostly discarded. Smaller amounts of octopus and squid, sponges, cnidarians, and echinoderms were also often caught and discarded.

When combined into broader taxonomic groups, bony fish (excluding rattails, tuna, flatfish, and eels) contributed the most bycatch (15.9% of the total catch), followed by sharks and dogfishes (1.9%), morid cods (1.2%), crustaceans (1.2%), and rattails (0.3%).

More than 75% of the sharks, dogfishes, and rattails were discarded, whereas most of the catch of the other groups was retained. The fish species discarded in the greatest amounts were spiny dogfish, rattails, and silver dory. Of the invertebrates, most were discarded, but crustaceans, octopuses, and other molluscs were sometimes retained.

Estimated total annual discards ranged from about 1300 t in 2013–14 to about 16 000 t in 2002–03 and, like bycatch, showed a significant decline over time (Anderson & Edwards, 2018). Discards were an even mix of QMS species (about 44% for all years) and non-QMS species (41%), with lesser amounts of invertebrate species (15%), and arrow squid (8%) (Figure 9.24)

TRENDS IN ESTIMATED BYCATCH BY SPECIES FROM THE ARROW SQUID TRAWL FISHERY

Finucci et al. (in prep) estimated the level of the main individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most commonly caught bycatch species were barracouta (*Thyrsites atun*, BAR), silver warehou (*Seriolella punctata*, SWA), and spiny dogfish (*Squalus acanthias*, SPD).
- Of the 347 bycatch species examined, 15 showed a significant decrease in catch over time and 29 had a significant increase in catch.
- The species showing the greatest declines were paddle crab (*Ovalipes catharus*, PAD), jack mackerels (*Trachurus* spp., JMA), and thresher shark (*Alopias vulpinus*, THR) (Figure 9.25).
- The species showing the greatest increases were giant spider crab (*Jacquinotia edwardsii*, GSC), smooth red swimming crab (*Nectocarcinus bennetti*, NCB), and GON (*Gonorynchus forsteri* & *G. greyi*) (Figure 9.25).

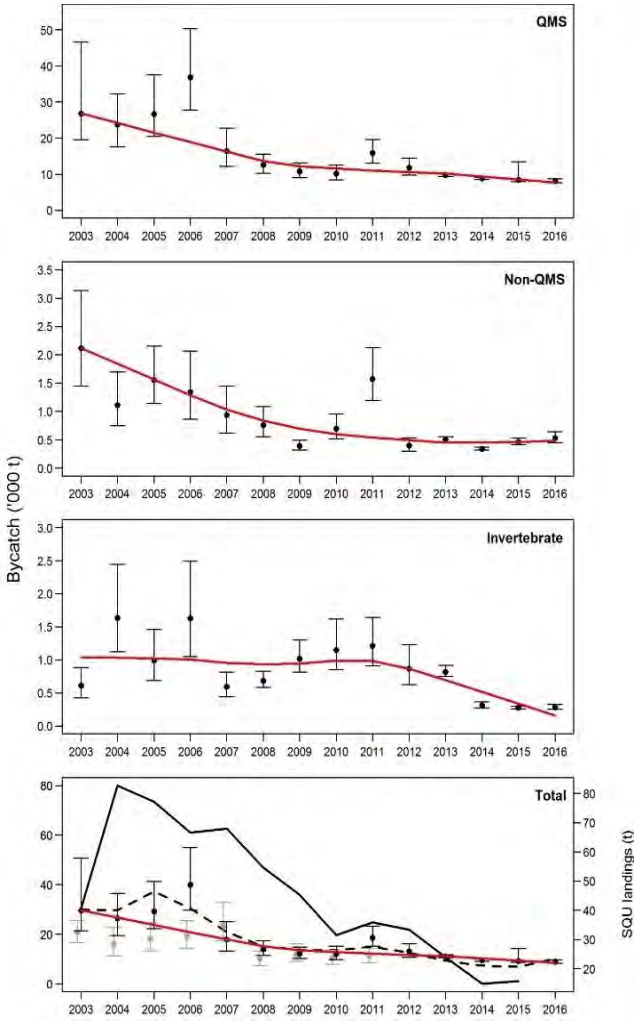


Figure 9.23: Annual estimates of bycatch in the arrow squid trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16 (Anderson & Edwards, 2018). Also shown (in grey) are estimates of total bycatch calculated for 2002–03 to 2010–11 (Anderson 2013a). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported trawl-caught landings of arrow squid, and the dashed line shows annual effort (scaled to have mean equal to that of total bycatch).

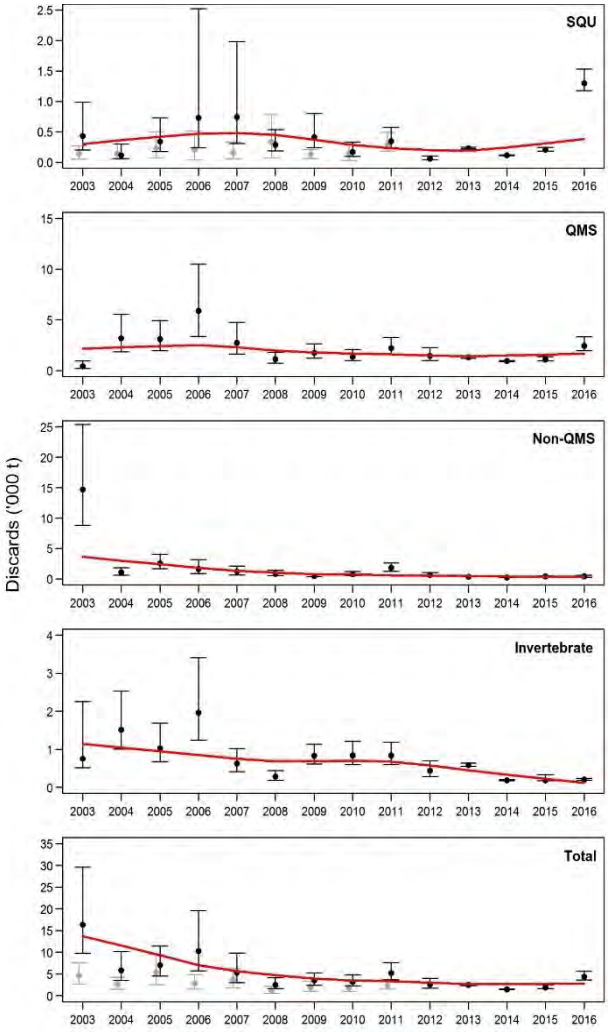


Figure 9.24: Annual estimates of discards in the arrow squid trawl fishery, for arrow squid (SQU), QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16 (Anderson & Edwards, 2018). Also shown (in grey) are estimates of arrow squid and total discards calculated for 2002–03 to 2010–11 (Anderson 2013a). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

.

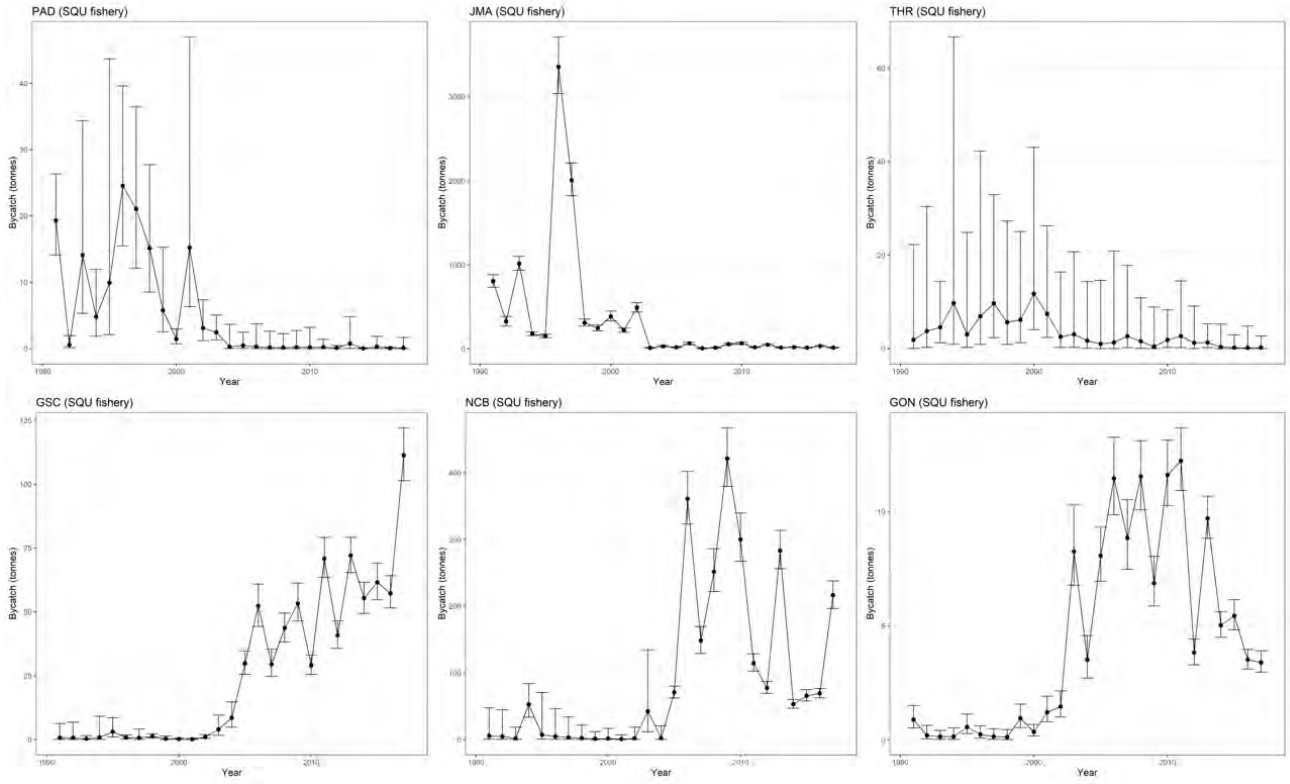


Figure 9.25: Annual bycatch estimates in the target arrow squid trawl fishery for the species that have the most bycatch between 1990-91 and 2016–17, with 95% c.i.s, in descending order of total catch (Finucci et al. in prep). See text above or http://marlin.niwa.co.nz for species code definitions. Note: the scale changes on the y-axis between plots.

9.3.13 HOKI/HAKE/LING TRAWL FISHERY

Earlier analyses were limited to the hoki target fishery but were subsequently expanded to cover bycatch and discards from hoki, hake and ling target fisheries combined, and most recently included silver warehou and white warehou in the definition of the target fishery; hoki nevertheless dominates this fishery, accounting for over 90% of the catch (Anderson et al. in prep). The most recent bycatch and discard report for this fishery included a more rigorous analysis of data, with the use of additional covariates (gear type and vessel class) (Anderson et al. in prep). Between 2002–03 and 2016–17, observer sampling levels have been highest in the west coast South Island, the Sub-Antarctic areas, and Puysegur, with lower levels in the Chatham Rise. These areas comprise the majority of the fishery in any year; little sampling has occurred outside of these main fishery areas. Observer coverage was spread throughout the year, with modest increase in effort during the hoki spawning season (June to early September), meaning that the spawning period was relatively undersampled and off-season oversampled in many years. Observer effort was found to be more closely matched to overall effort in some years between 2010–11 and 2015–16, which is likely to be due to coverage requirements for foreign charter vessels.

Hoki, hake and ling accounted for 85% (73%, 6.7% and 5.2%, respectively) of the total observed catch from trawls targeting hoki, hake, and ling between 2002–03 and 2016–17. The remaining 15% comprised a large range of species, in particular silver warehou (3.9%), javelinfish (1.9%), rattails (1.6%), spiny dogfish (1.4%), and white warehou (1.3%) (Figure 9.26). In total, over 800 species or species groups were identified by observers, the majority of these species were non-QMS species caught in low numbers. Chondrichthyans in general, often unspecified but including shovelnose dogfish and Baxters dogfish, accounted for much of the non-commercial catch. Echinoderms, squids, crustaceans, and other unidentified invertebrates were also well represented in the bycatch of this fishery.

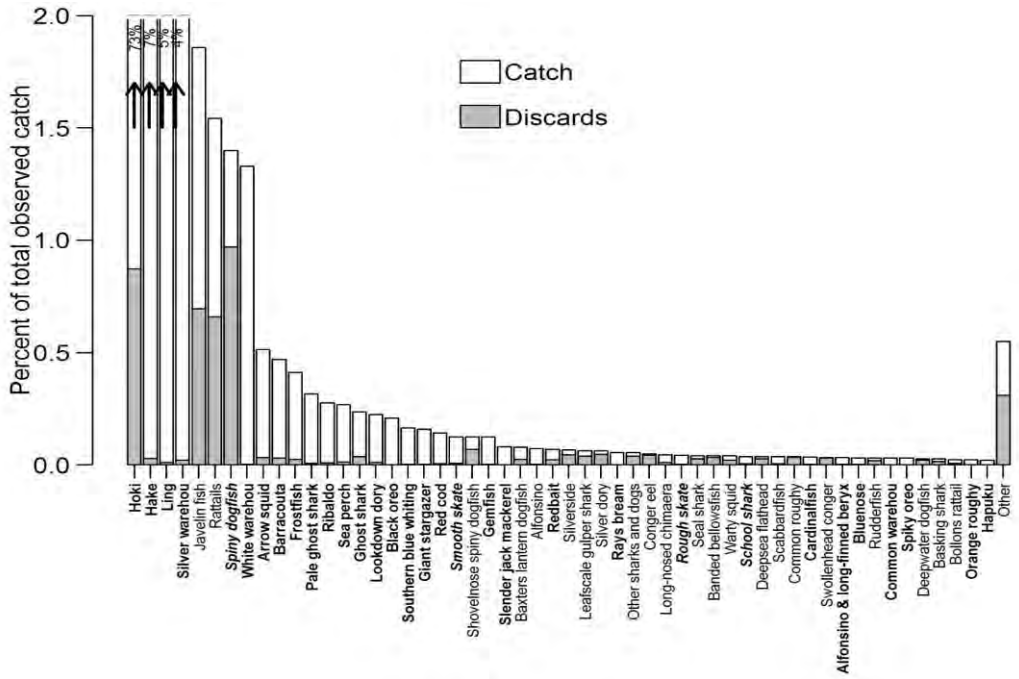


Figure 9.26: Percentage of the total catch contributed by the main bycatch species (those representing 0.05% or more of the total catch; white) in the observed portion of the hoki/hake/ling fishery (1990–91 to 2012–13), and the percentage discarded (grey; Anderson et al. in prep). QMS species are shown in bold.

Total bycatch in the hoki, hake, and ling fishery between 1990–91 and 2016–17 was 18 000–50 000 t per year (compared to the combined total landed catch of hoki, hake, and ling of about 100 000–300 000 t) (Anderson et al. in prep). Overall, total bycatch increased during the 1990s to a peak in the early 2000s, then declined slowly. Annual bycatch for the 1990–01 to 2016–17 period was also estimated for QMS species, non-QMS species, and invertebrates. Roughly similar amounts of QMS species and non-QMS species were caught overall, and each showed a similar pattern over time to total bycatch; invertebrate catch was less than 1000 t in most years, but peaked at about 1800 t in 2001–02 (Figure 9.27).

Total annual discard estimates for 1990–91 to 2016–17 were 5000–25 000 t per year with the main species observed discarded including spiny dogfish, rattails, javelinfish, and hoki (Anderson et al. in prep). Estimated annual discards of the target species combined ranged from 76–2340 t per year, with increasing levels since 2007–08. Estimates of total annual discards were variable but have generally declined over time, from 25 000 t in 2002–03 to 5000–8000 t in the most recent five years of the time series (Figure 9.28). Discard rates have been shown to be strongly influenced by the use of fishmeal plants on fishing vessels; with discards of non-commercial species on factory vessels without meal plants up to twice the level of discards for vessels with meal plants (Ballara et al. 2010). Vessel class is also very influential, with greater rates of discarding attributed to FCV (or former FCV) vessels compared with other vessel types.

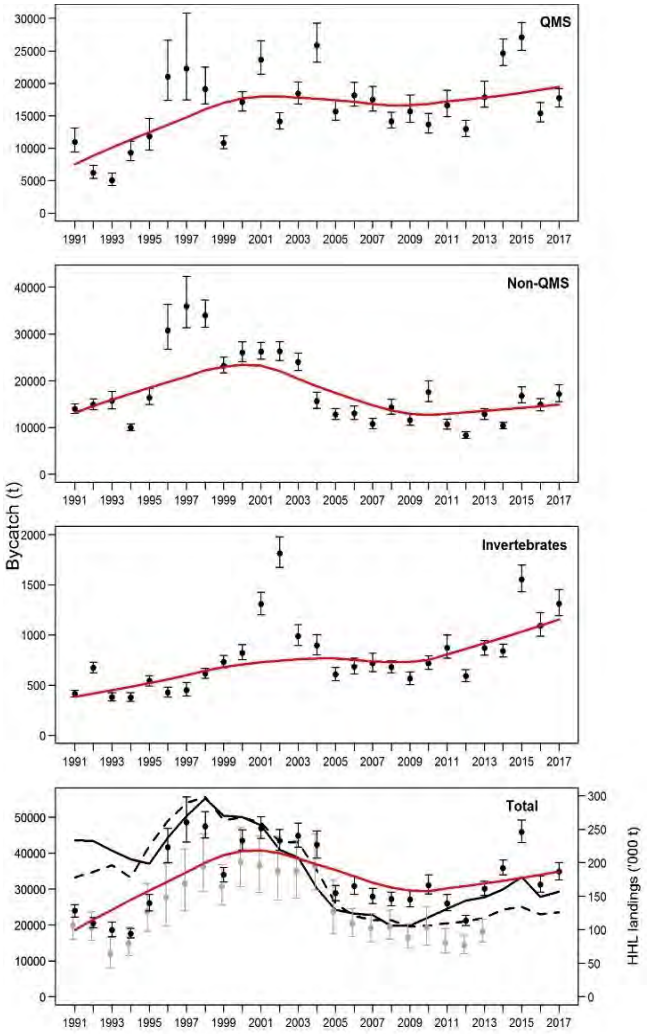


Figure 9.27: Annual estimates of fish bycatch in the target hoki, hake and ling trawl fishery, calculated for QMS species, non-QMS species, invertebrates, and overall for 1990–91 to 2016–17 (Anderson et al. in prep). Also shown (in grey) are earlier estimates of bycatch (Ballara & O'Driscoll, 2015). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported trawl-caught landings of hoki, hake, or ling and the dashed line shows annual effort (scaled to have mean equal to that of total bycatch).

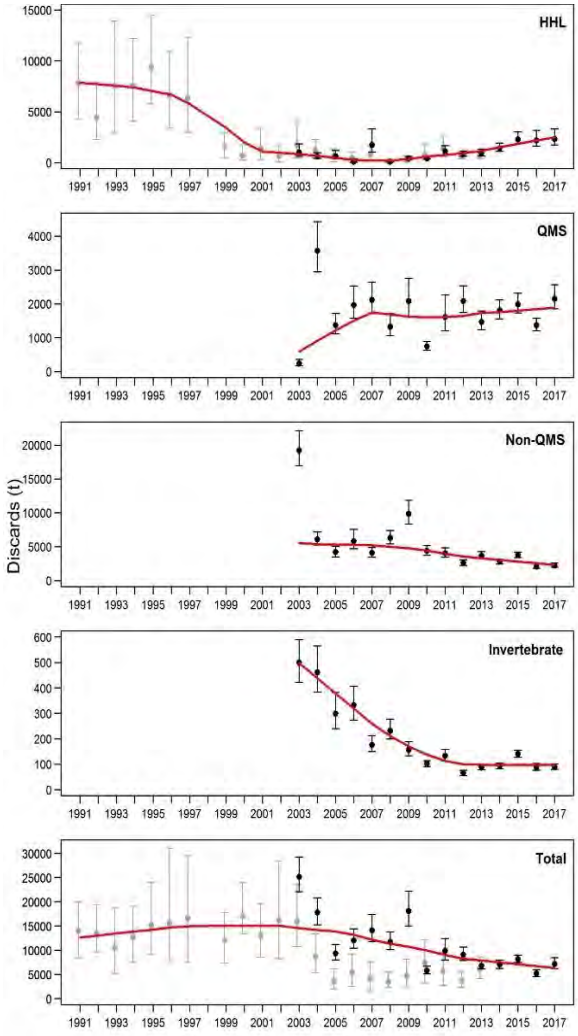


Figure 9.28: Annual estimates of discards in the target hoki, hake, ling, silver warehou or white warehou trawl fishery, by species category, for 2002–03 to 2016–17 (black dots) (Anderson et al. in prep). Also shown (in grey) are earlier estimates of arrow squid and total discards calculated for 1991–92 to 2016–17 (Ballara & O'Driscoll 2015). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally-weighted polynomial regression to annual discards

TRENDS IN BYCATCH BY SPECIES FROM THE HOKI, HAKE, AND LING TRAWL FISHERY

Finucci et al. (in prep) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most commonly caught bycatch species were javelinfish (*Lepidorhynchus denticulatus*, JAV), unspecified rattails (Macrouridae, RAT), and silver warehou (*Seriolella punctata*, SWA).
- Of the 493 bycatch species examined, 35 had a significant decrease in catch over time and 83 a significant increase in catch.
- The species showing the greatest decline were unspecified skates (SKA), lanternshark (*Etmopterus* spp., ETM), and moonfish (*Lampris guttatus*, MOO) (Figure 9.29). Notably SKA and ETM are generic

codes that have been replaced by more specific codes, which probably explains these declines.

- The species showing the greatest increase were umbrella octopus (*Opisthoteuthis* spp., OPI) Tam 'o Shanter urchins (Echinothuriidae & Phormosomatidae, TAM), and floppy tubular sponge (*Hyalascus* sp., HYA) (Figure 9.29).

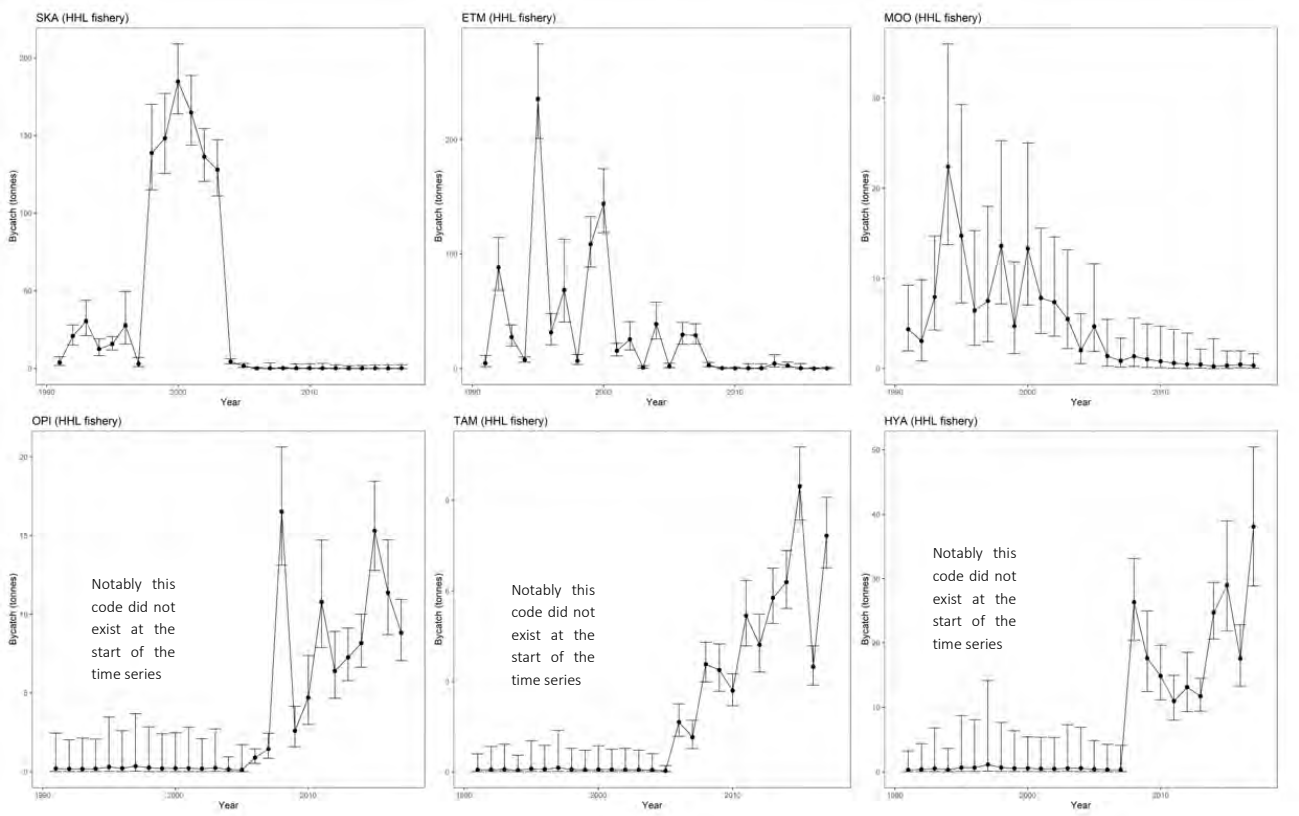


Figure 9.29: Annual bycatch estimates in the hoki, hake, and ling trawl fishery for the species which had the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2016–17 (Finucci et al. in prep). Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3.1), and may be area-specific (see text above). See text above for species codes.

9.3.14 JACK MACKEREL TRAWL FISHERY

Estimates of annual bycatch in this fishery are available for fishing years up to 2013–14, with the most recent analysis focusing on the 2002–03 to 2013–14 period (Anderson et al. 2017b). Both the ratio and the statistical model method of estimation were used in this analysis, with comparable results overall although the statistical model provided tighter confidence intervals. The annual level of observer coverage in this fishery was 8–30% of the target fishery catch before 2007–08 but rapidly increased to be 80–95% after 2010–11. This elevated level of coverage was due to a commitment by the Ministry to full observer coverage on foreign charter vessels, which have historically taken a large part of the catch in this fishery (Ministry for Primary Industries 2013b). Observer effort in each year has generally been focused on the main fishery, off the west coasts of the North and South Islands, with some additional coverage on the Stewart/Snares Shelf and Chatham Rise fisheries. This was variable, however, and in 2003–04 and 2004–05 there were only 12 trawls observed outside of the western fishery (notably since 2002–03 over 90% of the effort in this fishery has been in the west coast fisheries). The fishery occurs mostly in October–February and April–August, and although there were lengthy periods in some years when commercial fishing effort was not observed, coverage has been well matched to the main fishing periods for all years combined.

Jack mackerel species comprised 75% of the total observed catch from all trawls targeting jack mackerel from 2002–03 to 2012–13. The remaining 30% mostly comprised other QMS species; especially barracouta (13%), blue mackerel (3.4%), frostfish (3.4%), and redbait (2.5%) (Figure 9.30). Overall about 320 species or species groups were identified by observers during this period, many of which were non-QMS species caught in small numbers. The species most discarded was the spiny dogfish[1] (which entered the QMS in October 2004), comprising about 0.3% of the total catch. Of the invertebrates, only molluscs (mostly arrow squid) were observed caught in substantial amounts (about 500 t) and these were mostly retained. Lesser amounts of cnidarians, sponges, and echinoderms were observed caught (about 13 t in total), and almost all were discarded.

Total bycatch in the jack mackerel trawl fishery from 2002–03 to 2013–14 was 6600–18 800 t. Estimates of total bycatch for 1990–91 to 2003–04 from earlier projects were 5400–15 500 t. Bycatch has mainly comprised QMS species, especially in more recent years (Figure 9.31).

Total annual discards varied between about 80 t and 400 t between 2002–03 and 2013–14, with no trend during this time (Anderson et al. 2017b). This compares with a much higher level in 1997–98 (1850 t), when annual discards were at their greatest. Annual discards of the target species were about 200–400 t prior to 1998–99 but thereafter decreased to about 10 t, mainly due to the absence of recorded losses of large quantities of fish through rips in the net or intentional releases of fish during landing. Discards comprised roughly equal amounts of QMS and non-QMS, although the ratio varied considerably from year to year (Figure 9.32).

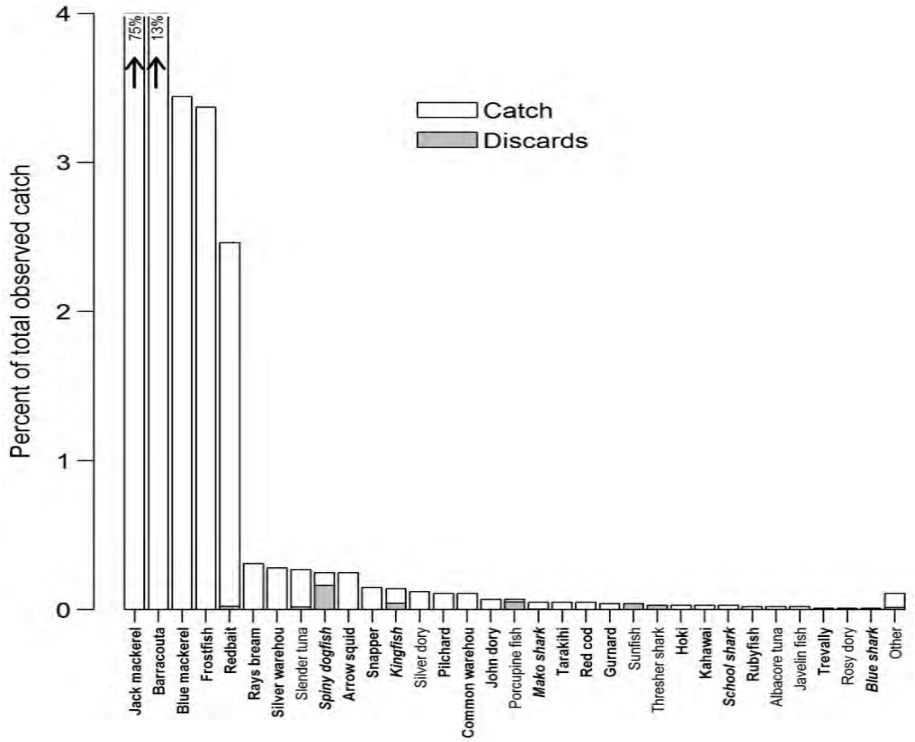


Figure 9.30: Percentage of the total catch contributed by the main bycatch species (those representing 0.01% or more of the total catch; white) in the observed portion of the jack mackerel trawl fishery between 2002–03 and 2013–14, and the percentage discarded (grey; Anderson et al. 2017b). The 'Other' category is the sum of all bycatch species representing less than 0.01% of the total catch. Names in bold are QMS species, names in italics are QMS species that can be legally discarded under Schedule 6 of the Fisheries Act (1996).

[1] Notably it is legal to discard spiny dogfish under Schedule 6 of the Fisheries Act.

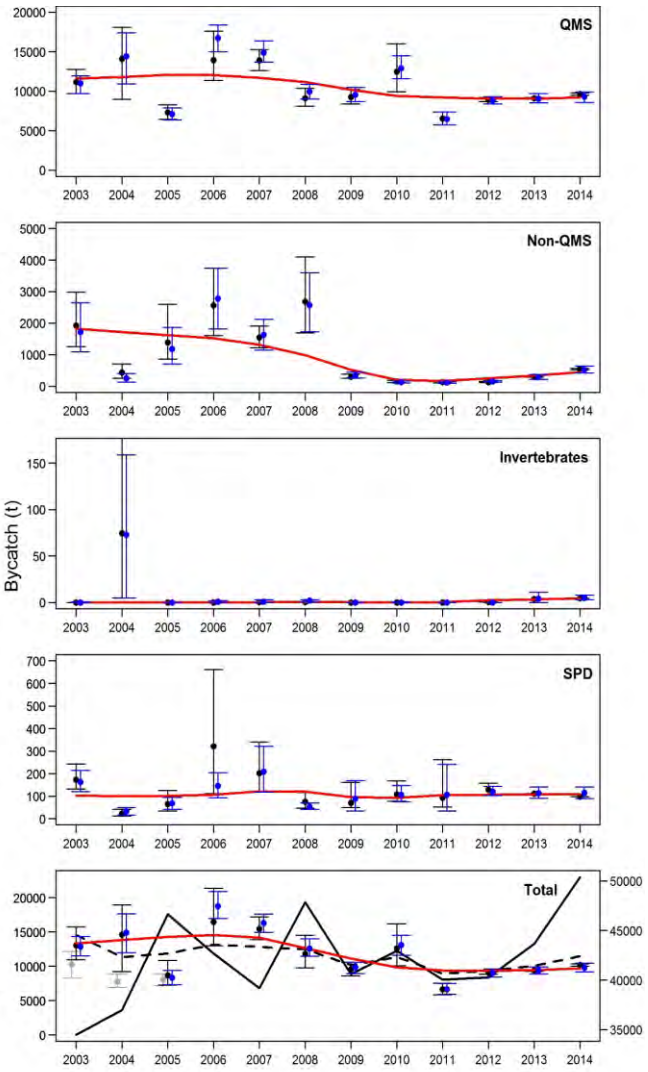


Figure 9.31: Annual estimates of bycatch in the jack mackerel trawl fishery, for QMS species, non-QMS species, invertebrates, and overall for 2002–03 to 2013–14: black dots, ratio method; blue dots, statistical model method (Anderson et al. 2017b). Also shown (in grey) are earlier estimates of total bycatch calculated for 2002–03 to 2004–05 (from Anderson 2007). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported landings of jack mackerels, and the dashed line shows annual effort (number of tows), scaled to have mean equal to that of total bycatch.

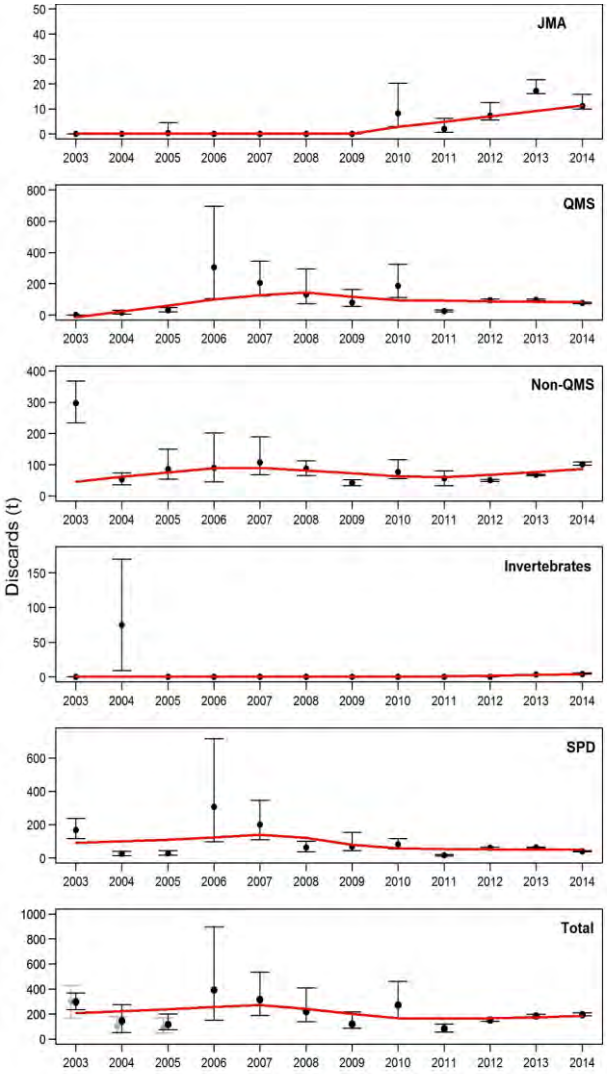


Figure 9.32: Annual estimates of discards in the jack mackerel trawl fishery, for jack mackerels (JMA), QMS species, non-QMS species, invertebrates, spiny dogfish (SPD), and overall (Total) for 2002–03 to 2013–14 (Anderson et al. 2017b). Also shown (in grey) are earlier estimates of total discards calculated for 2002–03 to 2004–05 (from Anderson 2007). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

TRENDS IN BYCATCH BY SPECIES FROM THE JACK MACKEREL TRAWL FISHERY

Finucci et al. (in prep) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most commonly caught bycatch species were barracouta (BAR), frostfish (*Lepidopus caudatus*, FRO), and blue mackerel (*Scomber australasicus*, EMA).
- Of the 163 bycatch species examined, 30 showed a significant decrease in catch over time and seven showed an increase.

- Species with significant declines included dark ghost shark (GSH), red cod (*Pseudophycis bachus,* RCO), and sea perch (*Helicolenus* spp., SPE), (Figure 9.33).
- Species showing significant increases included albacore tuna (*Thunnus alalunga*, ALB), pilchard (*Sardinops sagax*, PIL), and kingfish (*Seriola lalandi*, KIN) (Figure 9.33).

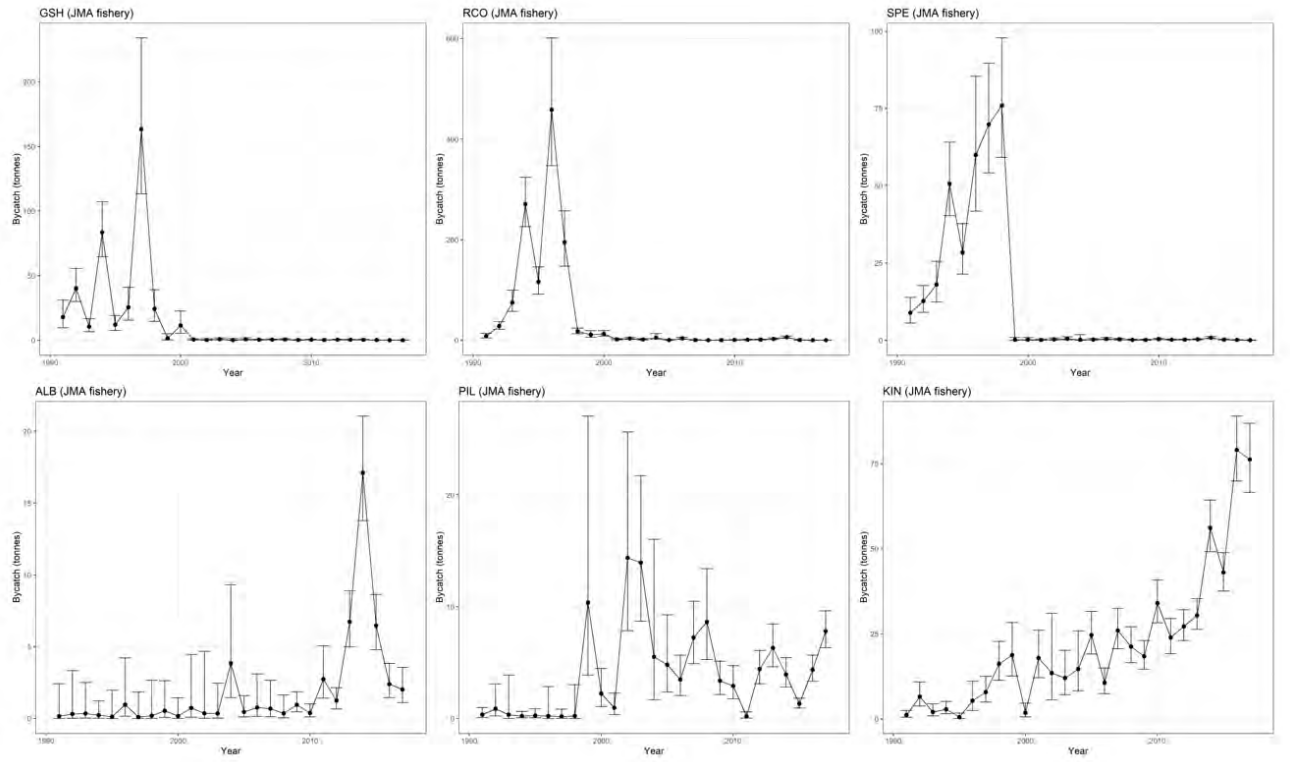


Figure 9.33: Annual bycatch estimates in the jack mackerel trawl fishery for the species which have shown the greatest decrease (top) and greatest increase (bottom) between 1990-91 and 2016-17 (Finucci et al. in prep). See text above for explanation of the species codes. Note: the scale changes on the y-axis between plots; lines are joined only where there are data points for consecutive years.

9.3.15 SOUTHERN BLUE WHITING TRAWL FISHERY

In a study that covered data from 2002–03 to 2006–07, the ratio estimator used to calculate bycatch and discard rates in this fishery was based on trawl duration (Anderson 2009b). Linear mixed-effect models (LMEs) identified fishing depth as the key variable influencing bycatch rates and discard rates in this fishery, and regression tree methods were used to optimise the number of levels of this variable in order to stratify the calculation of annual bycatch and discard totals in each catch category.

The key categories of catch/discards examined were; southern blue whiting, other QMS species combined, commercial species combined (as defined above for hoki/hake/ling), non-commercial species combined, and three commonly caught individual species, hake, hoki and ling.

The level of observer coverage represented was 22–53% of the target fishery catch from 2002–03 to 2006–07 and similar levels were reported from 1990–91 to 2001–02. The spread of observer data, across a range of variables, had no obvious shortcomings, due to a combination of the highly restricted distribution of the southern blue whiting fishery over space and time of year, a stable and uniform fleet composition, and a high level of observer effort.

Southern blue whiting were more than 99% of the total estimated catch from all observed trawls targeting southern blue whiting from 2002–03 to 2006–07. About half the remaining total catch was made up of ling (0.2%), hake (0.1%), and hoki (0.1%) (Figure 9.34). These three species, along with other QMS species, comprised over 80%

of the total bycatch. In all, over 120 species or species groups were identified by observers, most were non-commercial species caught in low numbers. Porbeagle sharks (introduced into the QMS in 2004), javelinfish and other rattails, and silverside, accounted for much of the remaining bycatch. Invertebrate species (mainly sponges, crabs, and echinoderms) were also recorded by observers, but no taxon accounted for more than 0.01% of the total observed catch.

Estimated total annual bycatch from 2002–03 to 2006–07 was 40–390 t, compared with approximate target species catches in the same period of about 22 000 to 42 000 t. This bycatch was split between commercial species (55%) and non-commercial species (45%), although QMS species accounted for about 80% of the total bycatch during this period. Total annual bycatch decreased during the period, to an all-time low of 40 t in 2006–07. Total annual bycatch estimates for 1990–91 to 2001–02, from earlier reports, were mostly 60–500 t but reached nearly 1500 t in 1991–92 (Figure 9.35). This year immediately preceded the introduction of southern blue whiting into the QMS, and the effort and catch were exceptionally high.

Total annual discard estimates from 2002–03 to 2006–07 were 90–250 t per year (Anderson 2009b). Discard amounts sometimes exceeded bycatch due to the large contribution of the target species (50–230 t per year) to total discards – the result usually of fish losses during recovery of the trawl. Discarding of commercial species was virtually non-existent in most years and discards of non-commercial species amounted to only 10–50 t per year. The main species discarded were southern blue whiting, rattails and porbeagle sharks. Total annual discard estimates for 1990–91 to 2001–02, from earlier reports, were mostly 140–750 t but were about 1200 t in 1991–92 (Figure 9.36). Discards of southern blue whiting (and therefore total discards) decreased substantially at the end of the 1990s and remained at low levels, below 250 t per year, up to 2006–07.

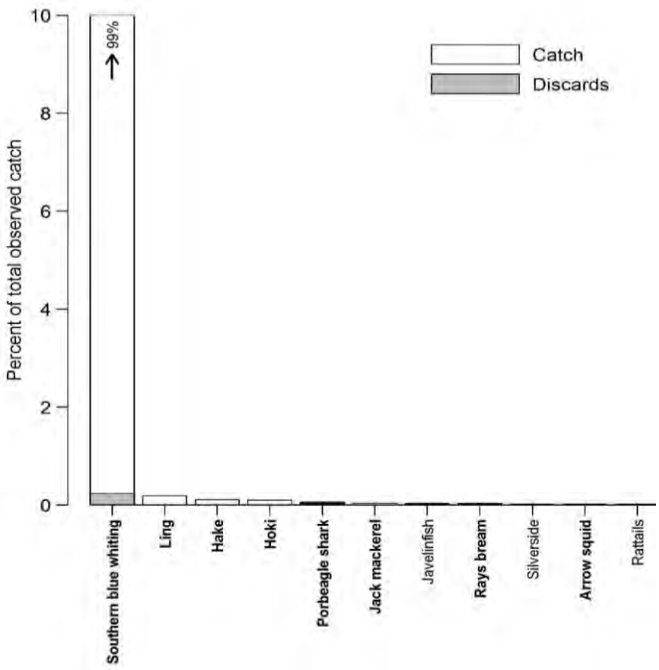


Figure 9.34: Percentage of the total catch contributed by the main bycatch species (those representing 0.05% or more of the total catch; white) in the observed portion of the southern blue whiting fishery, 2002–03 to 2006–07, and the percentage discarded (grey; Anderson 2009b). QMS species are shown in bold.

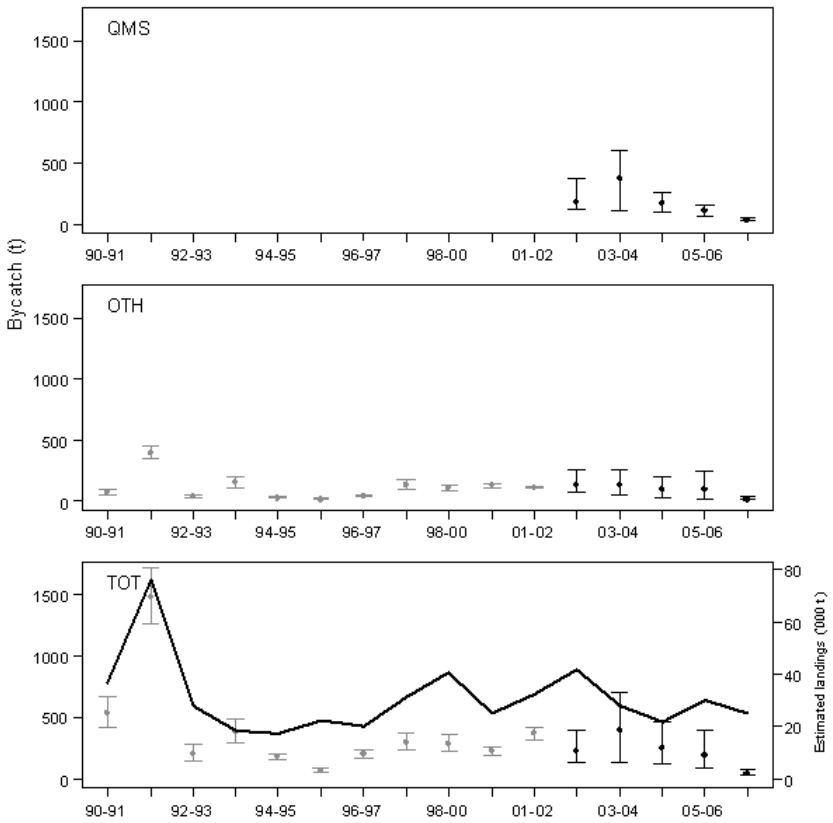


Figure 9.35: Annual estimates of fish bycatch in the southern blue whiting trawl fishery, calculated for QMS species, non-commercial species (OTH), and overall (TOT) for 2002–03 to 2006–07 (in black) (Anderson 2009b). Also shown (in grey) are estimates of bycatch in each category (excluding QMS) for 1990–91 to 2001–02 (Anderson 2004a). Error bars show the 95% confidence intervals. Note: the 98–00 fishing year encompasses the 18 months between September 1998 and March 2000, the transitional period during a change from an Oct–Sep to Apr–Mar fishing year. The dark line in the bottom panel shows the total annual estimated landings of SBW (Ministry for Primary Industries 2013a).

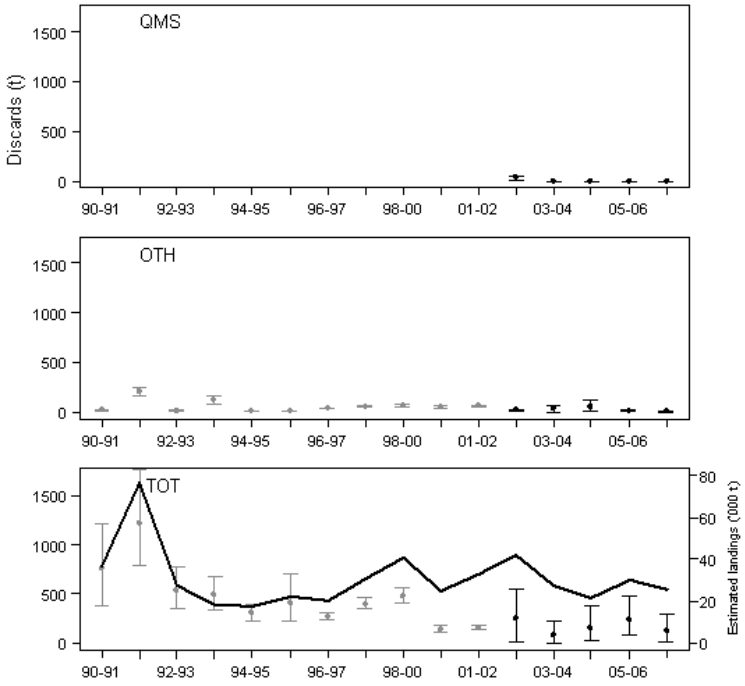


Figure 9.36: Annual estimates of fish discards in the southern blue whiting trawl fishery, calculated for the target species (SBW), QMS species, non-commercial species (OTH), and overall (TOT) for 2002–03 to 2006–07 (in black) (Anderson 2009b). Also shown (in grey) are estimates of discards in each category (excluding QMS) calculated for 1990–91 to 2001–02 by Anderson (2004a). Error bars show the 95% confidence intervals. The dark line shows the total annual estimated landings of SBW (Ministry for Primary Industries 2013a).

TRENDS IN BYCATCH BY SPECIES FROM THE SOUTHERN BLUE WHITING TRAWL FISHERY

Finucci et al. (in prep) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most commonly caught bycatch species were ling (*Genypterus blacodes*, LIN), hake (*Merluccius australis*, HAK), and hoki (*Macruronus novaezelandiae*, HOK).
- Of the 109 bycatch species examined, six had a significant decrease in catch over time and one had a significant increase in catch.
- The species showing the greatest decline were moonfish (*Lampris guttatus*, MOO), unspecificed rattails (RAT), and dark ghost shark (*Hydrolagus novaezealandiae*, GSH) (Figure 9.37).
- The species showing the greatest increase were opah (*Lampris immaculatus,* PAH), ray's bream (*Brama brama*, RBM), and pale ghost shark (*Hydrolagus bemisi*, GSP) (Figure 9.37).

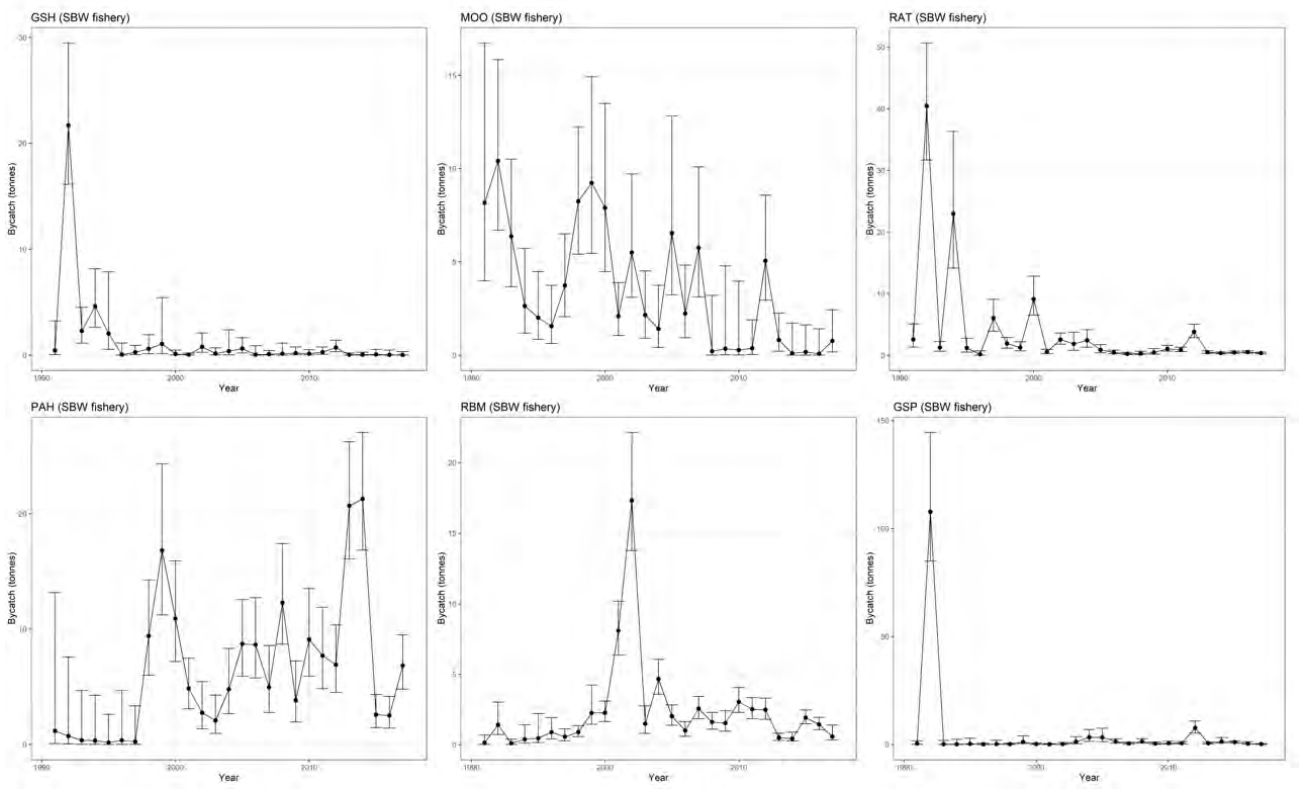


Figure 9.37: Annual bycatch estimates in the southern blue whiting trawl fishery for the species that had the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2016–17 (Finucci et al. in prep). Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3.1), and may be area-specific (see text above). See text above for species codes.

9.3.16 ORANGE ROUGHY TRAWL FISHERY

The most recent analysis of this fishery covered the period 2001–02 to 2014–15, and used both the ratio estimator and the statistical model method (Anderson et al. 2017a).

The key categories of catch/discards examined were; orange roughy, other QMS species combined, non-QMS species combined, and invertebrate species combined.

The level of observer coverage in this fishery since 1990–91 has been over 10% of the total fishery catch in all but one year, and over 50% in some years; between 2001–02 and 2014–15 coverage averaged 37% and was over 50% in five years. This coverage was relatively well spread across the orange roughy fishery (which is the most widespread New Zealand fishery). Some undersampling occurred of smaller vessels, on the east coast fisheries in QMAs ORH 2A, ORH 2B and ORH 3A (where only small vessels are allowed to operate), and oversampling occurred of fisheries outside of

the EEZ (where vessels are normally required to carry an observer).

Since 2001–02, orange roughy has comprised about 85% of the total observed catch. Much of the remainder of the total catch (about 9%) comprised oreo species: mainly smooth oreo (7%), and black oreo (1.6%). Rattails (various species, 0.7%) and shovelnose spiny dogfish (*Deania calcea*, 0.6%) were the species most caught by this fishery, with over 50% discarded (Figure 9.38). Other fish species frequently caught and usually discarded included deepwater dogfishes (family Squalidae), especially *Etmopterus* species, the most common was probably Baxter's dogfish (*Etmoptertus baxteri* – which is the most common species in the *Etmopterus* genus), slickheads, and morid cods, especially Johnson's cod (*Halargyreus johnsonii*) and ribaldo. In total, over 700 bycatch species or species groups were observed, most were non-commercial species, including invertebrate species, caught in low numbers. Squid (mostly warty squid, *Onykia* spp.) were the largest component of invertebrate catch, followed by various groups of protected corals, echinoderms (mainly starfish), and crustaceans (mainly king crabs, family Lithodidae).

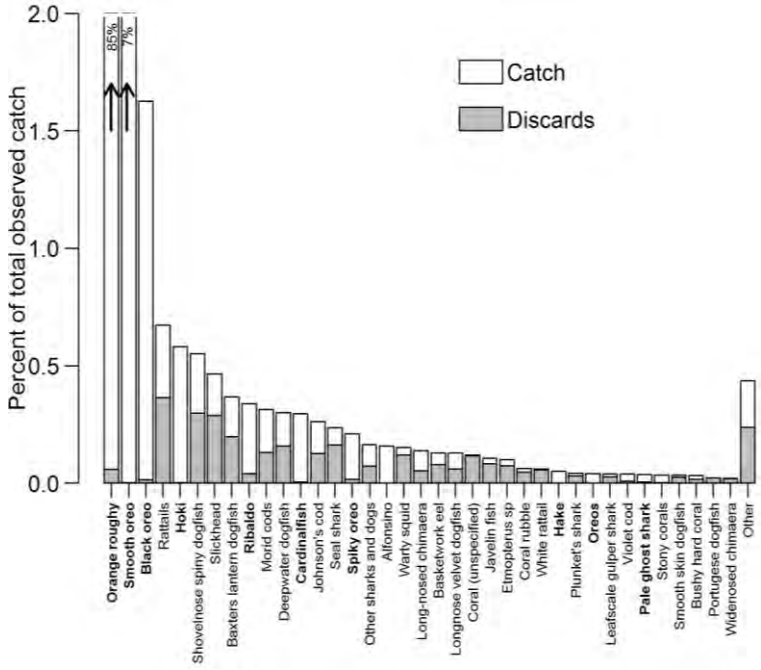


Figure 9.38: Percentage of the total catch contributed by the main bycatch species (those representing 0.02% or more of the total catch; white) in the observed portion of the target orange roughy trawl fishery for fishing years from 2001–02 to 2014–15, and the percentage discarded (grey; Anderson et al. 2017a). The 'Other' category is the sum of all bycatch species representing less than 0.02% of the total catch. Names in bold are QMS species, names in italics are QMS species that can be legally discarded under Schedule 6 of the Fisheries Act (1996).

Total annual bycatch in the orange roughy fishery since 2001–02 was highly variable, with greater levels (3093–6075 t per year) before 2009–10, and decreasing levels thereafter (706–1080 t per year), in line with decreasing orange roughy landings (Figure 9.39). Bycatch comprised similar amounts of QMS and non-QMS species, with invertebrate species bycatch below 200 t in most years and below 50 t since 2010–11.

Estimated total annual discards also decreased over time, from about 3400 t in 1990–91 (Anderson 2011) to less than 500 t since 2007–08 (Figure 9.40). Since about 2000, discards have comprised mostly non-QMS species. Large discards of orange roughy and other QMS species, more prevalent early in the fishery, were often due to fish lost from torn nets during hauling (and are accounted for in stock assessments). Improved fishing techniques and gear have substantially lowered the level of discards in these categories in more recent times.

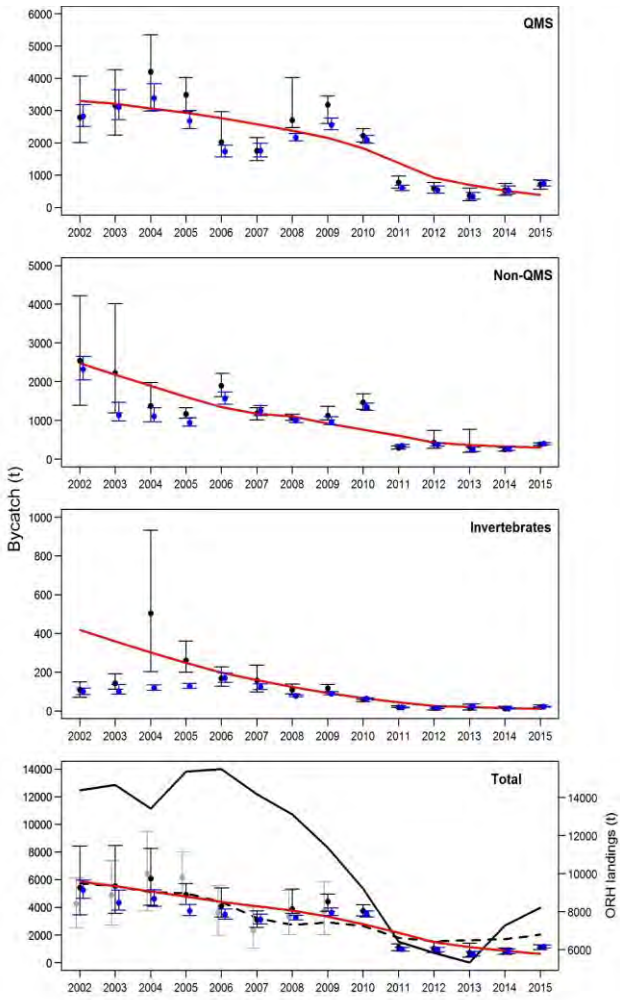


Figure 9.39: Annual estimates of bycatch (t) in the target orange roughy trawl fishery, species categories for 2001–02 to 2014–15: black dots, ratio method; blue dots, statistical model method (Anderson et al. 2017a). Also shown (in grey) are earlier estimates of total bycatch calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported landings of orange roughy, and the dashed line shows annual effort (number of tows), scaled to have mean equal to that of total bycatch.

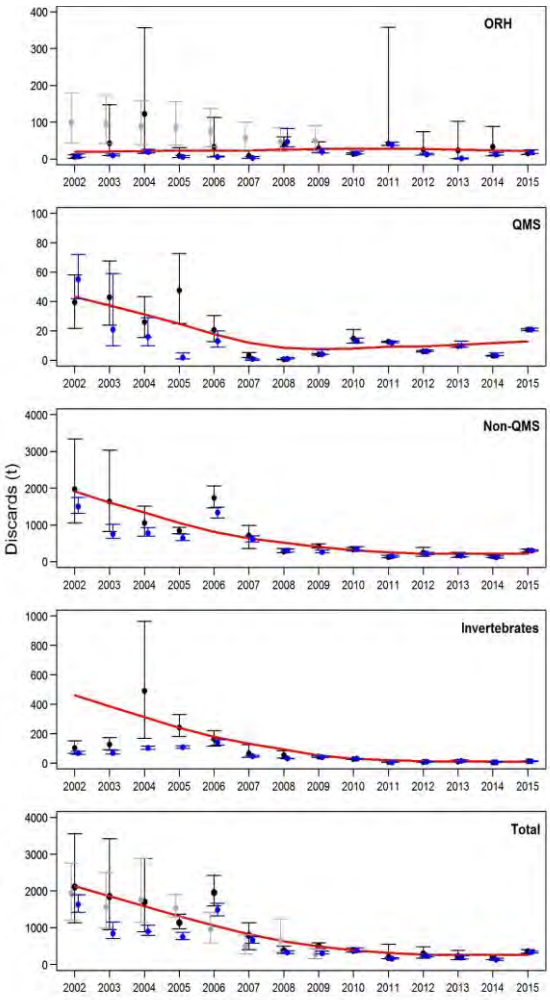


Figure 9.40: Annual estimates of discards (t) in the target orange roughy trawl fishery, for species categories for 2001–02 to 2014–15 (Anderson et al. 2017a). Also shown (in grey) are earlier estimates of total discards calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

TRENDS IN BYCATCH BY SPECIES FROM THE ORANGE ROUGHY TRAWL FISHERY

Finucci et al. (in prep) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most commonly caught bycatch species were smooth oreo (*Pseudocyttus maculatus*, SSO), black oreo (*Allocyttus niger*, BOE), and unspecified sharks (SHA).
- Of the 557 bycatch species examined, 29 showed a significant decrease in catch over time and 14 showed a significant increase in catch.
- The species showing the greatest decline were dark ghost shark (*Hydrolagus novaezealandiae*, GSH), black oreo (*Allocyttus niger*, BOE), and lanternshark (*Etmopterus* sp., ETM) (Figure 9.41).
- The species showing the greatest increase were longnose velvet dogfish (*Centroscymnus crepidater*, CYP), Portuguese dogfish (*Centroscymnus coelolepis*, CYL), and Owston's dogfish (*Centroscymnus owstonii*, CYO) (Figure 9.41).

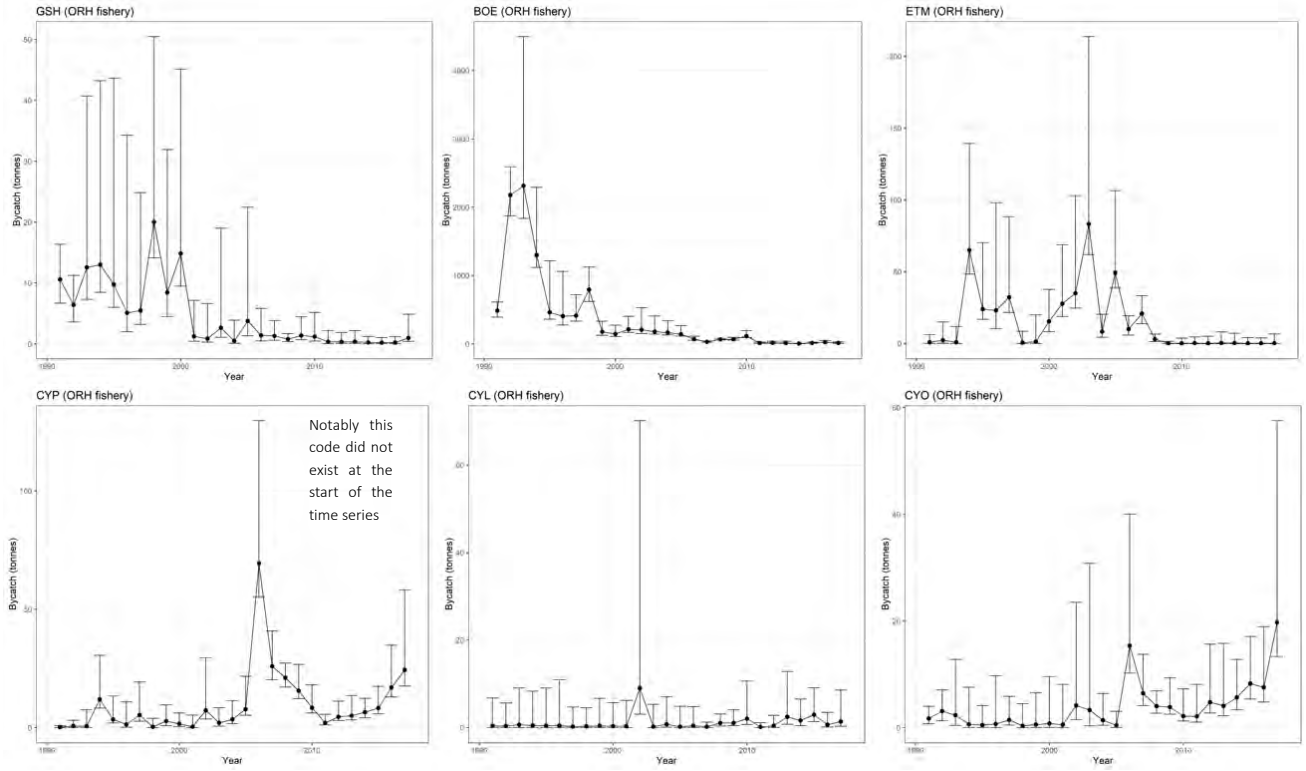


Figure 9.41: Annual bycatch estimates in the orange roughy trawl fishery for the species that have shown the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2016–17 (Finucci et al. in prep). See text above for explanation of the species codes. Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3.1). Note: the scale changes on the y-axis between plots; lines are joined only where there are data points for consecutive years.

9.3.17 OREO TRAWL FISHERY

The most recent analysis of this fishery covered the period 2001–02 to 2014–15, and used both the ratio estimator and the statistical model method (Anderson et al. 2017a).

The key categories of catch/discards examined were; orange roughy, other QMS species combined, non-QMS species combined, and invertebrate species combined.

The oreo fishery is strongly linked to the historically larger and more widespread orange roughy fishery, with an earlier study showing that about a third of observed trawls targeting oreos were from trips that predominantly targeted orange roughy (Anderson 2011). The observer coverage of the oreo fishery is therefore partly determined by the operations of the orange roughy fishery.

The annual number of observed trawls in the oreo fishery ranged from 30 in 1991–92 to 1011 in 2006–07 and the number of vessels observed ranged from 2 to 12. The level

of coverage remained at a relatively consistent level after the mid-1990s, despite a decrease in the total catch and effort, but declined after 2009–10 to a level of about 140–210 tows per year between 2012–13 and 2014–15. As a fraction of the total catch, observer coverage has been over 12% since 1999–2000, and approached 50% in a few years in the mid-2000s. Observer coverage has been mostly restricted to the main fisheries on the South Chatham Rise and further south. Within this region, few locations were not covered by observers during the period most recently examined, although the South Chatham Rise was undersampled and some southern fisheries oversampled in a few years. The full range of vessel sizes (mainly between 300 t and 3000 t, median length per area ranged from 26 to 66 m) was covered by observers, although small vessels were under-represented and large vessels over-represented.

Oreo species accounted for about 95% of the total estimated catch from all observed trawls targeting oreos after 1 October 2001. Orange roughy (1.9%) was the main bycatch species, with no other species or group of species accounting for more than 0.6% of the total catch. Baxter's dogfish was the next most common bycatch species, followed by rattails (which were mainly discarded) and hoki (Figure 9.42). In total, over 500 species or species groups were identified by observers in the target fishery, including numerous invertebrates. Corals (accounting for about 0.1% of the total catch), squids, and echinoderms were the main invertebrate groups caught.

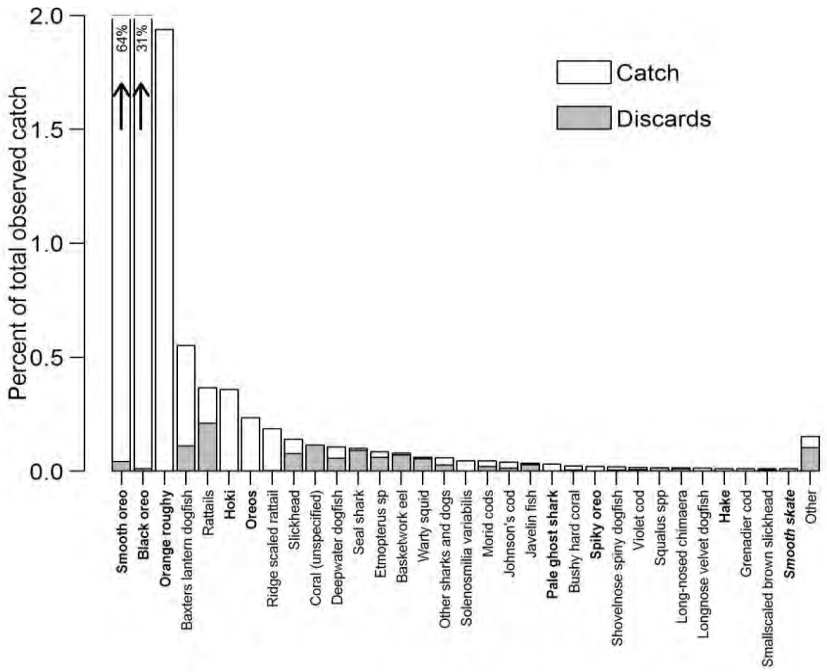


Figure 9.42: Percentage of the total catch contributed by the main bycatch species (those representing 0.01% or more of the total catch; white) in the observed portion of the oreo trawl fishery between 2001–02 and 2014–15, and the percentage discarded (grey; Anderson et al. 2017a). The 'Other' category is the sum of all bycatch species representing less than 0.01% of the total catch. Names in bold are QMS species, names in italics are QMS species that can be legally discarded under Schedule 6 of the Fisheries Act (1996)

Total bycatch in the oreo fishery has fluctuated in recent years with higher levels from 2001–02 to 2009–10 (range 579–1575 t per year), followed by lower levels from 2010–11 (352–535 t per year) (Figure 9.43). Bycatch was split almost evenly between commercial and non-commercial species overall, the ratio fluctuating without any trend over time.

Discards in the oreo fishery have slowly decreased over time, with the 14 t estimated for 2014–15 the lowest recorded for any year since 1990–91 (Figure 9.44). Discards mainly comprised non-QMS species, but included a varying amount of the target species in most years.

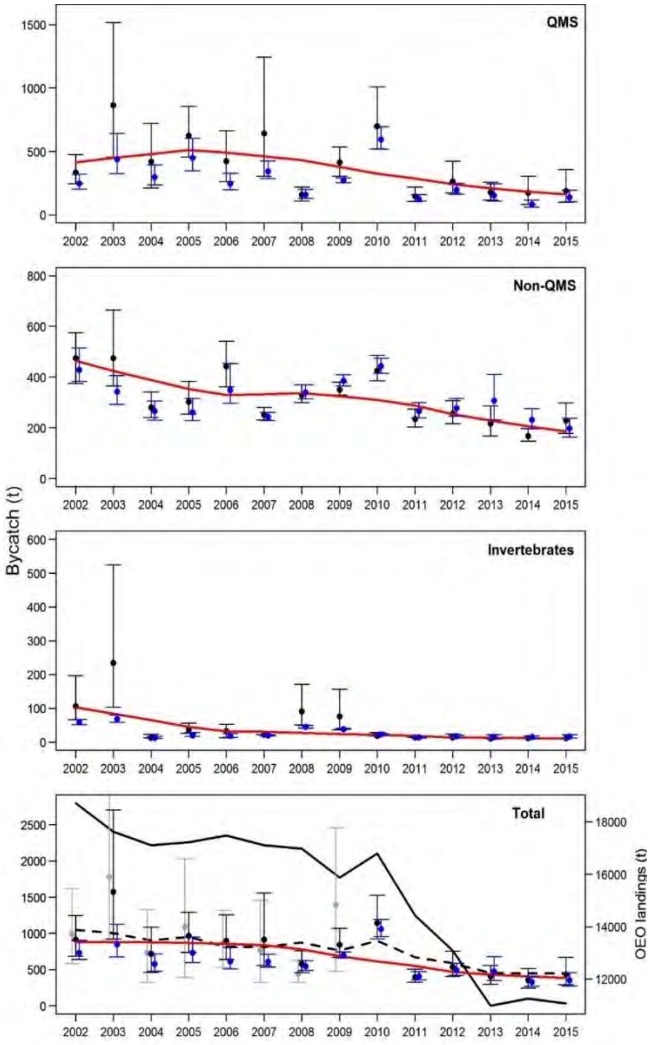


Figure 9.43: Annual estimates of bycatch in the oreo target trawl fishery for 2001–02 to 2014–15: black dots, ratio method; blue dots, statistical model method (Anderson et al. 2017a). Also shown (in grey) are earlier estimates of total bycatch calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported landings of oreos, and the dashed line shows annual effort (number of tows), scaled to have mean equal to that of total bycatch.

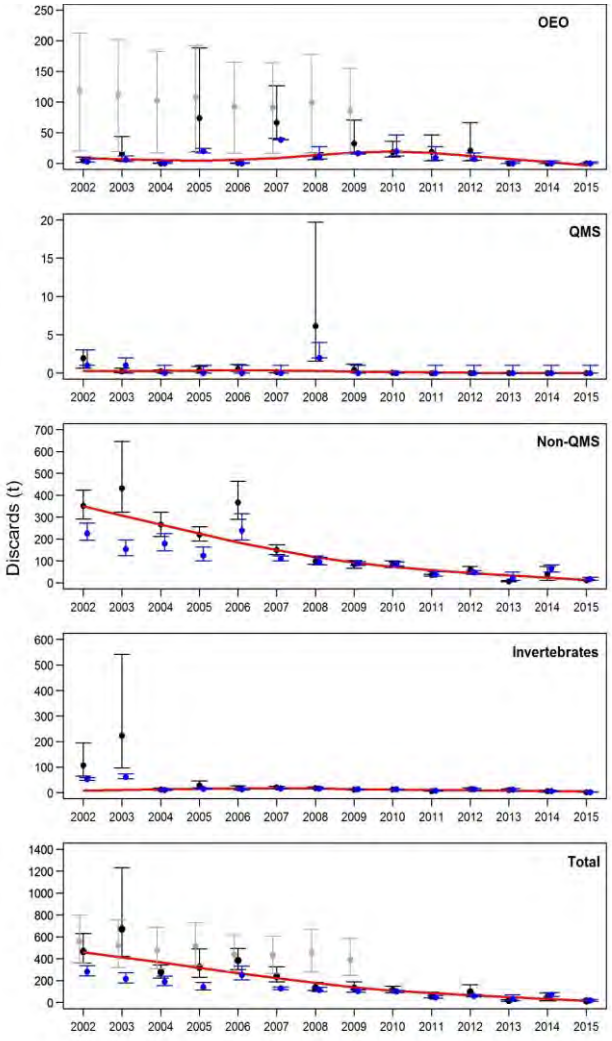


Figure 9.44: Annual estimates of discards in the oreo fishery, for species categories for 2001–02 to 2014–15 (Anderson et al. 2017a). Also shown (in grey) are earlier estimates of total discards calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

TRENDS IN BYCATCH BY SPECIES FROM THE OREO TRAWL FISHERY

Finucci et al. (in prep) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most commonly caught bycatch species were orange roughy (*Hoplostethus atlanticus*, ORH), unspecified sharks (SHA), and Baxter's dogfish (*Etmopterus granulosus*, ETB).
- Of the 228 bycatch species examined, 40 showed a significant decrease in catch over time and nine showed a significant increase in catch.
- The species showing the greatest decline were dark ghost shark (*Hydrolagus novaezealandiae*, GSH),

unspecified shark (SHA), [2] and lanternshark (*Etmopterus* spp., ETM) (Figure 9.45).

- The species showing the greatest increase were longnose velvet dogfish (*Centroselachus crepidater*, CYP), ridge scaled rattail (*Macrourus carinatus*, MCA), and Baxter's dogfish (*Etmopterus granulosus*, ETB) (Figure 9.45).

.

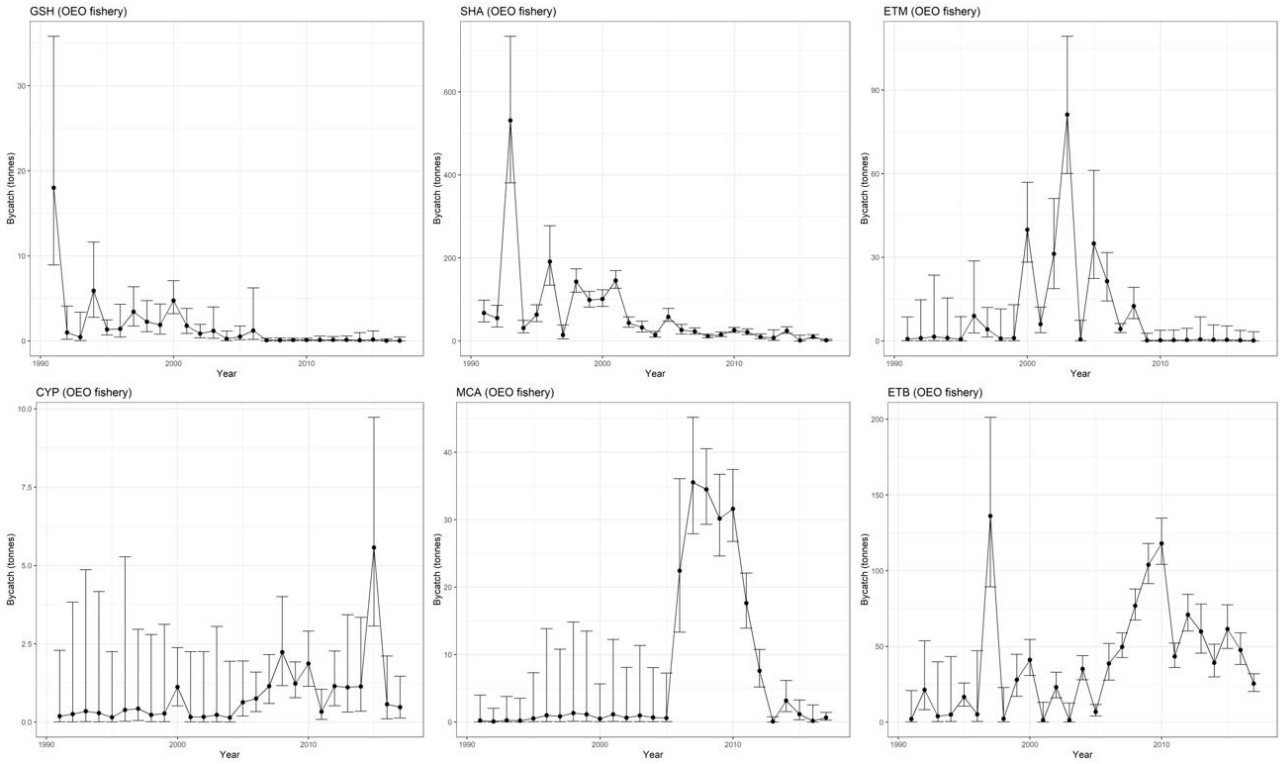


Figure 9.45: Annual bycatch estimates in the oreo trawl fishery for the species which have shown the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2016–17 (Finucci et al. in prep). See text above for explanation of the species codes. Sharks (SHA) may have been identified to an increasingly higher taxonomic level over time; rattails such as MCA may not have been well identified in earlier years. Note: the scale changes on the y-axis between plots; lines are joined only where there are data points for consecutive years.

9.3.18 SCAMPI TRAWL FISHERY

A detailed analysis of this fishery from 2002–03 to 2015–16 used the statistical model approach to calculate bycatch and discard levels in the scampi fishery, with effort based on the number of trawls (Anderson & Edwards, 2018).

The key categories of catch/discards examined were; all QMS species combined, all non-QMS species combined, and all invertebrate species combined, with membership of these categories adjusted from year to year as species were added to the QMS.

Observer coverage in the scampi fishery has been relatively low compared with most of the other deepwater fisheries assessed. The long-term level of observer coverage in the orange roughy, oreo, arrow squid, southern blue whiting and ling longline fisheries has covered more than 18% of the targeted catch (and over 40% for southern blue whiting) whereas in the scampi fishery (and also in the jack mackerel fishery) long-term coverage has been less than 12% of the targeted catch. For the 2002–03 to 2015–16 period most recently assessed, annual coverage was below 10% in 8 of the 14 years, reaching a maximum of 17% in 2002–03.

[2] Notably SHA is a generic code whose decline is probably due to better species level identification of sharks over time.

The annual number of observed trawls in the fishery ranged from 142 to 535, but was over 300 trawls in most years (2.7 to 15.5% of the total number of trawls). The number of vessels observed in each year ranged from 3 to 8 (equivalent to 33–75% of the fleet) and was relatively constant – 5 or 6 vessels in most years. Analysis of the spread of observer effort compared with that of the scampi fishery as a whole, across a range of variables, indicated that this coverage was reasonably well spread. Although some less important regions of the fishery received relatively low coverage (e.g., the eastern Chatham Rise, Puysegur, and west coast South Island), the main scampi fisheries were consistently sampled throughout the period examined. Vessels were mostly of a similar size, and the small amount of effort by larger vessels was adequately covered, as was the full depth range of the fishery and (despite highly intermittent sampling in several years) all periods of this year-round fishery.

Nearly 500 bycatch species or species groups were observed in the scampi target fishery catch, most being non-commercial species, including invertebrate species, caught in low numbers. Scampi accounted for about 19% of the total estimated catch from all observed trawls targeting scampi since 1 October 2002. The main bycatch species or species groups were javelinfish (18%), other (unidentified) rattails (12%), sea perch (*Helicolenus* spp., 10%), hoki (5%), and ling (4%). The first three of these bycatch groups were mostly discarded (Figure 9.46). Of the other invertebrate groups, unidentified crabs (0.9%) and unidentified starfish (0.8%) were caught in the greatest amounts. When combined into broader taxonomic groups, bony fish (excluding rattails and morid cods) contributed the most to total bycatch (33%), followed by rattails (30%), rays and skates (3.5%), sharks and dogfish (3.2%), chimaeras (3.1%), crustaceans (2.9%), echinoderms (1.6%), and morid cods (1.8%). A large percentage of the bycatch in these groups was discarded – over 80% for rattails, sharks, eels, crustaceans, echinoderms, octopuses, and other invertebrates.

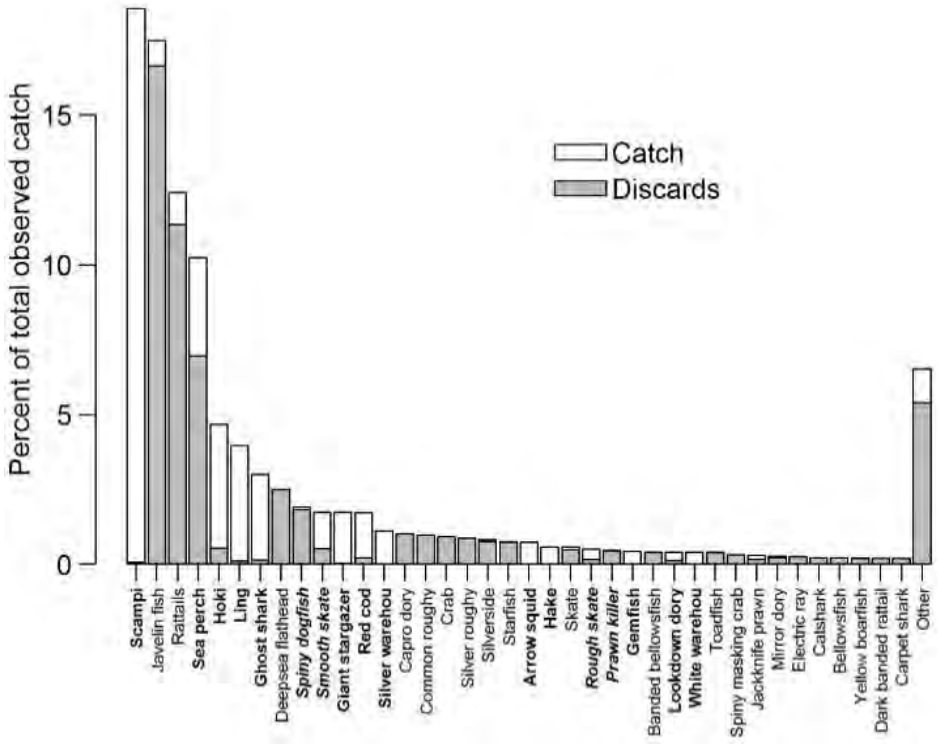


Figure 9.46: Percentage of the total catch contributed by the main bycatch species (those representing 0.2% or more of the total catch; white) in the observed portion of the scampi fishery, 2001–02 to 2015–16, and the percentage discarded (grey; Anderson & Edwards, 2018). The 'Other' category is the sum of all other bycatch species (fish and invertebrates) representing less than 0.2% of the total catch. QMS species are shown in bold, Schedule 6 species in italics.

.

Total annual bycatch since 2002–03 ranged from about 2400 t to 5600 t and, although highly variable in the early part of the period, showed no significant trend over time (Figure 9.47). Annual bycatch has overall been a relatively even mixture of QMS and non-QMS species, with invertebrate species accounting for only about 7% of the total bycatch for the whole period. Rattails (javelinfish and all other species combined) accounted for 45–95% of the annual non-QMS bycatch.

Total annual discards ranged from about 940 t in 2003–04 to about 4100 t in 2004–05 and, although quite variable from year to year, there was no significant trend in overall discard levels over time (Figure 9.48). Discards were dominated by non-QMS species (overall about 67%) followed by QMS species (24%) and invertebrates (9%). Rattail species accounted for about 75% of the non-QMS discards and about 50% of all discards.

TRENDS IN BYCATCH BY SPECIES FROM THE SCAMPI TRAWL FISHERY

Finucci et al. (in prep) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most commonly caught bycatch species were javelinfish (*Lepidorhynchus denticulatus*, JAV), unspecified rattails (Macrouridae, RAT), and sea perch (*Helicolenus* spp., SPE).
- Of the 332 bycatch species examined, 34 showed a significant decrease in catch over time while 58 showed a significant increase in catch.
- The species showing the greatest declines were skates (Rajidae and Arhynchobatidae, SKA), bluenose (*Hyperoglyphe antarctica*, BNS) and hapuku and bass (*Polyprion oxygeneios* & *P americanus*, HPB) (Figure 9.49).
- The species showing the greatest increases were geometric star (*Psilaster acuminatus*, PSI), smooth deepsea anemones (Actinostolidae, ACS) and Garrick's masking crab (*Leptomithrax garricki*, GMC) (Figure 9.49).

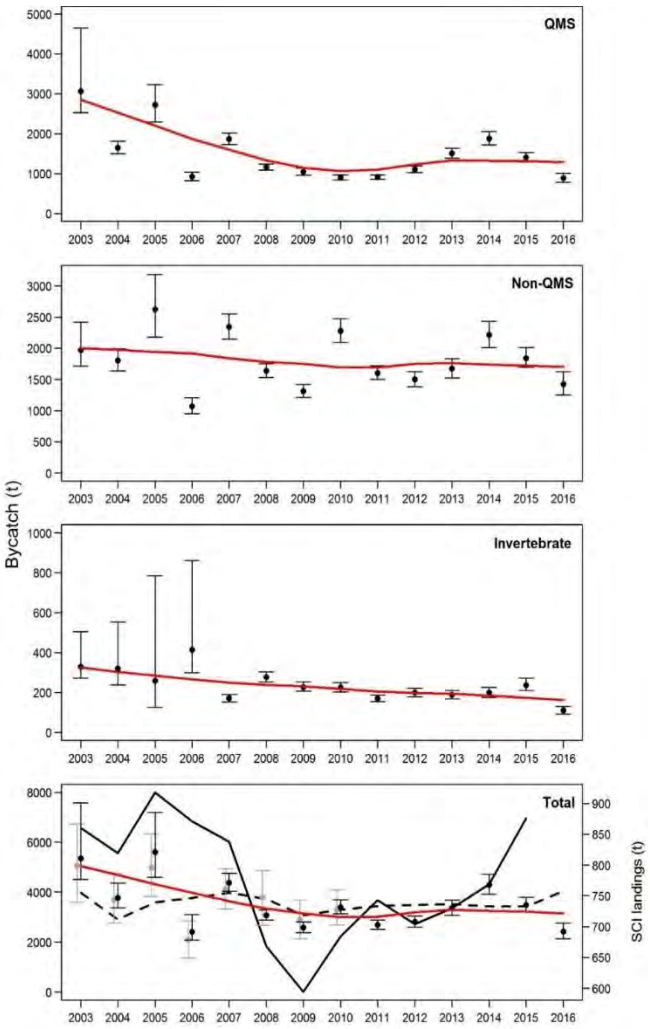


Figure 9.47: Annual estimates of bycatch in the scampi trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16 (Anderson & Edwards, 2018). Also shown (in grey) are estimates of Total bycatch calculated for 2002–03 to 2009–10 (Anderson 2012). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported landings of scampi and the dashed line shows annual effort (scaled to have mean equal to that of total bycatch).

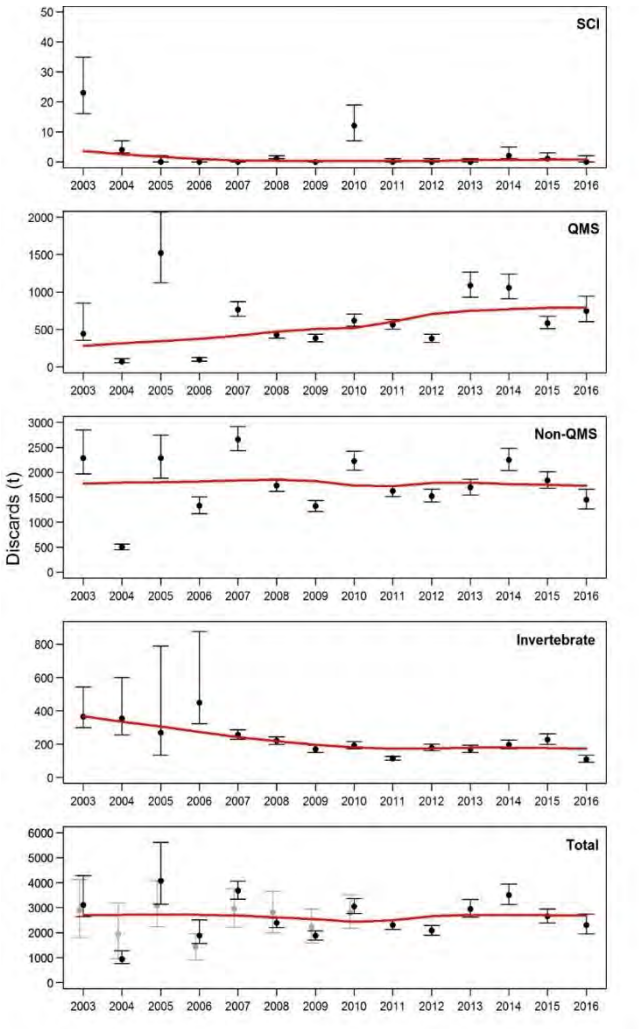


Figure 9.48: Annual estimates of discards in the scampi trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16. Also shown (in grey) are estimates of Total discards calculated for 2002–03 to 2009–10 (Anderson 2012). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted regression to annual discards.

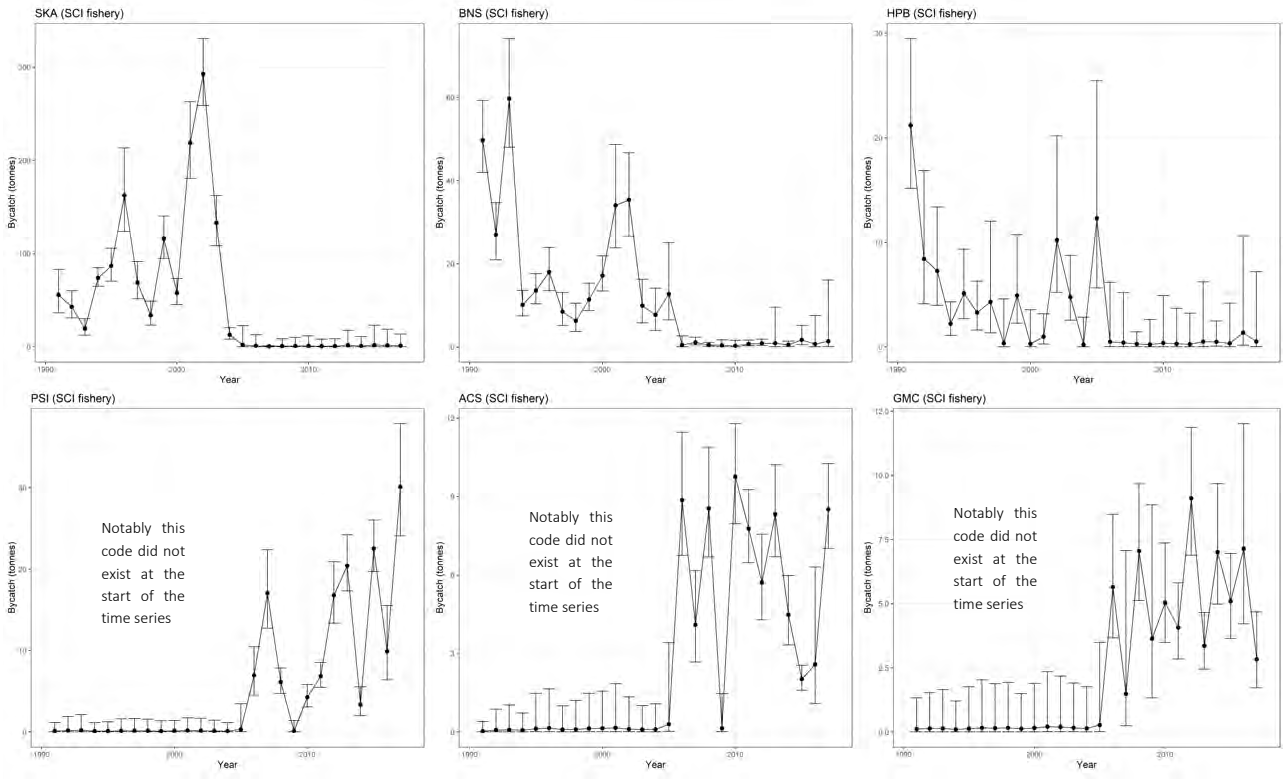


Figure 9.49: Annual bycatch estimates in the target scampi trawl fishery for the species which have the most bycatch between 1990–91 and 2016–17, with 95% c.i.s, in descending order of total catch. See text above or http://marlin.niwa.co.nz for species code definitions. Note: the scale changes on the y-axis between plots.

9.3.19 LING LONGLINE FISHERY

The first analysis of bycatch and discards in this fishery covered the period from 1990–91 to 1997–98 (Anderson et al. 2000), and a later analysis extended this to 2005–06 (Anderson 2008). The analysis was further updated in 2014, re-estimating annual bycatch and discards for all fishing years up to 2011–12, within the standard species categories (QMS, non-QMS, and Invertebrate) (Anderson 2014a).

The ratio estimator used in these analyses to calculate bycatch and discard rates was based on the number of hooks set. The ratios were applied to hook number totals calculated from commercial catch-effort data to make annual estimates for the target fishery as a whole.

Linear mixed-effect models (LMEs) were used to identify key factors influencing variability in the observed rates of bycatch and discarding, resulting in an area based stratification for the calculations.

Between 1992–93 and 2011–12 only 9% of the vessels operating in this fishery were observed (24 vessels) but these tended to be the main operators (including most of the larger autoliners) and accounted for up to 52% (by catch) and 60% (by effort) of the total fishery, although coverage was low (less than 10%) prior to 1999–2000).

Ling made up 68% of the total estimated catch from all observed sets targeting ling between 1992–93 and 2011–12, and spiny dogfish (much of which was discarded[3]) about a further 13% (Figure 9.50). About half of the remaining 19% of the catch comprised other commercial species; especially ribaldo (*Mora moro*) (2.9%), smooth skates

[3] Spiny dogfish can legally be discarded under Section 6 of the Fisheries Act 1996.

(*Dipturus innominatus*) (1.8%), red cod (*Pseudophycis bachus*) (1.7%), rough skates (*Zearaja nasuta*, 1.7%), and sea perch (*Helicolenus* spp.) (1.2%). Altogether, 93% of the observed catch was comprised of QMS species. Over 230 species or species groups were identified by observers, the majority being non-commercial species caught in low numbers, especially Chondrichthyans, often unspecified but including shovelnose spiny dogfish (*Deania calcea*), *Etmopterus* species, and seal sharks (*Dalatias licha*). A large weight of echinoderms, especially starfish (of which about 39 t were observed caught during the period), anemones, crustaceans, and other invertebrates were also recorded by observers.

Total annual bycatch estimates for 1992–93 to 2011–12 were 2100–3900 t, compared with approximate target species catches in the same period of 3500–9800 t. Bycatch weights decreased slowly during the period, in line with decreasing effort in the fishery (Figure 9.51).

Total annual discard estimates for 1992–93 to 2011–12 were 1200–2500 t, and generally decreased during the period (Figure 9.52). About 45–70% of these discarded fish were quota species, mainly spiny dogfish,[4] the remainder being non-quota, generally non-commercial, species. Ling were discarded in small amounts (40–250 t per year), these generally being attributable to fish being lost on retrieval or predated by marine mammals and birds.

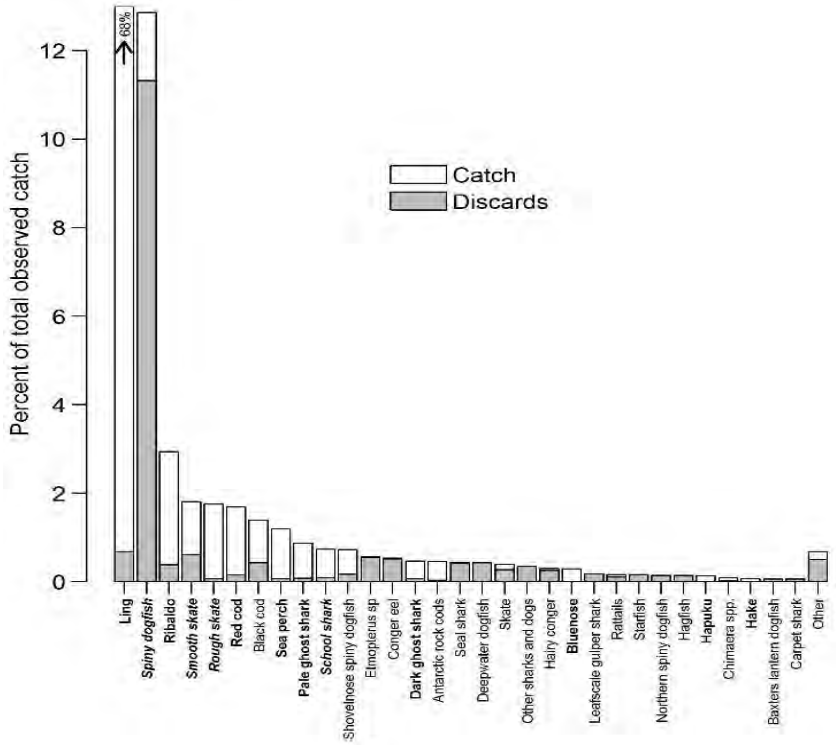


Figure 9.50: Percentage of the total catch contributed by the main bycatch species (those representing 0.05% or more of the total catch; white) in the observed portion of the ling longline fishery, 1992–93 to 2011–12 and the percentage discarded (grey; Anderson 2014a). QMS species are shown in bold.

[4] Spiny dogfish can legally be discarded under Schedule 6 of the Fisheries Act 1996.

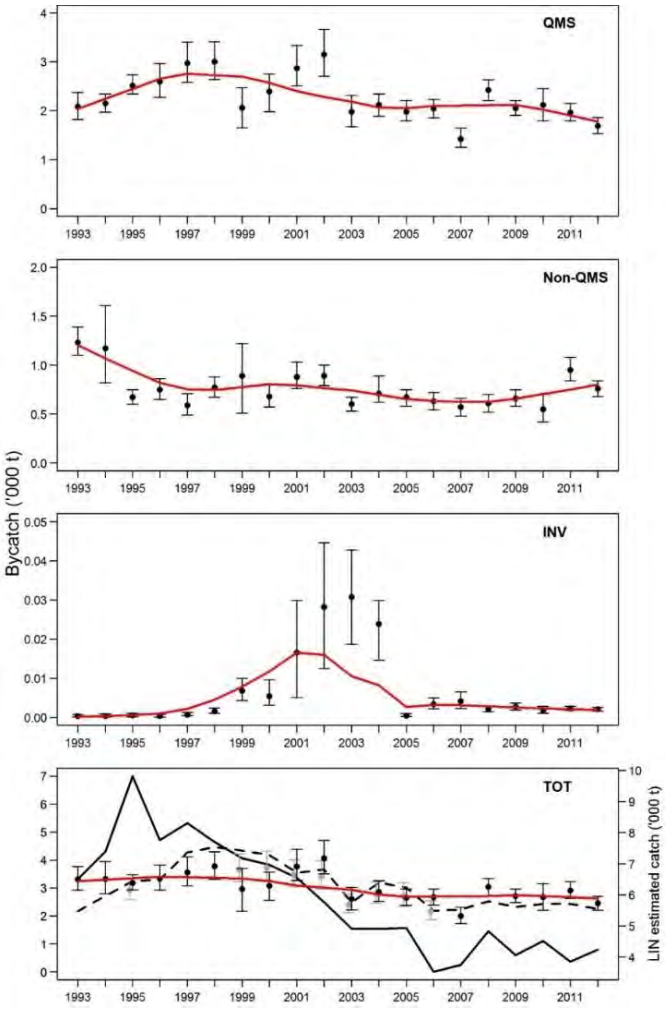


Figure 9.51: Annual estimates of bycatch in the ling longline fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 1992–93 to 2011–12 (Anderson 2014a). Also shown (in grey) are earlier estimates of total bycatch calculated for 1994–95 and 1998–99 to 2005–06 (Anderson 2008). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual estimated commercial longline-catch of ling, and the dashed line shows annual effort (number of hooks), scaled to have mean equal to that of total bycatch.

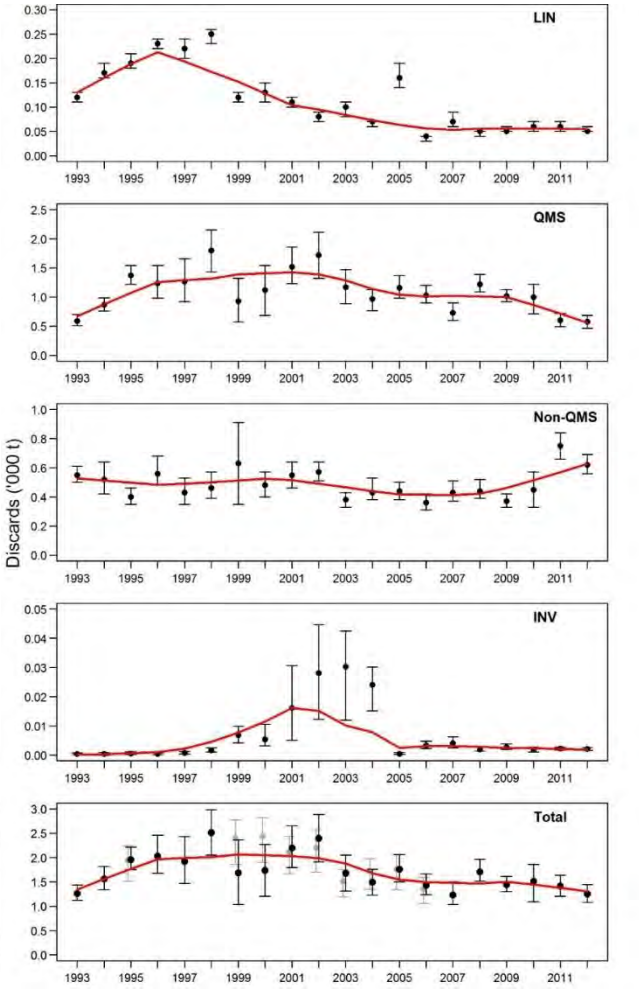


Figure 9.52: Annual estimates of discards in the ling longline fishery, for ling (LIN), QMS species, non-QMS species, invertebrates (INV), and overall for 1992–93 to 2011–12 (Anderson 2014a). Also shown (in grey) are earlier estimates of total discards calculated for 1994–95 and 1998–1999 to 2005–06 (Anderson 2008). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

TRENDS IN BYCATCH BY SPECIES FROM THE LING BOTTOM-LONGLINE FISHERY

Finucci et al. (in prep) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1992–93 to 2016–17. The following conclusions were made:

- The most commonly caught bycatch species were spiny dogfish (*Squalus acanthias*, SPD), ribaldo (*Mora moro*, RIB), and smooth skate (*Dipturus innominatus*, SSK).
- Of the 131 bycatch species examined, one had a significant decrease in catch over time and 34 had a signficant increase in catch.
- The species showing the greatest decline were unspecified skates (SKA), Antarctic rock cod (Nototheniidae, NOT), and lanternsharks (Etmopterus spp., ETM) (Figure 9.53).
- The species showing the greatest increase were the hairy conger (*Bassanago hirsutus*, HCO), hoki

(HOK), and swollenhead conger (*Bassanago bulbiceps*, SCO) (Figure 9.53).

.

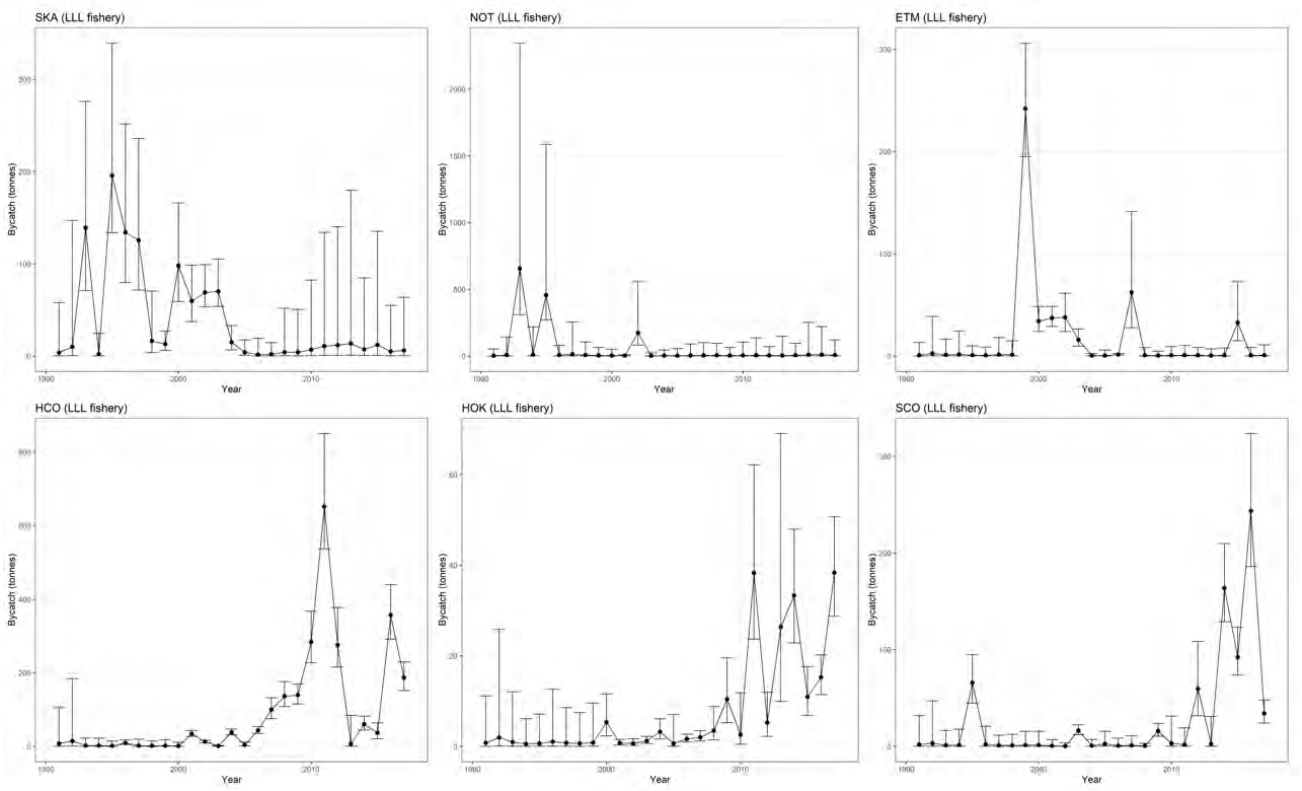


Figure 9.53: Annual bycatch estimates in the ling longline fishery for the species which had the greatest decrease (top) and greatest increase (bottom) between 1992–93 and 2012–13 (Finucci et al. in prep). Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3.1). See text above for species codes.

9.3.20 TUNA LONGLINE FISHERY

The New Zealand tuna longline fishery was dominated by the foreign licensed vessels during the 1980s, then was comprised of chartered Japanese vessels and New Zealand domestic vessels from 1993–94 to 2014–15. The domestic fishing fleet has dominated the fishery since 1993–94 (Figure 9.54).

The Japanese charter fleet mainly targeted southern bluefin tuna off the west coast South Island (WCSI), and domestic vessels targeted southern bluefin tuna and bigeye tuna and the fishery was concentrated on the east coast of the North Island (ECNI) with some fishing for southern bluefin tuna on the WCSI.

A detailed analysis of fish bycatch in tuna longline fisheries covered the 2006–07 to 2009–10 fishing years (Griggs & Baird 2013), and for 2010–11 to 2014–15 (Griggs et al. 2018). During 2010–11 to 2014–15, 137 492 fish and invertebrates from at least 60 species or species groups were observed. Most species were rarely observed, with only 37 species (or species groups) exceeding 100 observations between 1988–89 and 2014–15. The most commonly observed species over all years were blue shark, Ray's bream, and albacore tuna, these three making up nearly 70% of the catch by numbers. Blue shark and Ray's bream were the most abundant and second most abundant species during 2010–11 to 2014–15 (Table 9.2). Other important non-target species were albacore, porbeagle shark, lancetfish, dealfish, deepwater dogfish, swordfish, moonfish, bigscale pomfret, mako shark, and oilfish. The catch composition varied with fleet and area fished.

QMS bycatch species caught were blue shark, mako shark, porbeagle shark, school shark, moonfish, Ray's bream, and swordfish. Swordfish was also sometimes targeted.

Most blue, porbeagle, mako and school sharks were processed in some way, either being finned or retained for their flesh, but there were significant fleet differences. Blue sharks were mainly just finned. Since October 2014, shark finning has been banned in New Zealand waters. Most sharks were discarded in 2014–15, except for some mako and school shark retained for their flesh.

Most albacore, swordfish, yellowfin tuna, moonfish and Ray's bream were retained. Most bigscale pomfret, and rudderfish were discarded, while butterfly tuna, escolar and oilfish were often retained, with some year and fleet differences. Almost all deepwater dogfish, dealfish and lancetfish were discarded.

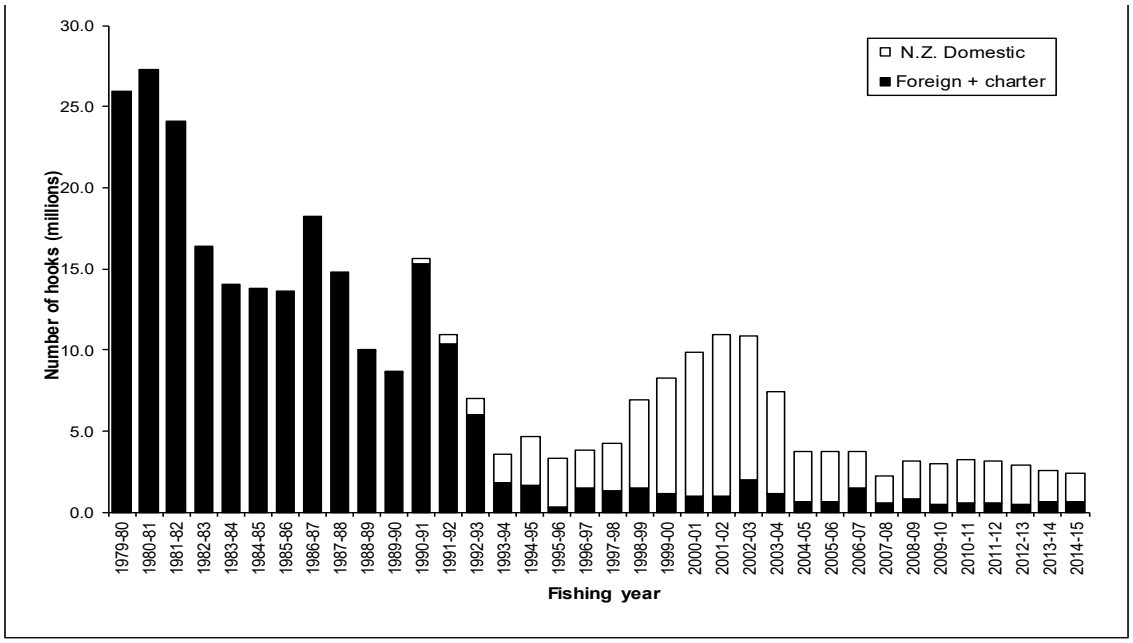


Figure 9.54: Effort (hooks set) in the tuna longline fishery. Black bars are Foreign and Charter vessels, white bars are NZ domestic vessels (Griggs et al. 2018).

Table 9.2: Numbers of fish reported by observers between 2010–11 and 2014–15, and the total observed catch since 1988–89. The top 30 species captured are ranked in descending order of abundance since 1988–89 (Griggs et al. 2018).

	Species	Scientific name	2010–11 to 2014–15	Total number
1	Blue shark	*Prionace glauca*	57 912	240 540
2	Ray's bream	*Brama brama*	26 427	124 632
3	Albacore tuna	*Thunnus alalunga*	9 707	111 023
4	Southern bluefin tuna	*Thunnus maccoyii*	19 149	62 440
5	Porbeagle shark	*Lamna nasus*	3 058	22 069
6	Lancetfish	*Alepisaurus ferox & A. brevirostris*	5 256	19 639
7	Dealfish	*Trachipterus trachypterus*	1 761	18 946
8	Deepwater dogfish	Squaliformes	2 459	11 571
9	Swordfish	*Xiphias gladius*	2 868	11 154
10	Moonfish	*Lampris guttatus*	1 070	10 204
11	Big scale pomfret	*Taractichthys longipinnis*	361	8 179
12	Mako shark	*Isurus oxyrinchus*	1 660	7 822
13	Oilfish	*Ruvettus pretiosus*	256	7 798
14	Escolar	*Lepidocybium flavobrunneum*	895	5 317
15	Rudderfish	*Centrolophus niger*	370	5 277
16	Bigeye tuna	*Thunnus obesus*	663	5 053
17	Butterfly tuna	*Gasterochisma melampus*	510	4 979
18	School shark	*Galeorhinus galeus*	157	3 777
19	Sunfish	*Mola mola*	746	3 501
20	Yellowfin tuna	*Thunnus albacares*	29	3 371
21	Pelagic stingray	*Pteroplatytrygon violacea*	475	2 873
22	Hoki	*Macruronus novaezelandiae*	20	2 041
23	Thresher shark	*Alopias vulpinus*	100	1 500
24	Skipjack tuna	*Katsuwonus pelamis*	50	1 201
25	Dolphinfish	*Coryphaena hippurus*	192	800
26	Flathead pomfret	*Taractes asper*	106	622
27	Striped marlin	*Tetrapturus audax*	39	507
28	Black barracouta	*Nesiarchus nasutus*	84	470
29	Barracouta	*Thyrsites atun*	3	360
30	Pacific bluefin tuna	*Thunnus orientalis*	42	264

9.3.21 ALBACORE TUNA TROLL FISHERY

This fishery was carried out by small domestic vessels fishing over the summer months mainly on the west coast of the North and South Island, especially WCSI.

Observers began to go to sea on troll vessels in 2007. The first two years were a trial period with one trip observed in each year. Targets were set in 2009. Coverage was 0.5–1.5% of days fished for the 2009–10 to 2012–13 fishing years.

Albacore was 94.4% of the observed catch over the past seven years, followed by Ray's bream (2.7%), Skipjack tuna (1.7%), and small numbers (less than 1%) of a few other species (Table 9.3).

Observer coverage on troll vessels was discontinued after 2012–13 as it was considered to not be representative enough of the fishery for length monitoring, which is carried out by port sampling.

Table 9.3: Species composition of observed albacore troll catches, 2006–07 to 2012–13.

Species	Scientific name	Number of fish caught							
		2006–07	2007–08	2008–09	2009–10	2010–11	2011–12	2012–13	Total of 7 years
Albacore tuna	*Thunnus alalunga*	1 684	1 776	1 755	5 403	4 905	2 772	3 881	22 176
Ray's bream	*Brama brama*		18	12	537	35	7	15	624
Skipjack tuna	*Katsuwonus pelamis*	1	2	26	20	359	2		410
Barracouta	*Thyrsites atun*			1		24	13	23	61
Kahawai	*Arripis trutta*			6		3	14	14	37
Kingfish	*Seriola lalandi*			2	4	4			10
Dolphinfish	*Coryphaena hippurus*				1				1
Mako shark	*Isurus oxyrinchus*						1	1	2
Unidentified		2			174				176

9.3.22 SKIPJACK TUNA PURSE SEINE FISHERY

Skipjack tuna was 97.0 % of the catch observed on purse seine vessels in New Zealand waters in 2015 and 2016. Catch composition from four observed purse seine trips operating within New Zealand fisheries waters in 2015 and 2016 can be seen in Table 9.4.

Table 9.4: Catch composition from six observed purse seine trips operating within New Zealand fisheries waters in 2015 and 2016. [Continued on next page]

Common name	Scientific name	Observed catch weight (kg)	% of catch
Skipjack tuna	*Katsuwonus pelamis*	3 478 271	97.05
Jack mackerel	*Trachurus* spp.	80 573	2.25
Sunfish	*Mola mola*	8 867	0.25
Blue mackerel	*Scomber australasicus*	5 646	0.16
Frigate tuna	*Auxis thazard*	2 839	0.08
Spine-tailed devil ray	*Mobula japanica*	1 641	0.05
Striped marlin	*Tetrapturus audax*	1 190	0.03
Jack mackerel	*Trachurus novaezelandiae*	1 030	0.03
Albacore tuna	*Thunnus alalunga*	734	0.02
Blue marlin	*Makaira mazara*	650	0.02
Marlin unspecified		600	0.02
Frostfish	*Lepidopus caudatus*	390	0.01
Mako shark	*Isurus oxyrinchus*	385	0.01
Jellyfish		266	0.01
Slender tuna	*Allothunnus fallai*	177	<0.01
Southern bluefin tuna	*Thunnus maccoyii*	130	<0.01
Flying fish	Exocoetidae	92	<0.01
Bigeye thresher shark	*Alopias superciliosus*	80	<0.01
Yellowfin tuna	*Thunnus albacares*	80	<0.01
Squid	Teuthoidea	56	<0.01
Giant stargazer	*Kathetostoma giganteum*	50	<0.01
Stingray	Dasyatididae	45	<0.01
Smooth skate	*Dipturus innominatus*	35	<0.01
Discfish	*Diretmus argenteus*	30	<0.01

Table 9.4 [Continued]:

Common name	Scientific name	Observed catch weight (kg)	% of catch
Porcupine fish	*Allomycterus jaculiferus*	18	<0.01
Dolphinfish	*Coryphaena hippurus*	15	<0.01
Octopus		12	<0.01
Ray's bream	*Brama brama*	10	<0.01
Snapper	*Pagrus auratus*	10	<0.01
School shark	*Galeorhinus galeus*	8	<0.01
Electric ray	*Torpedo fairchildi*	6	<0.01
Pilotfish	*Naucrates ductor*	6	<0.01
Barracouta	*Thyrsites atun*	5	<0.01
Pelagic ray	*Pteroplatytrygon violacea*	5	<0.01
Ling	*Genypterus blacodes*	5	<0.01
Tarakihi	*Nemadactylus macropterus*	4	<0.01
Flatfish		3	<0.01
John Dory	*Zeus faber*	3	<0.01
Kingfish	*Seriola lalandi*	3	<0.01
Skate		3	<0.01
Dealfish	*Trachipterus trachypterus*	2	<0.01
Pale ghost shark	*Hydrolagus bemisi*	1	<0.01
Spotted gurnard	*Pterygotrigla picta*	1	<0.01
Leatherjacket	*Parika scaber*	1	<0.01
Louvar	*Luvaris imperialis*	1	<0.01
NZ northern arrow squid	*Nototodarus gouldi*	1	<0.01
Opah	*Lampris immaculatus*	1	<0.01
Starfish		1	<0.01
Unidentified		1	<0.01

9.4 INDICATORS AND TRENDS

A standard measure that can be used to characterise a fishery is the level of annual discards as a fraction of the catch of the target species. The most recent estimates (mean of last four years) are provided in Table 9.5 for those fisheries where the necessary data were available. The largest mean discard fraction comes from the scampi trawl fishery where 3.8 kg of bycatch is discarded for every kilogram of scampi caught, and the smallest discard fractions are seen in the oreo, jack mackerel, and southern blue whiting fisheries (0.01 kg).

Comparison of estimates of total bycatch over time from all the deepwater trawl fisheries (Figure 9.55) shows the substantial contribution from the large combined hoki/hake/ling/silver warehou/white warehou trawl fishery (2017–18 hoki total TACC of 150 000 t) even though the relative rate of *discards* from these fisheries is low (see Table 9.5). This figure also shows the relatively large bycatch from the scampi fishery (2017–18 scampi total TACC of 1244 t) and the arrow squid fishery (2017–18 arrow squid total TACC of 82 120 t).

Some general trends were identified in some fisheries, especially those examined in recent Fisheries New Zealand projects where the determination of trends in the rates and levels of bycatch over time was an explicit objective (Table 9.6).

Table 9.5: Utilisation rates. Kilograms of discards per kilogram of target species catch. The numbers are the most recent estimate (mean of the most recent four years available) from referenced reports.

Fishery	Discards/target species catch (kg)
Arrow squid trawl	0.12
Ling longline	0.34
Hoki/hake/ling trawl	0.04
Jack mackerel trawl	0.01
Southern blue whiting trawl	0.01
Orange roughy trawl	0.04
Oreo trawl	0.01
Scampi trawl	3.83

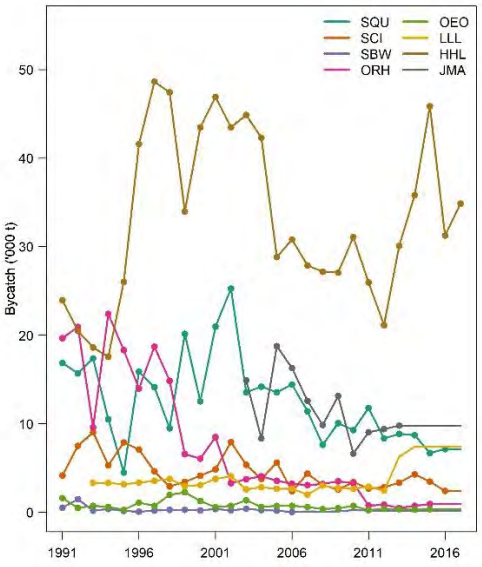


Figure 9.55: Comparison of total estimated bycatch for all the deepwater trawl fisheries 1990–91 to 2016–17. Dots are model based estimates; no dots are course based estimates (see Finucci et al. in prep for more details) For species codes see Table 9.1).

Table 9.6: Trends in non-protected species bycatch from recent MPI projects where trend determination was an objective.

Fishery	Trends
Arrow squid trawl	Linear regressions of annual bycatch estimates since 2002–03 indicated decreasing bycatch over time (negative slopes) in each of the major species categories examined (i.e., QMS species, Non-QMS species, Invertebrate species, and all species combined). These trends were statistically significant (p<0.01) in each case. Linear regressions of annual discard estimates since 2002–03 also indicated decreasing levels over time in each catch category excluding the target species. These declines were statistically significant for non-QMS species, invertebrate species, and all species combined.
Jack mackerel trawl	Linear regressions showed decreasing bycatch of QMS and non-QMS species, and increasing bycatch of invertebrate species and spiny dogfish, but these trends were not statistically significant. Linear regressions showed decreasing discards of non-QMS species, invertebrate species and spiny dogfish, and increasing discards of jack mackerels and QMS species, but these trends were only significant for jack mackerel.
Orange roughy trawl	Increased non-QMS species bycatch quantities between the mid-1990s and mid-2000s were shown to strongly correlate with an overall increase in mean trawl length in the fishery resulting from increased effort away from undersea features (Anderson 2009a). Linear regressions indicated significantly decreasing levels of both bycatch and discards since 2001–02 for several species categories: QMS species, non-QMS species, invertebrates, morid cods, sharks, slickheads, and all species combined.
Oreo trawl	Linear regressions indicated significantly decreasing levels of both bycatch and discards since 2001–02 for non-QMS species, invertebrates, morid cods, rattails, and all species combined.
Scampi trawl	Linear regressions of annual bycatch estimates since 2002–03 indicated decreasing bycatch over time (negative slopes) in each of the major species categories examined (i.e., QMS species, non-QMS species, invertebrate species, and all species combined). None of these trends were statistically significant (p < 0.01). Linear regressions of annual discard estimates since 2002–03 indicated decreasing levels over time for the target species and in the Invertebrate species category, and increasing levels in the QMS and non-QMS species categories, and for all species combined. However, none of these trends were statistically significant (p < 0.01).
Hoki, hake, ling trawl	Linear regressions of annual bycatch estimates since 1990–91 indicated that bycatch in the QMS species, invertebrate species, and all species combined categories generally increased over time, and these trends were statistically significant for QMS species and invertebrate species; but for the largest category, non-QMS species, there was a (non-significant) decline in bycatch over time.
Ling longline	Linear regression modelling of observer catch data indicated increasing bycatch rates (kg/hook) for both QMS species and non-QMS species in LIN 2, and for QMS species in COOK; and decreasing bycatch rates for both QMS species and non-QMS species in BNTY, and QMS species in LIN 4. There were also decreasing discard rates of ling and QMS species in some areas, notably the Bounty and Campbell Plateaus, and increasing discard rates for both QMS species and non-QMS species in LIN 2.

Finucci et al. (in prep) analysed temporal (1990–91 to 2016–17) bycatch trends for individual species or species groups for seven deepwater trawl and one bottom-longline (ling) fisheries. Bycatch regression slope coefficients assessed for each species and fishery showed a consistent increase (in six or more of the eight fisheries) for pale ghost shark (*Hydrolagus bemisi*), rough skate (*Zearaja nasuta*), leafscale gulpher shark (*Centrophorus squamosus*), and Baxters dogfish (*Etmopterus granulosus*); and consistent decline for skates (Rajidae and Arhynchobatidae), dark ghost shark (*Hydrolagus novaezealandiae*), unidentified sharks and rattails, and bluenose (*Hyperoglyphe antarctica*). Some of the trends may be attributable to changes in reporting behaviour, e.g., increased reporting of specific skates and reduced use of the generic skate category. It seems likely that a bycatch decline for well-known species such as bluenose may represent a change in availability, abundance or distribution of that species.

9.5 REFERENCES

Alverson, D L; Freeberg, M H; Murawski, S A; Pope, J G (1994) A global assessment of fisheries bycatch and discards. *FAO Technical Paper No. 339*. FAO, Rome. 233 p.

Anderson, O F (2004a) Fish discards and non-target fish catch in the fisheries for southern blue whiting and oreos. *New Zealand Fisheries Assessment Report 2004/9*. 40 p.

Anderson, O F (2004b) Fish discards and non-target fish catch in the trawl fisheries for arrow squid, jack mackerel, and scampi in New Zealand waters. *New Zealand Fisheries Assessment Report 2004/10*. 61 p.

Anderson, O F (2007) Fish discards and non-target fish catch in the New Zealand jack mackerel trawl fishery, 2001–02 to 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 8*. 36 p.

Anderson, O F (2008) Fish and invertebrate bycatch and discards in ling longline fisheries, 1998–2006. *New Zealand Aquatic Environment and Biodiversity Report No. 23*. 43 p.

Anderson, O F (2009a) Fish discards and non-target fish catch in the New Zealand orange roughy trawl fishery, 1999–2000 to 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 39*. 40 p.

Anderson, O F (2009b) Fish and invertebrate bycatch and discards in southern blue whiting fisheries, 2002–07. *New Zealand Aquatic Environment and Biodiversity Report No. 43*. 42 p.

Anderson, O F (2011) Fish and invertebrate bycatch and discards in orange roughy and oreo fisheries from 1990–91 until 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 67*. 60 p.

Anderson, O F (2012) Fish and invertebrate bycatch and discards in New Zealand scampi fisheries from 1990–91 until 2009–10. *New Zealand Aquatic Environment and Biodiversity Report No. 100*. 65 p.

Anderson, O F (2013a) Fish and invertebrate bycatch and discards in New Zealand arrow squid fisheries from 1990–91 until 2010–11. *New Zealand Aquatic Environment and Biodiversity Report No. 112*. 62 p.

Anderson, O F (2013b) Fish and invertebrate bycatch in New Zealand deepwater fisheries from 1990–91 until 2010–11. *New Zealand Aquatic Environment and Biodiversity Report No. 113*. 57 p.

Anderson, O F (2014a) Fish and invertebrate bycatch and discards in New Zealand ling longline fisheries from 1992–93 until 2011–12. *New Zealand Aquatic Environment and Biodiversity Report No. 138*. 66 p.

Anderson, O F (2014b) Fish and invertebrate bycatch in New Zealand deepwater fisheries from 1990–91 until 2011–12. *New Zealand Aquatic Environment and Biodiversity Report No. 139*. 60 p.

Anderson, O F (2017) Fish and invertebrate bycatch in New Zealand deepwater fisheries from 1990–91 until 2013–14. *New Zealand Aquatic Environment and Biodiversity Report No. 181*. 75 p.

Anderson, O F; Ballara, S L; Edwards, C T T (2017a) Fish and invertebrate bycatch and discards in New Zealand orange roughy and oreo trawl fisheries from 2001–02 until 2014–15. *New Zealand Aquatic Environment and Biodiversity Report No. 190*. 216 p.

Anderson, O F; Clark, M R (2003) Analysis of bycatch in the fishery for orange roughy, *Hoplostethus atlanticus*, on the South Tasman Rise. *Marine and Freshwater Research 54*: 643–652.

Anderson, O F; Clark, M R; Gilbert, D J (2000) Bycatch and discards in trawl fisheries for jack mackerel and arrow squid, and in the longline fishery for ling, in New Zealand waters. *NIWA Technical Report 74*. 44 p.

Anderson, O F; Edwards, C T T (2018) Fish and invertebrate bycatch and discards in New Zealand arrow squid and scampi trawl fisheries from 2002–03 until 2015–16. *New Zealand Aquatic Environment and Biodiversity Report No. 199*. 136 p.

Anderson, O F; Edwards, C T T; Ballara, S (in prep) Fish and invertebrate bycatch and discards in New Zealand hoki, hake, and ling trawl fisheries from 2002–03 until 2016–17. Draft New Zealand Aquatic Environment and Biodiversity Report prepared for Fisheries NZ project DAE2017-01.

Anderson, O F; Edwards, C T T; Roux, M-J (2017b) Fish and invertebrate bycatch and discards in New Zealand jack mackerel trawl fisheries from 2002–03 until 2013–14. *New Zealand Aquatic Environment and Biodiversity Report No. 177*. 71 p.

Anderson, O F; Gilbert, D J; Clark, M R (2001) Fish discards and non-target catch in the trawl fisheries for orange roughy and hoki in New

Zealand waters for the fishing years 1990–91 to 1998–99. *New Zealand Fisheries Assessment Report 2001/16*. 57 p.

Anderson, O F; Smith, M H (2005) Fish discards and non-target fish catch in the New Zealand hoki trawl fishery, 1999–2000 to 2002–03. *New Zealand Fisheries Assessment Report 2005/3*. 37 p.

Anon (2013) New Zealand Annual Report to the Commission – Part 1. Information on fisheries, research, and statistics, New Zealand. Western and Central Pacific Fisheries Commission Scientific Committee Ninth regular session, 6–14 August 2013, Pohnpei, Federated States of Micronesia. WCPFC-SC9-AR/CCM-15 2013. 29 p.

Anon (2017) Tenth Annual Report to the Commission – Part 1. Information on fisheries, research, and statistics, New Zealand. Western and Central Pacific Fisheries Commission Scientific Committee, 9–17 August 2017, Rarotonga, Cook Islands. WCPFC-SC-2017-AR/CCM-16. 36 p.

Ayers, D; Francis, M P; Griggs, L H; Baird, S J (2004) Fish bycatch in New Zealand tuna longline fisheries, 2000–01 and 2001–02. *New Zealand Fisheries Assessment Report 2004/46*. 47 p.

Ballara, S L (2015) Fish and invertebrate bycatch in New Zealand deepwater fisheries from 1990–91 until 2012–13. *New Zealand Aquatic Environment and Biodiversity Report No. 158.* 79 p.

Ballara, S L; Anderson, O F (2009) Fish discards and non-target fish catch in the trawl fisheries for arrow squid and scampi in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 38*. 102 p.

Ballara, S L; O'Driscoll, R L (2015) Fish and invertebrate bycatch and discards in New Zealand hoki, hake and ling trawl fisheries from 1990–91 until 2012–13. *New Zealand Aquatic Environment and Biodiversity Report No. 163.* 120 p.

Ballara, S L; O'Driscoll, R L; Anderson, O F (2010) Fish discards and non-target fish catch in the trawl fishery for hoki, hake, and ling in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 48*. 100 p.

Bellido, J M; Santos, B M; Pennino, G M; Valeiras, X; Pierce, G J (2011) Fishery discards and bycatch: solutions for an ecosystem approach to fisheries management? *Hydrobiologia 670*: 317–333.

Borges, L; Zuur, A F; Rogana, E; Officer, R (2005) Choosing the best sampling unit and auxiliary variable for discards estimations. *Fisheries Research 75*: 29–39.

Bradford, E (2002) Estimation of the variance of mean catch rates and total catches of non-target species in New Zealand fisheries. *New Zealand Fisheries Assessment Report 2002/54*. 60 p.

Casini, M; Vitale, F; Cardinale, M (2003) Trends in biomass and changes in spatial distribution of demersal fish species in Kattegatt and Skagerrak between 1981 and 2003. ICES CM 2003/Q: 14.

Clark, M R; Anderson, O F; Gilbert, D J (2000) Discards in trawl fisheries for southern blue whiting, orange roughy, hoki, and oreos in New Zealand waters. *NIWA Technical Report 71.* 73 p.

Davies, R W D; Cripps, S J; Nickson, A; Porter, G (2009) Defining and estimating global marine fisheries bycatch. *Marine Policy 33*: 661–672.

Edwards, C T T; Doonan, I; Anderson, O F (2015) Comparison of bycatch estimation for fish species using a ratio estimator and model-based method. *New Zealand Aquatic Environment and Biodiversity Report No. 154.* 30 p.

Fernandes, P G; Coull, K; Davis, C; Clark, P; Catarino, R; Bailey, N; Fryer, R; Pout, A (2011) Observations of discards in the Scottish mixed demersal trawl fishery. *ICES Journal of Marine Science 68*: 1734–1742.

Finucci, B.; Edwards, C.T.T; Anderson, O.F.; Ballara, S.L. (in prep). Fish and invertebrate bycatch in New Zealand deepwater fisheries from 1990–91 until 2016–17. Draft New Zealand Aquatic Environment and Biodiversity Report prepared for Fisheries NZ project DAE2017-04.

Griggs, L H; Baird, S J (2013) Fish bycatch in New Zealand tuna longline fisheries in 2006–07 to 2009–10. *New Zealand Fisheries Assessment Report 2013/13*. 73 p.

Griggs, L H; Baird, S J; Francis, M P (2018) Fish bycatch in New Zealand tuna longline fisheries 2010–11 to 2014–15. *New Zealand Fisheries Assessment Report 2018/29.* 91 p.

Griggs, L; Doonan, I; McGregor, V; McKenzie, A (2014) Monitoring the length structure of commercial landings of albacore (*Thunnus alalunga*) during the 2012–13 fishing year. *New Zealand Fisheries Assessment Report 2014/30*. 50 p. Retrieved from http://fs.fish.govt.nz/Page.aspx?pk=113&dk=23652.

Kelleher, K (2005) Discards in the world's marine fisheries. An update. *FAO Fisheries Technical Paper No. 470*. FAO, Rome. 131 p.

McCaughran, D A (1992) Standardized nomenclature and methods of defining bycatch levels and implications. In: Schoning, R W; Jacobson, R W; Alverson, D L; Gentle, T G; Auyong, J (Eds.), Proceedings of the National Industry Bycatch Workshop. pp 200–201. 4–6 February 1992, Oregon.

Ministry for Primary Industries (2012) Operational plan to manage the incidental capture of New Zealand sea lions in the Southern Squid Trawl Fishery (SQU6T). ISBN Online: 978-0-478-42003-6; ISSN Online: 2253-3923.

Ministry for Primary Industries (2013a) Fisheries Assessment Plenary, May 2013: stock assessments and yield estimates. Compiled by the Fisheries Science Group, Ministry for Primary Industries, Wellington, New Zealand. 1357 p.

Ministry for Primary Industries (2013b) Jack Mackerel Chapter, National Fisheries Plan for deepwater and middle depth fisheries. 63 p.

Pope, J G; MacDonald, D S; Daan, N; Reynolds, J D; Jennings, S (2000) Gauging the impact of fishing mortality on non-target species. *ICES Journal of Marine Science 57*: 689–696.

Ramm, K (2012) Conservation Services Programme Observer Report: 1 July 2010 to 30 June 2011. Final Report. Conservation Services Programme, Department of Conservation.

Saila, S (1983) Importance and assessment of discards in commercial fisheries. *FAO Circular No. 765*. FAO, Rome. 62 p.

Starr, P J; Kendrick, T H (2012) GUR 3 Fishery Characterisation and CPUE Report. SINS-WG-2012-14v2. 72 p. (Unpublished document held by Fisheries New Zealand, Wellington.)

Starr, P J; Kendrick, T H (2013) ELE 3 & 5 Fishery Characterisation and CPUE. *New Zealand Fisheries Assessment Report 2013/38*. 95 p.

Starr, P J; Kendrick, T H; Bentley, N (2010a) Report to the Adaptive Management Programme Fishery Assessment Working Group: Characterisation, CPUE analysis and logbook data for SCH 3. Document 2010/07-v2. 62 p. (Unpublished document held by Fisheries New Zealand, Wellington.) Retrieved from http://cs.fish.govt.nz/forums/thread/3874.aspx.

Starr, P J; Kendrick, T H; Bentley, N (2010b) Report to the Adaptive Management Programme Fishery Assessment Working Group: Characterisation, CPUE analysis and logbook data for SCH 5. Document 2010/08-v2. 65 p. (Unpublished document held by Fisheries New Zealand, Wellington.) Retrieved from http://cs.fish.govt.nz/forums/thread/3875.aspx.

Starr, P J; Kendrick, T H; Bentley, N (2010c) Report to the Adaptive Management Programme Fishery Assessment Working Group: Characterisation, CPUE analysis and logbook data for SCH 7 and SCH 8. Document 2010/09-v2. 149 p. (Unpublished document held by Fisheries New Zealand, Wellington.) Retrieved from http://cs.fish.govt.nz/forums/thread/3876.aspx.

Thompson, S K (1992) Sampling. John Wiley & Sons, Inc., New York. 343 p.

10 CHONDRICHTHYANS (SHARKS, RAYS AND CHIMAERAS)

Status of chapter	This chapter has been fully updated for AEBAR 2018.
Scope of chapter	This chapter outlines the relevant biology of New Zealand chondrichthyans, the nature of any fishing interactions, the management approach, and trends in key indicators of fishing effects. This chapter covers Quota Management System (QMS), non-QMS and protected sharks.
Area	All of the New Zealand EEZ and Territorial Sea.
Focal localities	This differs depending upon the species or fishery examined.
Key issues	Sustainability of fisheries extractions with a focus on non-QMS species
Fisheries NZ research (current)	SHA2017-01 Shark fishery characterisation and indicator analysis; ENV2018-06 Improved distribution information for higher risk non-QMS shark species.
Other government research (current)	DOC CSP research: DOC19302 Updated analysis of spine-tailed devil ray post release survival.
University research	Biological and behavioural information is currently known from research at Waikato, Otago, Victoria and Auckland universities on a variety of species.
Related issues/chapters	See the Non-protected species (fish and invertebrates) bycatch chapter. More detail is provided for QMS species in the stock assessment plenary (Ministry for Primary Industries 2017, Fisheries New Zealand 2018).

10.1 CONTEXT

Chondrichthyans (cartilaginous fishes) comprise all fish species (except lampreys and hagfish) that lack true bone in their skeletons, specifically sharks, rays, skates and chimaeras. In New Zealand, seven chondrichthyans are totally protected under the Wildlife Act (1953). The impacts of fishing on chondrichthyans are managed under the Fisheries Act (1996), with eleven species subject to the Quota Management System (QMS) and two species prohibited as target species (although the two species, hammerhead and sharpnose sevengill sharks, may still be taken as bycatch). The management policy framework is contained in Fisheries Plans developed for Deepwater, Highly Migratory, and Inshore fisheries (see Chapter 1 for fuller descriptions and web links).

New Zealand has international obligations to collaborate with other countries in the assessment and management of shared and migratory chondrichthyan stocks. New Zealand participates in a number of Regional Fisheries Management Organisations that have some responsibility for chondrichthyans, including Western and Central Pacific Fisheries Commission (which manages tuna fisheries and the associated species), Commission for the Conservation of Southern Bluefin Tuna (southern bluefin tuna), Commission for the Conservation of Antarctic Marine Living Resources (toothfish), and the South Pacific Regional Fisheries Management Organisation (SPRFMO; multiple non-Highly Migratory Species). New Zealand is also a signatory to conventions that play a role in the management of some species, including the Convention on International Trade in Endangered Species of Wild Fauna and Flora, and the Convention on the Conservation of Migratory Species of Wild Animals.

To address global concerns about the management of chondrichthyans,[1] the Food and Agriculture Organisation of the United Nations (FAO) developed an International Plan of Action for the Conservation and Management of Sharks (IPOA).[2] The IPOA builds upon the FAO Code of Conduct for Responsible Fisheries and was endorsed by the FAO Council

[1] In the IPOA and in the NPOA–Sharks, 'sharks' are defined to include all chondrichthyans, viz. sharks, rays and chimaeras. However, in this chapter, we use the terms chondrichthyans, sharks, rays, chimaeras in their strict sense to avoid confusion. Skates are a type of ray and are grouped with rays.

[2] FAO, International Plan of Action for Conservation and Management of Sharks, http://www.fao.org/ipoa-sharks/en.

in June 1999 and subsequently adopted by the November 1999 FAO Conference. The overarching goal of the IPOA is: 'to ensure the conservation and management of sharks and their longterm sustainable use.' To achieve this goal the IPOA suggests that each member state of FAO that regularly catches sharks, either as target or incidental catch, should develop a National Plan of Action for the Conservation and Management of Sharks (NPOA-Sharks).

New Zealand developed an NPOA–Sharks that came into effect in October 2008 (Ministry of Fisheries 2008), and this was superseded by the NPOA-Sharks 2013 (Ministry for Primary Industries 2013). The purpose of the NPOA-Sharks 2013 is:

> '*To maintain the biodiversity and the long-term viability of all New Zealand shark populations by recognising their role in marine ecosystems, ensuring that any utilisation of sharks is sustainable, and that New Zealand receives positive recognition internationally for its efforts in shark conservation and management.*'

It aims to achieve this purpose by identifying goals and five-year objectives in the following key areas:

- Biodiversity and long-term viability of shark populations;
- Utilisation, waste reduction and the elimination of shark finning;
- Domestic engagement and partnerships;
- Non-fishing threats;
- International engagement;
- Research and information.

It is a comprehensive plan that will improve our knowledge of shark populations and their interactions with fisheries, and assist us to base conservation and management actions on an assessment of risks. The NPOA-Sharks 2013 will be reviewed in 2019.

The NPOA-Sharks applies to all chondrichthyans that are found within New Zealand's Exclusive Economic Zone (EEZ) and Territorial Sea (New Zealand fisheries waters), migratory species that frequent New Zealand fisheries waters, and species taken by New Zealand-flagged vessels fishing on the High Seas (including the Ross Sea, Antarctica). Appendix 19.11.1 provides a list of all 115 known New Zealand and Ross Sea chondrichthyans, along with their management class and IUCN and Department of Conservation threat classes.

10.2 BIOLOGY

The population dynamics of chondrichthyans differ markedly from those of bony fishes. Chondrichthyans have a mammal-like reproductive strategy of producing a small number of well-developed young, rather than spawning large numbers of undeveloped eggs as do most bony fishes. Chondrichthyans either lay large yolky eggs on the seabed or give birth to live young, but in both reproductive modes the number of young produced annually is usually in single digits or in the low tens. A few species may produce more than 100 young per litter (e.g., blue shark has up to 135 young; Last & Stevens 2009), but even in these more fecund species, large litter sizes are exceptional and the average number of young per female is much lower (30–40 in the blue shark; Last & Stevens 2009). Gestation periods and reproductive cycles last 10 months to two years in many species, and may be as high as three years (e.g., school shark, mako shark; Mollet et al. 2000, Walker 2005). Fecundity may increase with the size of females (e.g., rig and school shark; Francis & Mace 1980, Walker 2005), so if human activities reduce the average size of females in a population (as often happens in fisheries) the reproductive output may decline faster than the rate of population decline. These characteristics mean that chondrichthyans have a much closer, potentially almost linear, relationship between population size and recruitment. They also have limited capacity for density-dependent compensation that might boost reproductive output at low population sizes, e.g. through increased growth and reproductive rates.

Many cartilaginous fishes are also slow growing, further reducing their capacity for recovering from population declines. Many species have ages at maturity greater than 10 years and longevities in excess of 20 years, although some are faster growing and are therefore more productive (e.g., rig; Francis & Ó Maolagáin 2000). The combination of low reproductive rate and low growth rate makes chondrichthyans particularly vulnerable to overfishing (Camhi et al. 1998, Smith et al. 1998, Dulvy et al. 2003, 2017, Pikitch et al. 2008, Simpfendorfer & Kyne 2009).

Recent studies have been carried out to estimate the age, growth, length at maturity, and size and age composition of the catch of three main pelagic sharks taken as bycatch in surface longline fisheries (blue, porbeagle and mako sharks) (Francis 2015, Francis 2016b, Francis & Ó Maolagáin 2016). Similar studies have also been carried out on a group of small inshore and deepwater sharks and rays (Francis et al. 2018a, 2018b). Age estimation was equivocal for some of

these species (especially blue shark), and no ageing validation was carried out for any of the species, so our knowledge of growth and age-related parameters (maturity, longevity) remains uncertain. Nevertheless, vertebral band counts of common electric ray, blind electric ray, and carpet shark indicated that these species were relatively fast growing and therefore productive, whereas band counts of dorsal fin spines in several deepwater sharks (Owston's dogfish, longnose velvet dogfish, Plunket's shark) confirmed Australian studies indicating that they are slow-growing and long-lived.

Satellite tags have been deployed on porbeagle, mako and hammerhead sharks to identify their movement patterns, migratory behavior, habitat requirements and use of the water column (Francis et al. 2015b, Francis 2016a; M. Francis unpublished data). Porbeagle sharks make seasonal north-south migrations, and are vertical migrators, spending daytime at depths of 200–600 m and nighttime at 50–100 m. Mako sharks may remain resident in coastal waters for several months, but periodically (mainly in winter) migrate northwards to subtropical and tropical waters. Juvenile hammerheads inhabit coastal waters for their first few years of life.

Biological parameters (e.g. growth, longevity, reproduction, stock identity) were reviewed for Pacific pelagic sharks, providing a readily-available summary of inputs for subsequent population modelling (Clarke et al. 2015).

Anthropogenic threats other than fishing were reviewed for rig populations (Jones et al. 2015). Kaipara Harbour is the most important rig nursery area in the country, and it is significantly impacted by agricultural activities, with the southern area especially vulnerable to the effects of future urbanisation. Raglan and Kawhia harbours may also represent significant nursery areas that are impacted mainly by agricultural activities rather than urbanisation.

Hernández et al. (2015) did a genetic study of South Pacific school shark and found evidence of mixing between New Zealand and Australia (which is consistent with previous tagging results), but genetic separation of the Australasian population from the Chilean population.

10.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

There are numerous examples worldwide of chondrichthyan stocks collapsing under fishing pressure, and until recently little attention has been focused on their management. This situation reflects the generally low importance of chondrichthyans in terms of quantity and value in commercial catches, and the consequent low research and management priority accorded to them. An important driver in the increase in shark fishing mortality globally over the last two decades was growing demand for shark and ray meat and fins, and other products such as mobulid gill plates, cartilage, skin or liver oil (Bräutigam et al. 2015). Although there is evidence of a decline in the shark fin trade since 2011 (Dent & Clarke 2015), many chondrichthyan populations are now believed to be severely depleted. There is also widespread public opposition to shark 'finning', in which only the fins are kept and the rest of the shark is discarded at sea, because of concerns about sustainability, wastage, and finning of live sharks. In New Zealand, live shark finning is an offence under the Animal Welfare Act 1999) and shark finning was banned in October 2014. The results of this ban are now becoming apparent through reduced landings of some species (e.g. blue and porbeagle sharks, Table 10.2).

Chondrichthyans are caught in nearly all parts of the world, ranging from tropical to arctic/antarctic waters, and from estuaries and shallow coastal waters to the deepest areas fished. Chondrichthyans are caught by most fishing methods, although trawling, netting and lining are the most important. Reported global landings of chondrichthyans increased steadily up to almost 900 000 t in the early 2000s but have been declining since then (Worm et al. 2013). However unreported catches are undoubtedly substantial so the true extent of chondrichthyan catches remains unclear (Bonfil 1994, Camhi et al. 1998, Clarke et al. 2006, Worm et al. 2013). Estimates of the mortality rates of chondrichthyans at the time they are hauled to a fishing vessel are available for some species (e.g. Francis et al. 1999a, Campana et al. 2009, Griggs & Baird 2013), and increasing attention is being devoted to estimating the survival of sharks released alive by fishers (Moyes et al. 2006, Campana et al. 2009, Musyl et al. 2011, Hutchinson et al. 2013, Ellis et al. 2017). A large research programme currently underway aims to estimate the post-release mortality of mako and silky sharks released from tuna longlines in New Zealand, Fiji, New Caledonia and Marshall Islands (Clarke et al. 2017). A similar study on the mortality of spinetail devilrays released from purse seiners fishing for skipjack tuna in New Zealand is ongoing (Francis & Jones 2016).

There is ample evidence that many chondrichthyan populations are now overfished and that fishing effort is still expanding in habitats containing some of the most vulnerable species, especially deepwater chondrichthyans (Kyne & Simpfendorfer 2007, Simpfendorfer & Kyne 2009, Rice & Harley 2012a, 2012b). Management measures have been implemented by many countries, particularly for targeted species, and Regional Fisheries Management Organisations are paying greater attention to the need to manage species that occur in international waters or waters that straddle national waters. Efforts are also focusing on reducing shark finning, particularly in fisheries catching pelagic sharks, by requiring fins to be attached to sharks at the point of landing, or to comprise no more than 5% of the landing by weight. It is not clear that this requirement has been effective in reducing catches (Clarke et al. 2012, Worm et al. 2013).

10.4 STATE OF KNOWLEDGE OF FISHERIES INTERACTIONS IN NEW ZEALAND

A total of 115 chondrichthyans are known from New Zealand waters (including the Ross Sea) (Appendix 19.11.1), however that number is expected to change as taxonomic studies continue on deepwater species. Of these species, 12 are chimaeras, 29 are skates and rays, and 74 are sharks. Many New Zealand species also occur elsewhere in the world (some have worldwide distributions) but a high percentage (30%) are endemic to New Zealand. New Zealand's chondrichthyan fauna is small compared with that in Australia, which has more than 322 species (Last & Stevens 2009), and that partly reflects New Zealand's lack of tropical environments. The high percentage of endemic species makes New Zealand's fauna unique and distinctive.

The largest threat to chondrichthyan populations is from fishing activities, although other potential impacts include underwater noise, dredging, sonar surveys, electromagnetic fields generated by power stations and undersea cables, loss of habitat, eutrophication and sedimentation, entrapment by aquaculture facilities, and shark ecotourism (Francis & Lyon 2013, Jones et al. 2015). More than 70 of New Zealand's chondrichthyan species are caught (deliberately or incidentally) by fishers (Ministry for Primary Industries 2013). Eleven chondrichthyans are managed under the QMS (Ministry for Primary Industries 2017, Fisheries New Zealand 2018), seven are protected (Francis & Lyon 2012), two cannot be targeted, and the remainder are Non-QMS species (Appendix 19.11.1). Due to reporting requirements commercial landings of chondrichthyans are relatively well known, but less is known about recreational and customary catches.

A nationwide survey from 1 October 2011 to 30 September 2012 provides the most reliable and recent estimates of recreational chondrichthyan catches (Table 10.1) (Wynne-Jones et al. 2014). This survey was repeated in 2017–18, the results of which will be published in 2019. The majority of the recreational catch is from inshore QMS species, although mako shark inhabits both inshore and offshore regions. 'Stingray' is likely to include more than one species and 'sand shark' is likely to refer mainly to rig or school shark. Mako sharks are also targeted/bycatch in the gamefish charter boat fishery, so estimates for mako are potentially underestimates as the survey was not designed to sample gamefishers on charter boats. Estimates in tonnes are only available for rig and spiny dogfish and these constituted 4.0% and 0.4%, respectively, of the reported commercial landings in the same year for those species. All subsequent data reported in this chapter are from the commercial fishery.

Commercial catches of chondrichthyan species during the five-year period 2012–13 to 2016–17 are shown in Table 10.2 and Figure 10.1. Spiny dogfish produced by far the greatest catches, followed by school shark. Rough skate, dark ghost shark, rig and elephantfish formed a second tier of species, and blue shark, pale ghost shark and smooth skate formed a third tier; the remaining species had relatively low catches (less than about 400 t per year on average). In implementing the objectives of the NPOA Sharks 2013, Fisheries New Zealand has successfully worked with the fishing industry to increase the use of species codes and decrease the use of generic codes (Unspecified sharks (OSD) and Deepwater dogfish (DWD)). OSD was an important category (402 t/year), but use of this code has declined by one-third since the late 2000s. DWD has become a minor category, declining by over 70% since the late 2000s. This indicates that fishers have become better at reporting their shark catch to the species level.

Reported discards in 2014–15 to 2016–17 were significant for spiny dogfish, porbeagle shark, and many non-QMS species (e.g. nearly all carpet shark, stingrays, electric ray and three-quarters of northern spiny dogfish) (Table 10.3). Live releases of eight specified chondrichthyans are permitted under Schedule 6 of the Fisheries Act, and from 2006–07 such releases were not counted against quota (Table 10.4). Spiny dogfish may also be discarded dead, but

they must be counted against a fisher's Annual Catch Entitlement (ACE) and the total allowable catch limit for that species against quota. Live releases in 2014–15 to 2016–17 were a large proportion of the catch of blue, porbeagle and mako sharks (59–98% discarded or released), and 14% of smooth skates (Table 10.3). The conditions of Schedule 6 releases have been amended for mako, porbeagle, and blue shark. From 1 October 2014, fishers have been allowed to return these three species to the sea both alive and dead, although the status must be reported accurately. Those returned to the sea dead are counted against a fisher's Annual Catch Entitlement (ACE) and the total allowable catch limit for that species. The survival rate of discarded and released sharks is unknown, and probably varies enormously with species, fishing method, handling, and other factors.

10.4.1 QMS SPECIES

The eleven chondrichthyans managed under the QMS are shown in Table 10.4 with their Total Allowable Commercial Catches (TACCs) and 2016–17 landings. Landings of all but one species (elephantfish) were below the TACCs.

QMS chondrichthyans are treated in detail in Fisheries New Zealand's annual Fisheries Assessment Plenary reports (Ministry for Primary Industries 2017, Fisheries New Zealand 2018) and that material is not repeated here. Quantitative stock assessments have been attempted for only three chondrichthyan stocks (rig in SPO 3 and SPO 7, and elephantfish in ELE 3) but only the assessment for SPO 7 was accepted and adopted by the MPI Southern Inshore Working Group. However, the 2006 SPO 7 stock assessment has not been updated since then, and the status of all rig stocks is now estimated from trends in standardised CPUE and trawl surveys.

A summary of the status of the stocks of QMS chondrichthyans is given in Appendix 19.11.2. Stock status has been estimated for six of the 11 QMS chondrichthyans, and 20 of the 45 non-nominal stocks. None of the stocks was considered to be below the 'hard limit' and 'soft limit' reference points. Two elephantfish stocks and one rig stock were considered to be 'about as likely as not' to be in an 'overfishing' state; the remainder of the stocks were considered to be in a favourable state.

Quantitative risk assessments have recently been conducted for the Southern Hemisphere porbeagle shark stock, and the Pacific Ocean bigeye thresher shark stock. Both assessments incorporated New Zealand data and can be considered applicable to the New Zealand portion of those stocks (Fu et al. 2016, Hoyle et al. 2017). For bigeye thresher sharks, total fishing mortalities from pelagic longline fisheries in the Pacific since 2000 were generally low (less than 5%), but exceeded the maximum impact sustainable threshold in some years. For porbeagle shark, the risk assessment indicated low fishing mortality rates in the three regions comprising the assessment area (eastern Atlantic Ocean, Indian Ocean, and western Pacific Ocean), and low risk from commercial pelagic longline fisheries to porbeagle shark over the entire Southern Hemisphere.

10.4.2 PROTECTED SPECIES

Seven chondrichthyans are currently protected in New Zealand fisheries waters: white pointer shark (also known as great white shark) was protected in 2007; spinetail devilray, manta ray, whale shark, deepwater nurse shark and basking shark in 2010; and oceanic whitetip shark in 2013.

Data from Fisheries NZ observers, along with fisher-reported data, are used to estimate protected species captures. Observer coverage has been reasonably good over the last decade (2005–06 to 2015–16) or longer in some large valuable fisheries (e.g., trawl fisheries for hoki (15.3–38.6% per year) and orange roughy (11.5–44.1% per year), and in the southern bluefin target longline fishery; 20.8–56.6% per year). Some trawl fisheries around southern New Zealand and skipjack tuna purse seine fisheries in northern New Zealand also had reasonable coverage over the last decade (southern blue whiting (25.2–100.0% per year), squid (12.9–87.1% per year) and tuna (13.8–29.1% per year 2005–06 to 2015–16), providing good information on captures of basking sharks, white pointer sharks and spinetail devilrays. However, observer coverage has not always been representative of the spatial and temporal distribution of these fisheries. Inshore fisheries have received only sparse observer coverage. These fisheries may have unobserved and unrecorded mortality of some protected species, especially basking shark, white pointer shark and deepwater nurse shark.

In recent years, fishers have been reporting increasing numbers of protected species bycatch on a dedicated reporting form, and commercial reports of basking shark and white pointer shark captures have exceeded those reported by observers (Francis 2017a, 2017b).

PROTECTED SPECIES GENETICS

Francis & Ritchie (2016) reviewed the available genetic information for the seven protected chondrichthyan species. They established a repository for genetic samples of protected fish species, conducted a stock-take of completed, current and planned genetic analyses for these species internationally, and provided recommendations on the most appropriate methods of furthering genetic analyses in order to inform management of New Zealand's protected fish species in relation to fisheries bycatch. The tissue repository contains good sample sizes from white pointer shark (N=102) and basking shark (N=56), but small or no samples from the remaining species.

BASKING SHARK

Fifty-nine basking sharks were reported taken as bycatch around southern New Zealand from 2010–11 to 2015–16 (Francis 2017b). The main capture locations are the east coast South Island off Banks Peninsula (FMA 3), the west coast South Island between Westport and Hokitika, Puysegur (FMA 7), the shelf edge south and east of Stewart Island (FMA 5) and the Snares Islands, and around the Auckland Islands (FMA 6). Captures (and sightings) of basking sharks also occurred around North Island but were relatively uncommon (Francis & Duffy 2002, Francis & Sutton 2012, Francis 2017b).

Most basking shark records came from trawl fisheries. The sharks were caught mainly by vessels targeting barracouta and hoki off east coast South Island, hoki off west coast South Island, and arrow squid off Southland-Auckland Island. Basking sharks are also caught in set nets (Francis & Duffy 2002) but have rarely been reported by fishers since they were protected in 2010 (Francis 2017b). The observer coverage of this fleet has been low, so the set net bycatch cannot be quantified. Basking sharks are rarely entangled in surface longlines (Francis & Duffy 2002).

Table 10.1: Recreational harvest estimates for New Zealand chondrichthyan species for the 2011–12 fishing year. Mean fish weights are only available for two species, otherwise only the counts are shown. Mgmt class = Management class, QMS is shown, all others are Non-QMS and non-protected species; CV = Coefficient of variation of the estimate to the left. Reproduced in part from Wynne-Jones et al. (2014). These data have not been updated since 2011–12.

Species	Mgmt class	Fishers (n)	Events (n)	Harvest (n)	CV	Mean weight (kg)	Harvest (t)	CV
Rig	QMS	159	241	47 718	0.14	1.09	52.05	0.14
School Shark	QMS	95	160	30 555	0.17	-	-	-
Spiny Dogfish Shark	QMS	97	119	22 200	0.19	1.02	22.60	0.19
Stingray		46	59	11 053	0.40	-	-	-
Elephant Fish	QMS	24	47	6 198	0.34	-	-	-
Sand Shark		10	18	3 719	0.54	-	-	-
Hammerhead Shark		10	12	1 429	0.34	-	-	-
Bronze Whaler Shark		5	5	570	0.52	-	-	-
Mako Shark	QMS	5	6	529	0.51	-	-	-
Carpet Shark		3	5	452	0.67	-	-	-

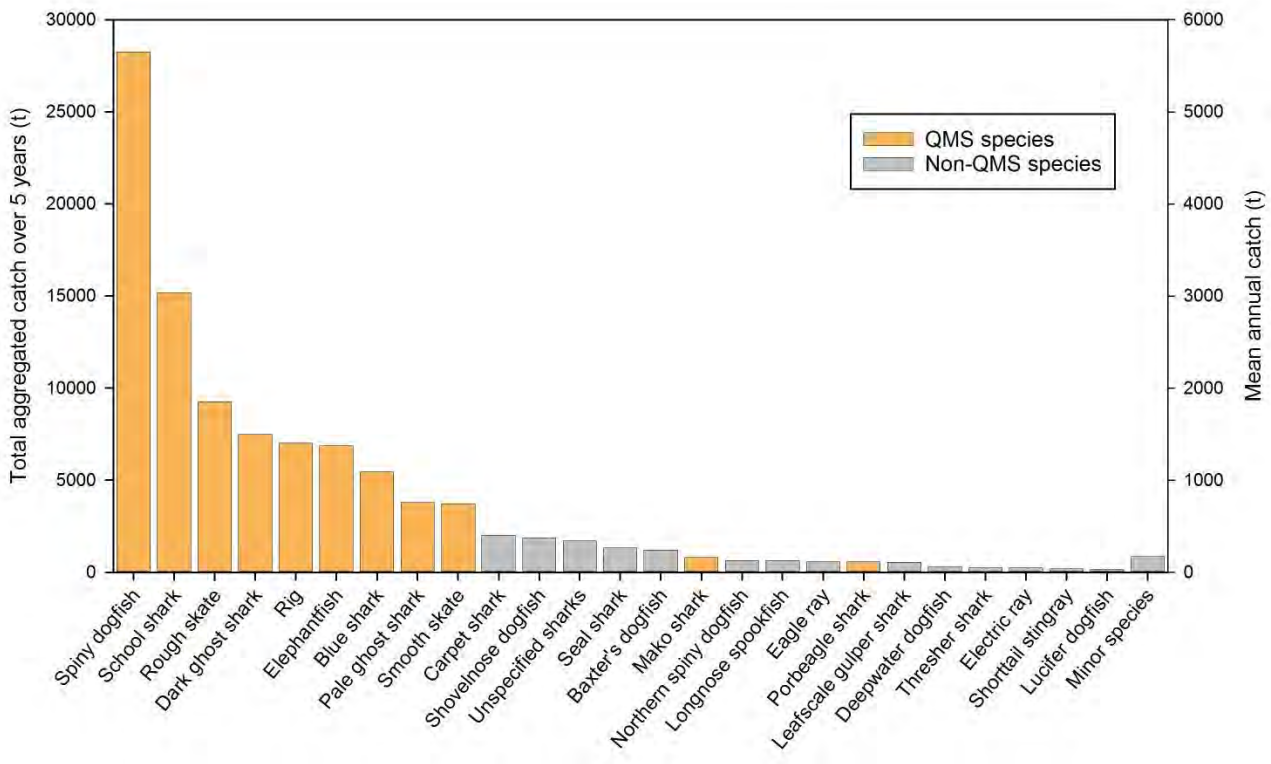


Figure 10.1: Reported total catches (landings, discards and live releases) for chondrichthyan species aggregated across 2012–13 to 2016–17. The average annual catches are shown on the right axis. Source: Fisheries New Zealand catch-effort database.

Table 10.2: Reported total catches (tonnes, including discards and live releases) for chondrichthyan species from 2012–13 to 2016–17, arranged in descending order of total catch. Only species with more than 5 t of aggregated catch are included. The management class is also shown. Source: Fisheries New Zealand catch-effort database. Note: Catches of QMS species differ from landings in Table 10.4 because they include discards and releases, and came from a different source. Protected species are reported by number, not weight, and are not included in this table.

Species	Code	Mgmt class	2012-13	2013-14	2014-15	2015-16	2016-17	Total
Spiny dogfish	SPD	QMS	5092	5963	6409	4521	6268	28252
School shark	SCH	QMS	3179	3134	3097	2934	2850	15194
Rough skate	RSK	QMS	1919	2158	1658	1626	1913	9275
Dark ghost shark	GSH	QMS	1726	1799	1273	1369	1322	7489
Rig	SPO	QMS	1308	1403	1439	1428	1437	7015
Elephantfish	ELE	QMS	1432	1392	1338	1389	1325	6876
Blue shark	BWS	QMS	1378	1115	1051	1001	897	5442
Pale ghost shark	GSP	QMS	701	720	748	712	898	3779
Smooth skate	SSK	QMS	649	713	687	753	919	3720
Carpet shark	CAR	Non-QMS	344	396	408	388	476	2013
Shovelnose dogfish	SND	Non-QMS	185	372	329	540	441	1866
Other sharks and dogs	OSD	Non-QMS	593	278	227	305	295	1698
Seal shark	BSH	Non-QMS	322	287	224	254	245	1332
Baxter's dogfish	ETB	Non-QMS	41	305	293	259	314	1211
Mako shark	MAK	QMS	171	161	198	193	91	814
Northern spiny dogfish	NSD	Non-QMS	94	110	130	135	174	643
Longnose spookfish	LCH	Non-QMS	117	126	113	134	139	628
Eagle ray	EGR	Non-QMS	92	131	104	116	134	575
Porbeagle shark	POS	QMS	115	133	143	97	72	560
Leafscale gulper shark	CSQ	Non-QMS	30	96	124	178	125	552
Deepwater dogfish	DWD	Non-QMS	35	63	67	60	70	295
Thresher shark	THR	Non-QMS	38	44	51	57	50	239
Electric ray	ERA	Non-QMS	41	48	47	42	53	231
Shorttail stingray	BRA	Non-QMS	14	29	31	44	67	186
Lucifer dogfish	ETL	Non-QMS	32	21	32	24	36	144
Broadnose sevengill shark	SEV	Non-QMS	20	21	27	21	23	113
Longtail stingray	WRA	Non-QMS	13	20	23	21	27	104
Slender smooth hound	SSH	Non-QMS	30	10	22	29	13	103
Longnose velvet dogfish	CYP	Non-QMS	8	38	10	21	26	103
Bronze whaler	BWH	Non-QMS	11	13	10	16	18	67
Hammerhead shark	HHS	Moratorium	10	11	12	15	14	62
Plunket's shark	PLS	Non-QMS	3	7	9	10	16	45
Other skates	OSK	Non-QMS	10	7	15	8	4	44
Purple chimaera	CHG	Non-QMS	13	3	7	5	12	41
Stingrays	STR	Non-QMS	3	8	11	6	13	40
Sixgill shark	HEX	Non-QMS	4	3	5	9	8	28
Owston's dogfish	CYO	Non-QMS	3	3	5	8	4	22
Cimaeras	CHI	Non-QMS	2	2	1	7	6	19
Prickly dogfish	PDG	Non-QMS	4	4	5	2	3	18
Deepwater spiny skate	DSK	Non-QMS	10	1	2	1	4	17
Rays	RAY	Non-QMS	12	1	1	1	1	16
Blind electric ray	BER	Non-QMS	3	1	2	2	1	8
Portuguese dogfish	CYL	Non-QMS	0	1	4	0	1	6

Table 10.3: Percentages of reported total catches that were landed, discarded and released alive for chondrichthyan species from 2014–15 to 2016–17, arranged in descending order of total catch. Note: Species order differs from that in Table 10.2 because the two tables cover different time periods. Protected species are not included (all are required to be discarded or released). For blue shark, mako shark and porbeagle shark, which are mainly caught by surface longline fisheries, the last column includes significant percentages of dead discards because they are not distinguished from live releases on TLCER forms. Source: Fisheries New Zealand catch-effort database.

Species	Code	Mgmt class	Catch (t)	Landed %	Discarded %	Released %
Spiny dogfish	SPD	QMS	17198	34	66	0
School shark	SCH	QMS	8881	99	1	0
Rough skate	RSK	QMS	5198	94	1	6
Rig	SPO	QMS	4305	98	0	1
Elephantfish	ELE	QMS	4052	100	0	0
Dark ghost shark	GSH	QMS	3965	95	5	0
Blue shark	BWS	QMS	2949	2	0	98
Smooth skate	SSK	QMS	2358	85	1	14
Pale ghost shark	GSP	QMS	2358	98	2	0
Shovelnose dogfish	SND	Non-QMS	1310	66	34	0
Carpet shark	CAR	Non-QMS	1272	2	98	0
Baxter's dogfish	ETB	Non-QMS	866	76	24	0
Other sharks and dogs	OSD	Non-QMS	828	44	56	0
Seal shark	BSH	Non-QMS	723	63	37	0
Mako shark	MAK	QMS	481	10	8	82
Northern spiny dogfish	NSD	Non-QMS	439	23	77	0
Leafscale gulper shark	CSQ	Non-QMS	426	67	33	0
Longnose spookfish	LCH	Non-QMS	385	85	15	0
Eagle ray	EGR	Non-QMS	353	51	49	0
Porbeagle shark	POS	QMS	312	9	32	59
Deepwater dogfish	DWD	Non-QMS	197	85	15	0
Thresher shark	THR	Non-QMS	158	20	80	0
Shorttail stingray	BRA	Non-QMS	143	1	99	0
Electric ray	ERA	Non-QMS	142	8	92	0
Lucifer dogfish	ETL	Non-QMS	91	66	34	0
Broadnose sevengill shark	SEV	Non-QMS	72	27	73	0
Longtail stingray	WRA	Non-QMS	71	5	95	0
Slender smooth hound	SSH	Non-QMS	64	35	65	0
Longnose velvet dogfish	CYP	Non-QMS	57	56	44	0
Bronze whaler	BWH	Non-QMS	44	56	44	0
Hammerhead shark	HHS	Moratorium	41	79	20	0
Plunket's shark	PLS	Non-QMS	35	37	63	0
Stingrays	STR	Non-QMS	30	2	98	0
Other skates	OSK	Non-QMS	26	25	75	0
Purple chimaera	CHG	Non-QMS	25	34	66	0
Sixgill shark	HEX	Non-QMS	21	10	90	0
Owston's dogfish	CYO	Non-QMS	17	40	60	0
Cimaeras	CHI	Non-QMS	15	23	77	0
Prickly dogfish	PDG	Non-QMS	10	31	69	0
Deepwater spiny skate	DSK	Non-QMS	6	7	93	0
Portuguese dogfish	CYL	Non-QMS	5	94	6	0
Blind electric ray	BER	Non-QMS	4	10	90	0
Catsharks	CSH	Non-QMS	4	56	44	0
Longnose deepsea skate	PSK	Non-QMS	3	82	18	0
Ghost sharks	HYD	Non-QMS	3	81	19	0
Brown chimaera	CHP	Non-QMS	2	64	36	0
Rays	RAY	Non-QMS	2	0	100	0
Sharpnose sevengill shark	HEP	Moratorium	2	24	76	0
Pacific spookfish	RCH	Non-QMS	2	89	11	0
Longtail skate	LSK	Non-QMS	1	0	100	0
Dawson's cat shark	DCS	Non-QMS	1	44	56	0
Pelagic stingray	DAS	Non-QMS	1	0	100	0

Table 10.4: TACCs and 2016–17 landings (tonnes) of the eleven chondrichthyans managed under the QMS. Also shown are the date of entry of each species into the QMS, and date of addition to Schedule 6 of the Fisheries Act that allows release of fish into the sea. Source: Monthly Harvest Returns (Ministry for Primary Industries 2017, Fisheries New Zealand 2018). Note: Landings differ from the catches in previous table because the latter include discards and releases, and came from a different source.

Species	Code	TACC (tonnes)	2016-17 landings	Entry into QMS	Addition to Schedule 6
Spiny dogfish	SPD	12660	5112	2004	2004
School shark	SCH	3436	2852	1986	2013
Rough skate	RSK	1986	1836	2003	2003
Rig	SPO	1966	1417	1986	2012
Dark ghost shark	GSH	3047	1382	1998	
Elephantfish	ELE	1304	1326	1986	
Pale ghost shark	GSP	1780	926	1999	
Smooth skate	SSK	849	827	2003	2003
Blue shark	BWS	1860	122	2004	2004
Mako shark	MAK	200	38	2004	2004
Porbeagle shark	POS	110	27	2004	2004

Francis & Sutton (2012) found a highly significant association between the numbers of basking sharks caught and vessel nationality in each of the three main fishery areas. This was due to relatively large numbers of sharks being caught by Japanese-owned trawlers in the late 1980s and early 1990s. Other operational fleet variables and environmental variables examined were not correlated with shark catch rates. Reasons for the high catch rates by Japanese trawlers are unknown, but may relate to targeting of the sharks for their liver oil and fins, or a relatively high abundance of sharks in the late 1980s and early 1990s (Francis & Sutton 2012).

Since 2010, protected species bycatch has been reported on Non-Fish / Protected Species Catch Returns (NF/PSCR). Over the six years 2011 to 2016, 59 basking sharks were reported on those forms (Francis 2017b). At an average weight of 3–4 t per shark (estimated weights from observers), this represents about 177–236 t of total catch, or 30–39 t per year. Few sharks were returned to the sea alive, and even fewer are likely to have survived their release.

WHITE POINTER SHARK

White pointer shark captures were reported from throughout mainland New Zealand and as far south as the Auckland Islands, but not from around the other outlying islands (Francis & Lyon 2012, Francis 2017a). Since 2008, 53 white pointer sharks have been reported caught by fishers, including 36 caught in set nets (Francis 2017a). Three small regions (Great Exhibition Bay (GEB), Taranaki (TAR) and Foveaux Strait (FOV)) accounted for 89% of the 36 white pointer sharks reported caught by set net vessels, but only 20% of the length of net set. Overall, 69% of sharks reported by fishers were said to be alive and in good condition, but the survival of live sharks after release is unknown.

Tagging and genetic studies have shown that New Zealand and eastern Australian white pointer sharks comprise a single stock (Duffy et al. 2012, Francis et al. 2015a). A close-kin genetics study which included New Zealand white pointer shark tissue samples estimated that the total population size of this stock of was 5460 (uncertainty range 2909 – 12 802) including 750 adults (uncertainty range 470 to 1030) (Bruce et al. 2018, Hillary et al. 2018). The trend in abundance was not significantly different from zero. However, there was evidence for a slight decline over the 2000s (i.e. strong evidence against a high upward trend) (Bruce et al. 2018).

WHALE SHARK

No captures of whale sharks have been reported by fishers or observers in New Zealand waters (Francis & Lyon 2012). However, a single individual was caught by a coastal trawler off South Canterbury in the late 1970s (as communicated to C. Duffy in Duffy 2005). This is exceptional, as whale sharks are typically only seen in north-eastern North Island waters during summer and are rare (Duffy 2002).

DEEPWATER NURSE SHARK (SMALLTOOTH SANDTIGER SHARK)

Deepwater nurse sharks have been reported frequently by fishers and observers from along the edge of the continental shelf between Otago Peninsula and south of the Snares Islands (Francis & Lyon 2012). Clusters of records are also available from the Chatham Islands, and off Banks Peninsula and Farewell Spit. However, the southern limit of the known distribution of deepwater nurse sharks in New Zealand is a line from Cape Kidnappers in Hawke Bay to Cape Egmont. Given that most of the records are from south of that range, and that many ODO weights were implausibly small, most records of this species are erroneous, probably owing to use of an incorrect species code. Plausible commercial and observer database records of deepwater nurse shark captures include three from FMA 2 and one from the Louisville Seamount Chain (Francis & Lyon 2012).

There are other published records of deepwater nurse sharks being caught in set nets off New Plymouth (Stewart 1997, Fergusson et al. 2008), trawl in Hawke Bay, and by the NIWA research trawl vessel *Tangaroa* on the Norfolk Ridge (Garrick 1974, Stewart 1997, Fergusson et al. 2008), confirming that the species is occasionally caught in northern waters. Duffy (2005) cited anecdotal information that deepwater nurse sharks were 'not uncommon' bycatch in a set-net fishery operating around White Island and Volkner Rocks in the eastern Bay of Plenty, but noted that this fishery had ceased. Duffy (2005) and Fergusson et al. (2008) also reported the capture of deepwater nurse sharks from the same location for display at Kelly Tarlton's Sea Life Aquarium from the mid-1980s to the early 2000s, but all of the sharks died and the practice was discontinued.

SPINETAIL DEVILRAY AND MANTA RAY

Spinetail devilrays and manta rays occur mainly in north-eastern North Island waters during summer (Duffy & Abbott 2003, Francis & Jones 2016). Most if not all mobulid rays reported caught in commercial fisheries were likely to have been spinetail devilrays (Paulin et al. 1982); no manta rays have been confirmed caught in New Zealand waters (Duffy 2005, Jones & Francis 2012). However, it is possible that manta rays are occasionally caught in purse seines along the north-east coast of North Island although observer coverage between 2005 and 2014 in FMA 1 skipjack tuna (0–31.8% per year) and mackerel purse seine fisheries (0–25.8% per year) have not confirmed any captures.

All commercial and observer records of mobulid rays were from the northern North Island in FMAs 1 and 9, and most records came from purse seine vessels (Francis & Lyon 2012, Jones & Francis 2012, Francis & Jones 2016). Most observer records were from the edge of the continental shelf between the Bay of Islands and Great Barrier Island. Commercial purse seine records are available from the eastern Bay of Plenty, and there are a few commercial and observer records from the North Taranaki Bight. Three devilrays have been reported caught on surface longlines, mainly near the 1000 m depth contour. Observer and commercial records were not available before 2001–02, although devilray bycatch in purse seine catches was documented between 1975 and 1981 by Paulin et al. (1982). All observed devilrays were returned to the sea by fishers. The three rays caught on surface longlines were alive when retrieved, but the life status of rays caught in purse seines was not recorded. Over the four fishing years 2010–11 to 2013–14, 153 spinetail devilrays were reported on Non-Fish / Protected Species Catch Returns. At an average weight of about 125 kg per ray (observer estimated weights), this represents about 19.1 t of total catch, or about 4.8 t per year.

Four out of seven rays (57%) released from purse seine nets died (Francis & Jones 2016), suggesting a high post-release mortality rate; however, more recent tagging has shown a reduced mortality rate (M. Francis unpublished data). Some released rays have travelled north to Vanuatu and south of Fiji, indicating that they make a seasonal migration between New Zealand and the tropics.

OCEANIC WHITETIP SHARK

The oceanic whitetip shark is a tropical species that enters northern New Zealand waters only in summer, and possibly only in summers that are warmer than normal (Francis et al. 1999b). Only 19 observer and two commercial fishery records are known (one of which occurred in both datasets) (Francis & Lyon 2014). All records came from surface longlines set in the Kermadec Fisheries Management Area or off the north-eastern coast of North Island. Most (84%) of the observed sharks were alive when hauled to the vessel, and about half were processed in some way with the remainder being discarded (those captures pre-dated protection of the species in 2013). Given the low commercial reporting rate (1 out of 19 observed sharks)

and the low observer coverage of domestic surface longliners, the interaction of the surface-longline fisheries with oceanic whitetips is considered substantially underestimated (Ministry for Primary Industries 2012, Francis & Lyon 2014).

10.4.3 NON-QMS SPECIES

More than 50 species of Non-QMS chondrichthyans are known to be caught by fishers in New Zealand waters, but records of non-QMS chondrichthyans catches are not believed reliable (due to identification issues and a limited set of species required to be reported).

Inshore rays and sharks are caught by a variety of fishing methods. Closures of strips of inshore waters to set netting and trawling to protect Hector's and Māui dolphin on the north-west coast of North Island and around much of South Island may have benefitted shark and ray species that occur there, and their habitats and nursery areas. However most of these species are highly vulnerable to trawl, set net and bottom longline, and have nurseries in shallow coastal waters and harbours that are still fished by set nets and longline, and to a lesser extent trawls. Little is known about the fishery interactions of these species (but for an analysis of hammerhead shark captures see Francis 2010). Similarly, there is little information on the biological productivity of most of the species, but many (all of the rays and thresher shark) have very low reproductive output (a few young per year) and are therefore highly susceptible to overfishing.

Deepwater chondrichthyans are caught incidentally in deepwater trawl tows, some species in considerable quantities (Blackwell 2010). Seven species of squaloid deepwater sharks, shovelnose dogfish, Baxter's dogfish, lucifer dogfish, Owston's dogfish, longnose velvet dogfish, leafscale gulper shark, and seal shark commonly occur over the middle and lower continental slope in depths greater than 600 m. Shovelnose dogfish has a wider distribution, as it also occurs on the upper and middle slope (400–600 m in depth). These seven shark species are commonly taken as bycatch in the middle depths and deepwater fisheries for hoki, orange roughy, and oreos. They are either discarded at sea, or processed for their fins and/or livers (Blackwell 2010).

Historical data are available from the Fisheries NZ Observer Programme (Figure 10.2), but coverage of the distribution of deepwater sharks has been unrepresentative. A critical issue when using observer data to monitor the abundance and biology of deepwater sharks is species identification: many deepwater sharks are superficially similar and difficult to distinguish, so they have often been mis-identified or lumped under generic codes by commercial fishers. With the recent availability of good species identification guides (McMillan et al. 2011a, 2011b, 2011c), observers and fishers now have the tools to identify deepwater sharks accurately to species level. In order to test current observer identification accuracy, McMillan et al. (2018) compared observer identifications of six species of deepwater sharks with NIWA identifications made using images and DNA analyses of tissue samples collected from the same specimens. Of 331 observer identifications, 302 (91%) were confirmed by NIWA; most of the incorrect identifications came from a single observer. It was concluded that observer identification of deepwater sharks is generally accurate.

Some species that are not caught or reported in quantities sufficient to be included in Table 10.2 may also be vulnerable to overfishing. These include endemic species with limited geographic and/or depth ranges that overlap in space with the operations of deepwater trawlers, for example Dawson's catshark (Francis 2006), and some of the rarer deepwater skates and chimaeras. Their low catch weights probably reflect their rarity.

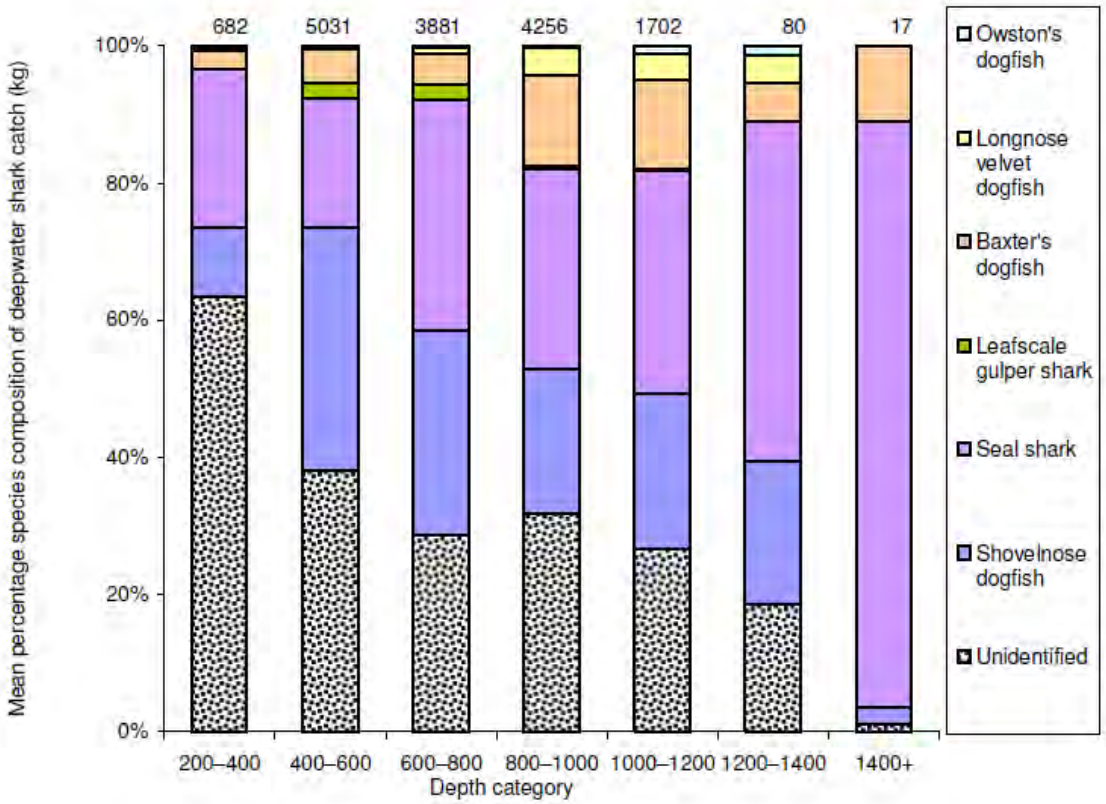


Figure 10.2: Mean catch composition of deepwater chondrichthyans reported from the Observer Programme database, all years 2001–02 to 2005–06, by major depth category (number of observations shown above bars). Source: Blackwell (2010).

10.5 INDICATORS AND TRENDS

RISK ASSESSMENT AND THREATENED SPECIES CLASSIFICATION

One of the objectives of the 2013 NPOA-Sharks was to establish a risk-based approach to prioritising management actions. MPI hosted a workshop in November 2014 that produced a qualitative (Level 1) risk assessment (RA) for all New Zealand chondrichthyan taxa (except for species with uncertain taxonomy) from commercial fishing (Ford et al. 2015). This was updated in 2017 (Ford et al. 2018).

The qualitative RA used a modified Scale Intensity Consequence Analysis approach. Before the workshops, data on catches, effort, distribution, abundance, and biological productivity were collated for all species and summarised to inform the RA. An expert panel then scored the relative risk to each taxon from commercial fishing, based on fishing information from the last five years, on an EEZ-wide scale. This process scored intensity and consequence of the fishery to the shark taxa, and the rationales for the scores were documented. These intensity and consequence scores were then multiplied together to get a total risk score (Ford et al. 2018). Results were reported within the three management classes of chondrichthyans – QMS, Non-QMS and Protected species.

Six QMS species attained the equal highest total risk score (dark ghost shark, elephantfish, rig, rough skate, school shark and spiny dogfish). Plunket's shark was the highest scoring Non-QMS species and basking shark and spinetail devilray were the highest scoring protected species (Appendix 19.11.3). The panel considered that the available information did not indicate that commercial fishing is currently causing, or in the near future could cause, serious unsustainable impacts to any sharks, rays or chimaera population examined. However, out of the 50 taxa considered in detail, the panel had low confidence in the risk scores for three of 11 QMS species, 26 of 36 non-QMS taxa and all three protected species.

The risk assessment was designed to help prioritise actions to conserve chondrichthyans, noting that protected species are also given priority under the NPOA-Sharks (2013). The panel made several recommendations for high-risk or protected species regarding potential research options. These included better use of existing data, data grooming

or analysis to improve inputs to assessment scores, improved taxonomy and training to underpin identification of sharks, and collection of more biological information to increase understanding of productivity (especially the ability of a taxon to withstand and to recover from fishing impacts). The RA panel also stressed that, particularly where abundance indices are lacking, the consequence scale was more relevant to risk than the total risk score which was often dominated by the level of intensity (masking differences in potential consequence). Taxa with high consequence scores have low productivity or presumed low productivity. In such cases, more information may improve the scores or our confidence in them, but in the interim a more precautionary approach to management was recommended by the panel (Ford et al. 2018).

Recently, there have been two updated assessments of the threatened species status of New Zealand chondrichthyans. In 2016, the Department of Conservation carried out a review of their threatened species classification (Duffy et al. 2018). The conservation status of two taxa has worsened: white pointer shark was assessed as Nationally Endangered (previously assessed as Gradual Decline) and basking shark moved to Nationally Vulnerable (from Gradual Decline). Conversely, four taxa had an improved conservation status (Galapagos shark, Kermadec smooth hound, sixgill shark and southern sleeper shark). In 2017, the IUCN Redlist categories of New Zealand chondrichthyans were reviewed and a number of changes made; however, the results have not yet been finalised or published. In the wider South Pacific Ocean, Australian researchers (with New Zealand input) are carrying out an ecological risk assessment for deepwater chondrichthyans in the SPRFMO Convention Area. Productivity-Susceptibility Analysis (PSA) and Sustainability Assessment for Fishing Effects (SAFE) methods are being used to assess the potential vulnerability of deepwater chondrichthyans to demersal trawl, midwater trawl and demersal longline gears.

QMS SPECIES

In the last five years, standardised CPUE analyses have been carried out to monitor trends in the relative abundance of some stocks of 6 of the 11 QMS chondrichthyans species (rig, school shark, elephantfish, blue shark, porbeagle shark and mako shark) (Table 10.5). School shark in QMAs 3 and 5, and elephantfish in QMA 5, are declining; all other stocks that are monitored are stable or increasing in recent years.

For blue, porbeagle and mako sharks, other abundance indicators have been developed in addition to standardized CPUE. They include high-CPUE (the proportion of half-degree rectangles having unstandardised CPUE greater than a specified threshold); proportion-zeroes (the proportion of half-degree rectangles having zero reported catches in a fishing year); geometric mean index (the geometric mean of the species abundances in catches; proportion of males in the catch; and median lengths of males and females (Francis et al. 2014). None of the indicators for the period 2005–13 suggested that any of the three shark species were declining. In fact, some of the indicators suggested positive trends for all three species (Francis et al. 2014, Francis & Large 2017).

Trawl survey relative abundance indices are used to monitor the populations of rig, school shark, spiny dogfish, elephantfish, rough and smooth skates, and pale and dark ghost sharks (Table 10.5). For 20 out of 21 species/FMA combinations, abundance is stable or increasing in recent years; however pale ghost shark in FMA 4 has shown a downward trend.

PROTECTED SPECIES

Of the seven protected chondrichthyan species, only the basking shark has any form of population monitoring and that is limited to assessing trends in relative abundance from incidental captures. Observer-based unstandardised CPUE analyses of trawl catches in three trawl fisheries (East Coast South Island EC, West Coast South Island WC, and Southland–Auckland Island SA) are shown in Figure 10.3 (Francis & Sutton 2012). Inter-annual variation was large, with peak observer records occurring in 1987–92, 1997–2000 and 2003–05 depending on the region. Some years had very low or zero CPUE. Francis & Smith (2010) used Bayesian predictive hierarchical models to estimate catches and catch rates in the three trawl fisheries from observer data between 1994–95 and 2007–08. The predicted strike rates showed no overall trend since 1994–95 in any of the three areas. A total of 95 sharks were observed in 49 165 tows in the 14-year period, an overall unstandardised capture rate of 1.9 per 1000 tows. The overall predicted capture rate was 2.5 sharks per 1000 tows, with area-specific rates of 3.9 (EC), 2.0 (WC), and 1.9 (SA) per 1000 tows. The total predicted number of captures from 1987 to 2012 was 922 individuals with a CV of 19%. Predicted captures peaked in 1997–98 and then declined steadily to low numbers. Much of the recent decline in basking shark bycatch was probably attributable to a decline in fishing

effort of about 50% between 2002–03 and 2006–08 in the three areas (Francis & Smith 2010). However, unstandardised catch rates from observer data were much higher in 1988–92 than at any time since. Those high rates may be attributable to targeting by Japanese vessels (Francis & Sutton 2012). Raw observer CPUE values were updated to the 2015–16 fishing year by Francis (2017b). CPUE continued the patterns shown in Figure 10.3; i.e. CPUE was zero or close to it in EC and WC fisheries, and fluctuated around low levels in the SA fishery.

The very low (often zero) CPUE in EC and WC regions since 2006, and lack of large numbers and aggregations of basking sharks observed in Department of Conservation aerial surveys for dolphins around Banks Peninsula during the last decade (C. Duffy, DOC, pers. comm.), are cause for concern. There may not have been large aggregations of basking sharks in New Zealand waters since 1992. Whether such a long period without large aggregations is part of a long-term, natural cycle, or evidence of a decline in population abundance, cannot yet be determined (Francis & Smith 2010).

Table 10.5: Trends in abundance of QMS species monitored by standardised CPUE analysis and trawl surveys. Changes in trends through time are indicated by forward slashes, and multiple substocks or multiple indices within QMAs are separated by commas. Blanks, none or unreliable. Source: Ministry for Primary Industries (2017), Fisheries New Zealand (2018) unless otherwise indicated. 'Recent years' refers to the last five years, but may be longer for long time series. Time series that have not been updated in the last five years are not included.

CPUE indices		QMA 1	QMA 2	QMA 3	QMA 4	QMA 5	QMA 6	QMA 7	QMA 8	Source
Rig	SPO	Nil	Nil	Nil, Up				Nil, Up		
School shark	SCH	Up/Nil		Down	Nil	Down		Nil	Nil	
Elephantfish	ELE			Nil		Down		Up		
Blue shark	BWS	Up, Up, Up								
Porbeagle shark	POS	Up, Down								Francis & Large (2017)
Mako shark	MAK	Up, Up, Up								
Trawl survey indices				FMA 3	FMA 4	FMA 5	FMA 6	FMA 7		
Rig	SPO			Nil				Nil		
School shark	SCH			Nil				Nil		
Spiny dogfish	SPD			Nil	Nil			Nil		
Elephantfish	ELE			Nil				Up		
Rough skate	RSK			Up/Nil	Nil			Up/Nil		
Smooth skate	SSK			Up/Nil	Down/Nil			Down/Up		
Dark ghost shark	GSH			Up	Nil	Nil		Nil		
Pale ghost shark	GSP				Down	Up				
Legend:										
	Trend up in recent years									
	Stable in recent years									
	Trend down in recent years									
Blanks = none or unreliable										

NON-QMS SPECIES

Indicator analyses have been carried out recently for eight shark and chimaera species: carpet shark, Baxter's dogfish, seal shark, longnose velvet dogfish, Plunket's shark, leafscale gulper shark, shovelnose dogfish, and longnose spookfish (Francis et al. 2016). The indicators calculated were relative biomass, median shark length, and proportion of male sharks (all from trawl surveys); and distribution (proportion of half-degree rectangles having raw catch per unit effort greater than a specified threshold), proportion of half-degree rectangles having zero reported catches in a fishing year, species composition, concentration (a measure of whether fishing effort focuses on or avoids areas of high shark abundance), and nominal and standardised CPUE (all from commercial catch-effort or observer data). Because of data limitations, or non-applicability of the method, only a subset of the indicators could be applied to each species, and then only to a few FMAs. None of the species showed clear and consistent evidence of recent declines in abundance. However, estimated trends were often uncertain, inconsistent among indicators, based on indicators that may be unreliable (e.g. trawl survey biomass estimates for species that are not well surveyed), and based on too few indicators (only trawl survey indicators were available for five out of eight species). For a number of species, one or more indicators

showed signs of decline, and ongoing monitoring is recommended (Francis et al. 2016).

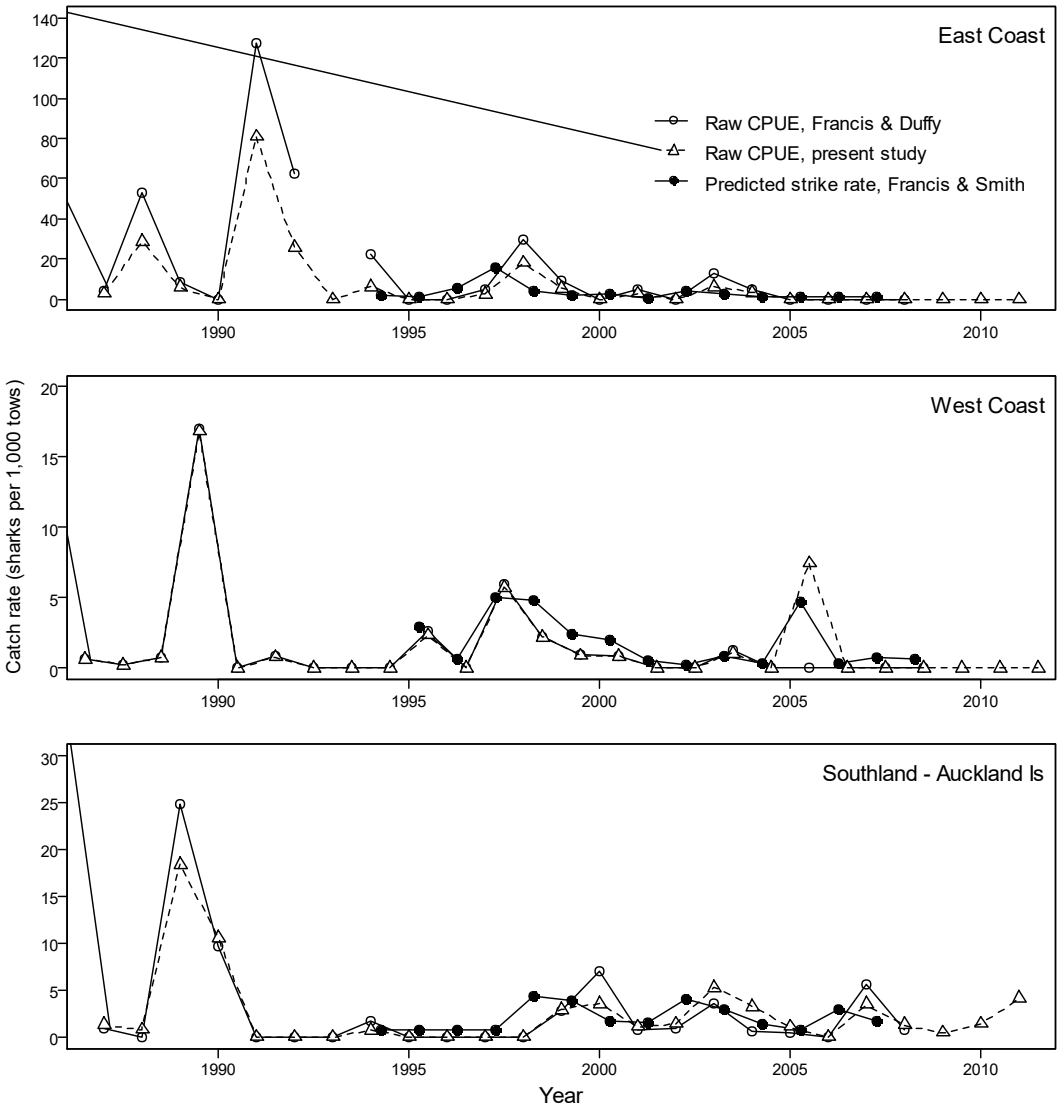


Figure 10.3: Basking shark catch rate indices for three fishery areas. For raw CPUE indices, years are calendar years for West Coast and July–June years (labelled as the greater of the two years) for East Coast and Southland-Auckland Is. For predicted strike rate, years are fishing years (labelled as the greater of the two years). Source: Francis & Sutton (2012).

10.6 REFERENCES

Anderson, O F (2013) Fish and invertebrate bycatch in New Zealand deepwater fisheries from 1990–91 until 2010–11. *New Zealand Aquatic Environment and Biodiversity Report No. 113*. 57 p.

Bagley, N W; Ballara, S L; O'Driscoll, R L; Fu, D; Lyon, W (2013) A review of hoki and middle depth summer trawl surveys of the Subantarctic, November December 1991–1993 and 2000–2009. *New Zealand Fisheries Assessment Report 2013/41*. 63 p.

Blackwell, R G (2010) Distribution and abundance of deepwater sharks in New Zealand waters, 2000–01 to 2005–06. *New Zealand Aquatic Environment and Biodiversity Report No. 57*. 51 p.

Bonfil, R (1994) Overview of world elasmobranch fisheries. *FAO Fisheries Technical Paper 341*. FAO, Rome. 119 p.

Bräutigam, A; Callow, M; Campbell, I R; Camhi, M D; Cornish, A S; Dulvy, N K; Fordham, S V; Fowler, S L; Hood, A ; McClennen, C; Reuter, E L (2015) Global priorities for conserving sharks and rays: a 2015–2025 strategy. Global Sharks and Rays Initiative. 28 p.

Bruce, B; Bradford, R; Bravington, M; Feutry, P; Grewe, P; Gunasekera, R; Harasti, D; Hillary, R; Patterson, T (2018). A national assessment of the status of white sharks. National Environmental Science Programme, Marine Biodiversity Hub, CSIRO, 41 p.

Camhi, M; Fowler, S; Musick, J; Bräutigam, A; Fordham, S (1998) Sharks and their relatives. Ecology and conservation. *Occasional paper of the IUCN Species Survival Commission 20*. 39 p.

Campana, S E; Joyce, W; Manning, M J (2009) Bycatch and discard mortality in commercially caught blue sharks *Prionace glauca* assessed using archival satellite pop-up tags. *Marine Ecology Progress Series 387*: 241–253.

Clarke, S; Coelho, R; Francis, M; Kai, M; Kohin, S; Liu, K-M; Simpfendorfer, C; Tovar-Avila, J; Rigby, C; Smart, J (2015) Report of the Pacific Shark Life History Expert Panel Workshop, 28-30 April 2015. Western Central Pacific Fisheries Commission Scientific Committee eleventh regular session No. WCPFC-SC11-2015/EB-IP-13. 111 p.

Clarke, S C; Harley, S J; Hoyle, S D; Rice, J S (2012) Population trends in Pacific oceanic sharks and the utility of regulations on shark finning. *Conservation Biology 27*: 197–209.

Clarke, S C; McAllister, M K; Milner-Gulland, E J; Kirkwood, G P; Michielsens, C G J; Agnew, D J; Pikitch, E K; Nakano, H; Shivji, M S (2006) Global estimates of shark catches using trade records from commercial markets. *Ecology Letters 9*: 1115–1126.

Clarke, S C; Smith, N; Lyon, W; Francis, M (2017) Western and Central Pacific Fisheries Commission shark post-release mortality tagging studies. WCPFC Scientific Committee 13th regular session No. WCPFC-SC13-2017/EB-IP-06. 43 p.

Dent, F; Clarke, S (2015) State of the global market for shark products. *FAO Fisheries and Aquaculture Technical Paper 590*. FAO, Rome. 188 p.

Duffy, C A J (2002) Distribution, seasonality, lengths, and feeding behaviour of whale sharks (*Rhincodon typus*) observed in New Zealand waters. *New Zealand Journal of Marine and Freshwater Research 36*: 565–570.

Duffy, C (2005) Rationale and background information justifying protection of additional marine fishes under the Wildlife Act. Department of Conservation. 14 p. (Unpublished report held by the Department of Conservation, Wellington.)

Duffy, C A J; Abbott, D (2003) Sightings of mobulid rays from northern New Zealand, with confirmation of the occurrence of *Manta birostris* in New Zealand waters. *New Zealand Journal of Marine and Freshwater Research 37*: 715–721.

Duffy, C; Francis, M; Dunn, M; Finucci, B; Ford, R; Hitchmough, R; Rolfe, J (2018) Conservation status of New Zealand chondrichthyans (chimaeras, sharks and rays), 2016. *New Zealand threat classification series 23*. 13 p.

Duffy, C A J; Francis, M P; Manning, M; Bonfil, R (2012) Regional population connectivity, oceanic habitat, and return migration revealed by satellite tagging of white sharks, *Carcharodon carcharias*, at New Zealand aggregation sites. *In*: Domeier, M.L. (ed.). Global perspectives on the biology and life history of the white shark, pp. 301–318. CRC Press, Boca Raton, USA.

Dulvy, N K; Sadovy, Y; Reynolds, J D (2003) Extinction vulnerability in marine populations. *Fish and Fisheries 4*: 25–64.

Dulvy, N K; Simpfendorfer, C A; Davidson, L N; Fordham, S V; Bräutigam, A; Sant, G; Welch, D J (2017) Challenges and priorities in shark and ray conservation. *Current Biology, 27(11)*: R565–R572.

Ellis, J R; McCully Phillips, S R; Poisson, F (2017) A review of capture and post-release mortality of elasmobranchs. *Journal of Fish Biology 90*: 653–722.

Fergusson, I K; Graham, K J; Compagno, L J V (2008) Distribution, abundance and biology of the smalltooth sandtiger shark *Odontaspis ferox* (Risso, 1810) (Lamniformes: Odontaspididae). *Environmental Biology of Fishes 81*: 207–228.

Fisheries New Zealand (2018) Fisheries Assessment Plenary May 2018: stock assessments and stock status. Compiled by the Fisheries Science and Information Group, Fisheries New Zealand, Wellington, New Zealand. 1660 p.

Ford, R B; Galland, A; Clark, M R; Crozier, P; Duffy, C A J; Dunn, M; Francis, M P; Wells, R (2015) Qualitative (Level 1) Risk Assessment of the impact of commercial fishing on New Zealand Chondrichthyans. *New Zealand Aquatic Environment and Biodiversity Report. No. 157*. 111 p.

Ford, R B; Francis, M P; Holland, L; Clark, M R; Duffy, C A J; Dunn, M R; Jones, E; Wells, R (2018) Qualitative (Level 1) risk assessment of the impact of commercial fishing on New Zealand

chondrichthyans: an update for 2017. *New Zealand Aquatic Environment and Biodiversity Report No. 201*. 103 p.

Francis, M; Large, K (2017) Updated abundance indicators for New Zealand blue, porbeagle and shortfin mako sharks. CCSBT Ecologically Related Species Working Group 12th meeting CCSBT-ERS/1703/14. WCPFC Scientific Committee 13th regular session WCPFC-SC13-2017/SA-IP-13. 19 p.

Francis, M; Ritchie, P (2016) Genetic studies of New Zealand's protected fish species. NIWA client report No. WLG2016-31. 33 p.

Francis, M P (2006) Distribution and biology of the New Zealand endemic catshark, *Halaelurus dawsoni*. *Environmental Biology of Fishes 75*: 295–306.

Francis, M P (2010) Review of research and observer data on hammerhead sharks (*Sphyrna zygaena*). Final Research Report for Ministry of Fisheries. 8 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Francis, M P (2015) Size, maturity and age composition of porbeagle sharks observed in New Zealand tuna longline fisheries. *New Zealand Fisheries Assessment Report 2015/16*. 30 p.

Francis, M P (2016a) Distribution, habitat and movement of juvenile smooth hammerhead sharks (*Sphyrna zygaena*) in northern New Zealand. *New Zealand Journal of Marine and Freshwater Research 50*: 506–525.

Francis, M P (2016b) Size, maturity and age composition of mako sharks observed in New Zealand tuna longline fisheries. *New Zealand Fisheries Assessment Report 2016/22*. 34 p.

Francis, M P (2017a) Bycatch of white sharks in commercial set nets. NIWA Client Report No. 2017113WN. 27 p.

Francis, M P (2017b) Review of commercial fishery interactions and population information for New Zealand basking shark. NIWA Client Report No. 2017083WN. 44 p.

Francis, M P; Clarke, S C; Griggs, L H; Hoyle, S D (2014) Indicator based analysis of the status of New Zealand blue, mako and porbeagle sharks. *New Zealand Fisheries Assessment Report 2014/69*. 109 p.

Francis, M P; Duffy, C (2002) Distribution, seasonal abundance and bycatch of basking sharks (*Cetorhinus maximus*) in New Zealand, with observations on their winter habitat. *Marine Biology 140*: 831–842.

Francis, M.P.; Duffy, C.; Lyon, W. (2015a). Spatial and temporal habitat use by white sharks (*Carcharodon carcharias*) at an aggregation site in southern New Zealand. *Marine and freshwater research 66*: 900–918.

Francis, M P; Griggs, L H; Baird, S J; Murray, T E; Dean, H A (1999a) Fish bycatch in New Zealand tuna longline fisheries. *NIWA Technical Report 55*. 70 p.

Francis, M P; Holdsworth, J C; Block, B A (2015b) Life in the open ocean: seasonal migration and diel diving behaviour of Southern Hemisphere porbeagle sharks (*Lamna nasus*). *Marine biology 162*: 2305–2323.

Francis, M P; Jones, E; Ó Maolagáin, C; Lyon, W S (2018a). Growth and reproduction of four deepwater sharks in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 196.*

Francis, M P; Jones, E G (2016) Movement, depth distribution and survival of spinetail devilrays (*Mobula japanica*) tagged and released from purse-seine catches in New Zealand. *Aquatic conservation: marine and freshwater ecosystems 27*: 219–236.

Francis, M P; Lyon, W S (2012) Review of commercial fishery interactions and population information for eight New Zealand protected fish species. NIWA client report WLG2012-64. 67 p.

Francis, M P; Lyon, W S (2013) Review of anthropogenic impacts other than fishing on cartilaginous fishes. *New Zealand Aquatic Environment and Biodiversity Report No. 107*. 17 p.

Francis, M P; Lyon, W S (2014) Review of commercial fishery interactions and population information for the oceanic whitetip shark, a protected New Zealand species. NIWA client report WLG2014-40. 15 p.

Francis, M P; Mace, J T (1980) Reproductive biology of *Mustelus lenticulatus* from Kaikoura and Nelson. *New Zealand Journal of Marine and Freshwater Research 14*: 303–311.

Francis, M P; Ó Maolagáin, C (2000) Age, growth and maturity of a New Zealand endemic shark (*Mustelus lenticulatus*) estimated from vertebral bands. *Marine and Freshwater Research 51*: 35–42.

Francis, M P; Ó Maolagáin, C (2016) Size, maturity and length composition of blue sharks observed in New Zealand tuna longline fisheries. *New Zealand Fisheries Assessment Report 2016/60*. 27 p.

Francis, M P; Ó Maolagáin, C; Lyon, W S (2018b). Growth and reproduction of carpet shark, common electric ray and blind electric ray in New Zealand waters. Draft New Zealand Fisheries Assessment Report.

Francis, M P; Roberts, J; MacGibbon, D J (2016) Indicator based analysis of the status of eight shark and chimaera species in New Zealand waters. *New Zealand Fisheries Assessment Report No. 2016/65*. 87 p.

Francis, M P; Smith, M H (2010) Basking shark (*Cetorhinus maximus*) bycatch in New Zealand fisheries, 1994–95 to 2007–08. *New Zealand Aquatic Environment and Biodiversity Report No. 49*. 57 p.

Francis, M P; Sutton, P (2012) Possible factors affecting bycatch of basking sharks (*Cetorhinus maximus*) in New Zealand trawl fisheries. NIWA client report WLG2012-48. 38 p.

Francis, M P; Worthington, C J; Saul, P; Clements, K D (1999b) New and rare tropical and subtropical fishes from northern New

Zealand. *New Zealand Journal of Marine and Freshwater Research 33*: 571–586.

Fu, D; Roux, M-J; Clarke, S; Francis, M; Dunn, A; Hoyle, S (2016) Pacific-wide sustainability risk assessment of bigeye thresher shark (*Alopias superciliosus*). NIWA client report 2016089WN. WCPFC Scientific Committee 13th regular session WCPFC-SC13-2017/SA-WP-11. 102 p.

Garrick, J A F (1974) First record of an odontaspidid shark in New Zealand waters. *New Zealand Journal of Marine and Freshwater Research 8*: 621–630.

Griggs, L H; Baird, S J (2013) Fish bycatch in New Zealand tuna longline fisheries 2006–07 to 2009–10. *New Zealand Fisheries Assessment Report 2013/13*. 73 p.

Hernández, S; Daley, R; Walker, T; Braccini, M; Varela, A; Francis, M P; Ritchie, P A (2015) Demographic history and the South Pacific dispersal barrier for school shark (*Galeorhinus galeus*) inferred by mitochondrial DNA and microsatellite DNA mark. *Fisheries Research 167*: 132–142.

Hillary, R M; Bravington, M V; Patterson, T A; Grewe, P; Bradford, R; Feutry, P; Gunasekera, R; Peddemors, V; Werry, J; Francis, M P; Duffy, C A J; Bruce, B D (2018) Genetic relatedness reveals total population size of white sharks in eastern Australia and New Zealand. *Nature Scientific Reports 8*: 2661.

Hutchinson, M; Itano, D; Muir J; Leroy, B; Holland, K (2013) Fishery interactions and post-release survival rates of silky sharks caught in purse seine fishing gear. Western Central Pacific Fisheries Commission, Scientific Committee Paper, WCPFC-SC9-2013/ EB-WP-12.

Hoyle, S D; Edwards, C T T; Roux, M-J; Clarke, S C; Francis, M P (2017) Southern Hemisphere porbeagle shark stock status assessment. WCPFC Scientific Committee 13th regular session No. WCPFC-SC13-2017/SA-WP-12 (rev. 2). 72 p.

Jones, E; Francis, M P (2012) Protected rays – occurrence and development of mitigation methods in the New Zealand tuna purse seine fishery. NIWA client report WLG2012-49, prepared for the Department of Conservation. 35 p.

Jones, E; Francis, M; Paterson, C; Morrison, M (2015) Habitats of particular significance for fisheries management: identification of threats and stressors to rig nursery areas. *New Zealand Aquatic Environment and Biodiversity Report No. 150*. 76 p.

Kyne, P M; Simpfendorfer, C A (2007) A collation and summarization of available data on deepwater chondrichthyans: biodiversity, life history and fisheries. A report prepared by the IUCN Shark Specialist Group for the Marine Conservation Biology Institute. 137 p.

Last, P R; Stevens, J D (2009) Sharks and rays of Australia. Second edition. CSIRO, Hobart. 644 p.

McMillan, P J; Francis, M P; James, G D; Paul, L J; Marriott, P J; Mackay, E; Wood, B A; Griggs, L H; Sui, H; Wei, F (2011a). New Zealand fishes. Volume 1: A field guide to common species caught by bottom and midwater fishing. *New Zealand Aquatic Environment and Biodiversity Report No. 68*. 329 p.

McMillan, P J; Francis, M P; Paul, L J; Marriott, P J; Mackay, E; Baird, S-J; Griggs, L H; Sui, H; Wei, F (2011b) New Zealand fishes. Volume 2: A field guide to less common species caught by bottom and midwater fishing. *New Zealand Aquatic Environment and Biodiversity Report No. 78*. 181 p.

McMillan, P J; Griggs, L H; Francis, M P; Marriott, P J; Paul, L J; Mackay, E; Wood, B A; Sui, H; Wei, F (2011c) New Zealand fishes. Volume 3: A field guide to common species caught by surface fishing. *New Zealand Aquatic Environment and Biodiversity Report No. 69*. 145 p.

McMillan, P J; Sutherland, J; Anderson, O (2018) Identification of six species of deepsea sharks sampled at sea by MPI observers, October 2016 to December 2017. *New Zealand Aquatic Environment and Biodiversity Report No 203*. 14 p.

Ministry for Primary Industries (2012) Protection of oceanic whitetip shark. Regulatory impact statement. Ministry for Primary Industries, Wellington. 5 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Ministry for Primary Industries (2013) National Plan of Action for the conservation and management of sharks 2013. Ministry for Primary Industries, Wellington. 32 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Ministry for Primary Industries (2017) Fisheries Assessment Plenary November 2017. Stock assessments and stock status. Compiled by the Fisheries Science Group, Ministry for Primary Industries, Wellington, New Zealand. 500 p.

Ministry of Fisheries (2008) New Zealand National Plan of Action for the conservation and management of sharks. Ministry of Fisheries, Wellington. 90 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Mollet, H F; Cliff, G; Pratt, H L; Stevens, J D (2000) Reproductive biology of the female shortfin mako, *Isurus oxyrinchus* Rafinesque, 1810, with comments on the embryonic development of lamnoids. *Fishery Bulletin 98*: 299–318.

Moyes, C D; Fragoso, N; Brill, R W; Musyl, M K (2006) Predicting postrelease survival in large pelagic fish. *Transactions of the American Fisheries Society 135*: 1389–1397.

Musyl, M K; Brill, R W; Curran, D S; Fragoso, N M; McNaughton, L M; Nielsen, A; Kikkawa, B S; Moyes, C D (2011) Postrelease survival, vertical and horizontal movements, and thermal habitats of five species of pelagic sharks in the central Pacific Ocean. *Fishery Bulletin 109*: 341–368.

O'Driscoll, R L; MacGibbon, D; Fu, D; Lyon, W; Stevens, D W (2011) A review of hoki and middle-depth trawl surveys of the Chatham Rise, January 1992–2010. *New Zealand Fisheries Assessment Report 2011/47*. 72 p.

Paulin, C D; Habib, G; Carey, C L; Swanson, P M; Voss, G J (1982) New records of *Mobula japanica* and *Masturus lanceolatus*, and further records of *Luvaris imperialis* (Pisces: Mobulidae,

Molidae, Louvaridae) from New Zealand. *New Zealand Journal of Marine and Freshwater Research 16*: 11–17.

Pikitch, E K; Camhi, M D; Babcock, E A (2008) Introduction. *In*: Camhi, M D; Pikitch, E K; Babcock, E A (Eds.), Sharks of the open ocean: biology, fisheries and conservation. pp 3–13. Blackwell Publishing, Oxford, United Kingdom.

Rice, J; Harley, S (2012a) Stock assessment of oceanic whitetip sharks in the western and central Pacific Ocean. Western Central Pacific Fisheries Commission Scientific Committee eighth regular session No. WCPFC-SC8-2012/SA-WP-06 Rev 1. 53 p.

Rice, J; Harley, S (2012b) Stock assessment of silky sharks in the western and central Pacific Ocean. Western Central Pacific Fisheries Commission Scientific Committee eighth regular session No. WCPFC-SC8-2012/SA-WP-07 Rev 1. 53 p.

Simpfendorfer, C A; Kyne, P M (2009) Limited potential to recover from overfishing raises concerns for deep-sea sharks, rays and chimaeras. *Environmental Conservation 36*: 97–103.

Smith, S E; Au, D W; Show, C (1998) Intrinsic rebound potentials of 26 species of Pacific sharks. *Marine and Freshwater Research 49*: 663–678.

Stewart, A (1997) Toothy sand tiger. *Seafood New Zealand October 1997*: 91–92.

Walker, T I (2005) Reproduction in fisheries science. *In*: Hamlett, W C (Ed.), Reproductive biology and phylogeny of Chondrichthyes. Pp 81–127. Science Publishers, Enfield, USA.

Worm, A; Davis, B; Kettemer, L; Ward-Paige, C A; Chapman, D; Heithaus, M R; Kessel, S T; Gruber, S H (2013) Global catches, exploitation rates, and rebuilding options for sharks. *Marine Policy 40*: 194–204.

Wynne-Jones, J; Gray, A; Hill, L; Heinemann, A (2014) National panel survey of marine recreational fishers 2011–12: harvest estimates. *New Zealand Fisheries Assessment Report 2014/67*. 139 p.

THEME 3: BENTHIC IMPACTS

11 BENTHIC (SEABED) IMPACTS

Status of chapter	This chapter has been partially updated for AEBAR 2018. Updated figures and tables have been identified in their captions.
Scope of chapter	This chapter outlines the main effects of mobile bottom (or demersal) fishing gear on seabed habitats and communities. All trawl gears contacting the seabed and shellfish dredges are included. Danish seines and more or less static methods like bottom longline and potting are excluded in this version, as are fisheries outside the EEZ.
Area	All of the New Zealand Territorial Sea (TS) and Exclusive Economic Zone (EEZ). There will be some relevance for out-of-zone bottom trawl fisheries.
Focal localities	Areas that are fished more frequently and habitats that are more sensitive to disturbance are likely to be most affected; areas that are closed to bottom impacting methods will not be directly affected. Bottom trawling offshore is most intense on the western flanks and to the south-west of the Chatham Rise, the edge of the Stewart-Snares Shelf, south-east of the Auckland Islands Shelf, and off the north-west coast of the South Island. In coastal waters shallower than 250 m, trawling is most intense along the east coast of North Island, south of East Cape, and in Tasman and Golden Bays. Shellfish dredges probably have the greatest effect but their footprint is much smaller than that of bottom trawl fisheries and generally in shallow waters.
Key issues	Habitat modification, potential loss of biodiversity, potential loss of benthic productivity, potential modification of important breeding or juvenile fish habitat leading to reduced fish recruitment.
Emerging issues	Potential for effects on habitats of particular significance to fisheries management (HPSFM). The need for (and opportunities presented by) better spatial information on inshore fisheries from finer scale reporting of fishing locations (including logbooks). Cumulative effects and interactions with other stressors (including existing effects, especially in the coastal zone, and climate change).
Fisheries New Zealand research (current)	DAE2018-04 *Taxonomic identification of benthic samples*; ZBD2016-11 *Quantifying benthic biodiversity across natural gradients*; ZBD2012-03 *Chatham Rise Benthos – Ocean Survey*; ZBD2014-10 *Benthic biodiversity*; BEN2014-03 *Monitoring Recovery of Benthic Fauna in Spirits Bay*; ZBD2017-05 Post voyage analyses – Spirits Bay; BEN2018-01 *Monitoring of trawl footprint (including coastal);* BEN2014-02 *Monitoring recovery of benthic fauna on the Graveyard complex*.
NZ government research (current)	MBIE programme: Sustainable Seas COIX1515 Sustainable Seas Ko Nga Moana Whakauka.
Related chapters/issues	Habitats of particular significance for fisheries management (HPSFM), marine environmental monitoring, marine mining/sand extraction, land-based effects.

11.1 CONTEXT

For the purpose of this document, the term 'mobile bottom fishing methods' includes all types of trawl gear that are used in contact with the seabed as well as shellfish dredges of various designs and Danish seines. Relative to the information about trawls and dredges there is little information available about the distribution and effects of Danish seining, so Danish seining is not considered in detail. The benthic effects of other methods of catching fish on or near the seabed that do not involve deliberately towing or dragging fishing gear across the seabed are thought to be considerably less than those of the mobile methods (although they are not always negligible) and these methods are not considered in this document.

Trawls and dredges are used to catch a relatively high proportion of commercial landings in New Zealand and such methods can represent the only effective and economic way of catching some species. However, the resulting disturbance to seabed habitats and communities

may have consequences for biodiversity and ecosystem services, including fisheries and other secondary production. The guiding sections of the Fisheries Act 1996 for managing the effects of fishing, including benthic effects, are s.8(2)(b), which specifies that 'ensuring sustainability' (s.8(1)) includes 'avoiding, remedying, or mitigating any adverse effects of fishing on the aquatic environment' and s.9, which specifies a principle that 'biological diversity of the aquatic environment should be maintained'. Also potentially relevant is the principle in s.9 that 'habitat of particular significance for fisheries management should be protected' (see the chapter on Habitats of Particular Significance for Fisheries Management for more details).

One approach to managing the effects of mobile bottom fishing methods is through the use of spatial controls. A wide variety of such controls apply in New Zealand waters (Figure 11.1). Some of these controls were introduced specifically to manage the effects of trawling, shellfish dredging, and Danish seining in areas or habitats considered sensitive to such disturbance (e.g., the bryozoan beds off Separation Point, between Golden and Tasman Bays, and the sponge-dominated fauna to the north of Spirits and Tom Bowling Bays in the far north). Other closures exist for other reasons but have the effect of protecting certain areas of seabed from disturbance by mobile bottom fishing methods. These include no-take marine reserves, pipeline and power cable exclusion zones, and areas set aside to protect marine mammals (e.g., see Figure 11.2 for areas where trawling is prohibited, Figure 11.3 for areas where gear and seasonal restrictions apply, and Figure 11.4 for areas related to marine reserves and marine farms, all mapped as at 2018). Marine reserves provide marine protection in a range of habitats within the Territorial Sea. Although marine reserves provide a higher level of protection by prohibiting all extractive activities, most tend to be small. New Zealand's 34 marine reserves protect about 7.6% of New Zealand's Territorial Sea; however, 99% of this is in two marine reserves in the territorial seas around offshore island groups in the far north and far south of New Zealand's EEZ (Helson et al. 2010). Until 2000, most closures that had the effect of protecting areas of seabed from disturbance by trawling and dredging were in the Territorial Sea.

In the Exclusive Economic Zone, 18 seamount closures were established in 2001 to protect representative underwater topographic features from bottom trawling and dredging (Brodie & Clark 2003; see Figure 11.1). These areas include 25 features, including 12 large seamounts more than 1000 m high, covering 2% (81 000 km^2) of the EEZ. The seamount areas are closed to all types of trawling and dredging. In 2006, members of the fishing industry proposed the closure of about 31% of the EEZ to bottom trawling and dredging in Benthic Protection Areas (BPAs), including the existing seamount closures. The design criteria for the BPAs were they should be large, relatively unfished, have simple boundaries, and be broadly representative of the marine environment. After a consultation process, a substantially revised package of BPAs (including three additional areas totaling 13 887 km^2, 10 additional active hydrothermal vents, and 35 topographic features) that complemented the existing seamount closures was implemented by regulation in 2007 (Helson et al. 2010; Figure 11.1). BPAs cover about 1.1 million km^2 (30%) of New Zealand's EEZ and are closed to trawling on or close to the bottom. Midwater trawling well off the bottom is permitted in the BPAs if two observers are on board and an approved net monitoring system is used. Much of the seabed within BPAs is below trawlable depth (maximum trawlable depth is about 1600 m) and all are outside the Territorial Sea. In combination, the seamount closures and the BPAs include: 28% of underwater topographic features (a term that includes underwater hills, knolls, and seamounts); 52% of seamounts over 1000 m high; and 88% of known active hydrothermal vents.

11.2 GLOBAL UNDERSTANDING

Concerns about the use of towed fishing gear on benthic habitats were first raised by fishermen in the fourteenth century in the UK (Lokkeborg 2005). They were worried about the capture of juvenile fish and the detrimental effects on food sources for harvestable fish. Despite this long history of concern, it is really only in the last 20 years that research efforts have focused strongly on the effects of mobile bottom fishing methods on benthic (seabed) communities, biodiversity, and production. This activity, combined with controversy around fishing effects, has spawned numerous reviews in the past 10 years that seek to summarise or synthesise the information (Jones 1992, Dayton et al. 1995, Jennings & Kaiser 1998, Watling & Norse 1998, Lindeboom & de Groot 1998, Auster & Langton 1999, Hall 1999, ICES 2000a, 2000b, Kaiser & de Groot 2000, NMFS 2002, NRC 2002, Dayton et al. 2002, Thrush & Dayton 2002, Lokkeborg 2005, Barnes & Thomas 2005, Clark & Koslow 2007).

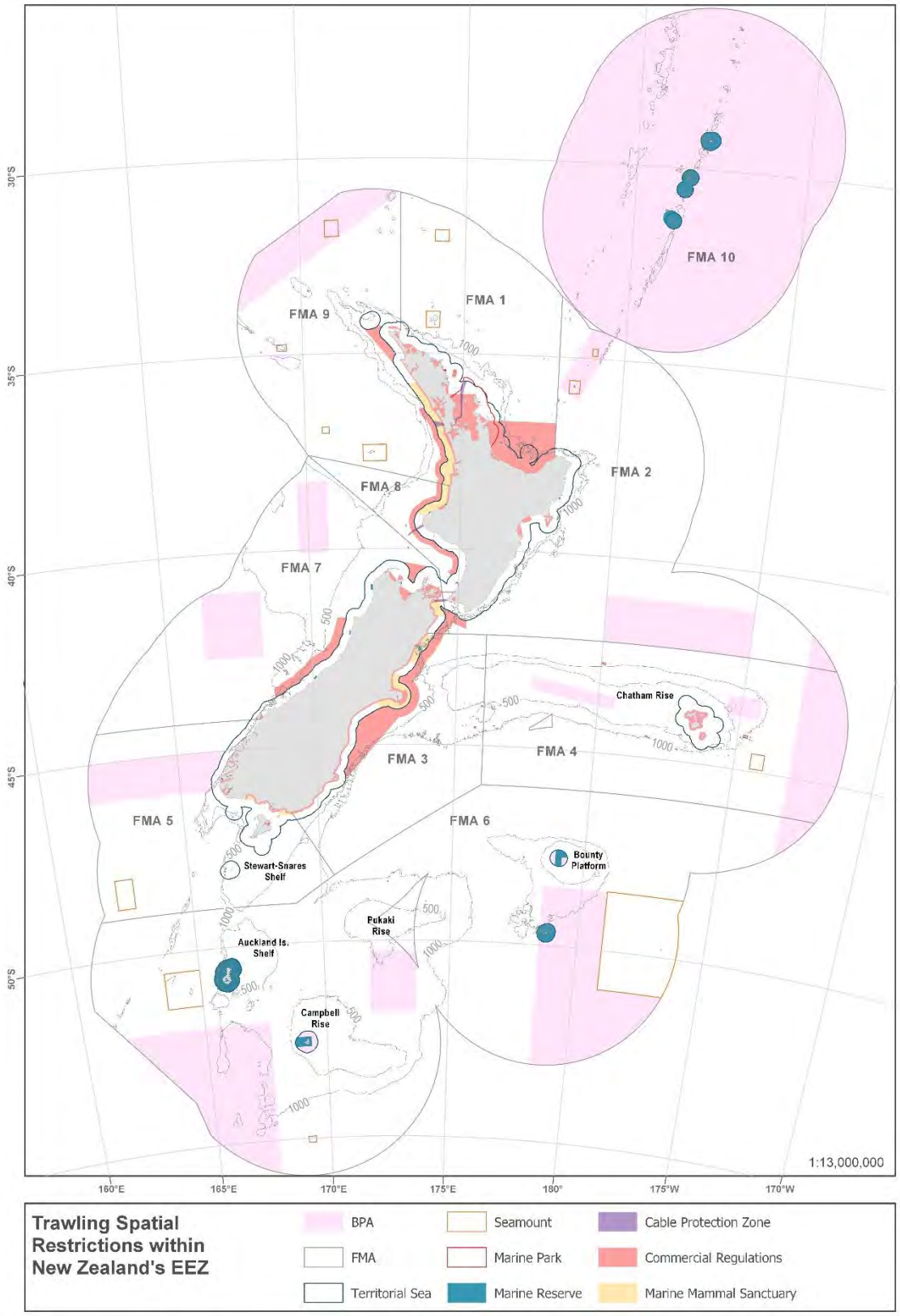


Figure 11.1: Map of the major spatial restrictions to trawling and Fisheries Management Areas (FMAs) within the outer boundary of the New Zealand EEZ. Vessels longer than 28 m may not trawl within the TS and additional restrictions are specified in the Fisheries (Auckland Kermadecs Commercial Fishing) Regulations 1986, the Fisheries (Central Area Commercial Fishing) Regulations 1986, the Fisheries (Challenger Area Commercial Fishing) Regulations 1986 the Fisheries (South East Area Commercial Fishing) Regulations 1986, and the Fisheries (Southland and Sub-Antarctic Areas Commercial Fishing) Regulations 1991. For more details of BPAs, see Helson et al. (2010).

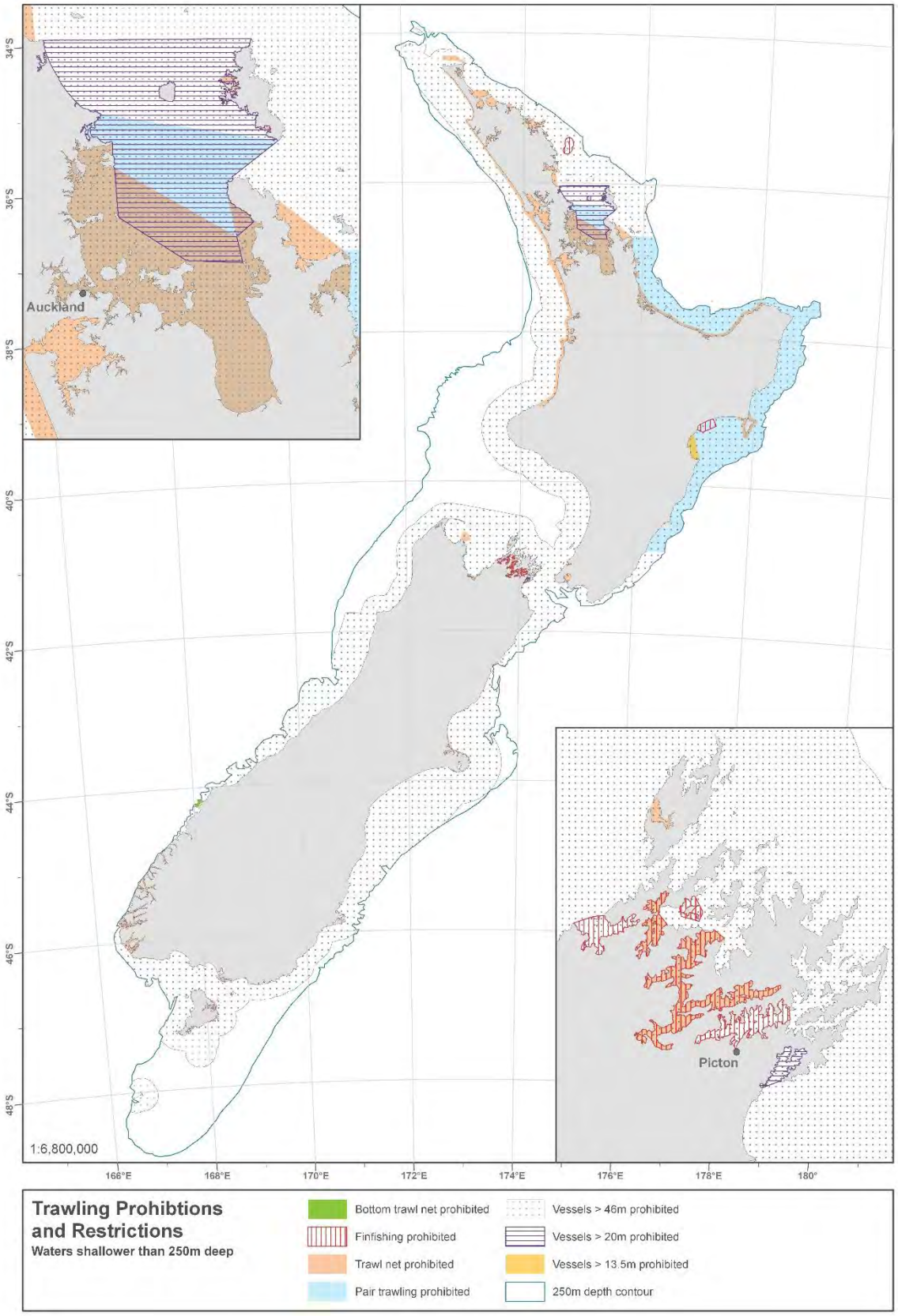


Figure 11.2: Areas showing where trawling is prohibited and other relevant restrictions apply in waters shallower than 250 m depth.

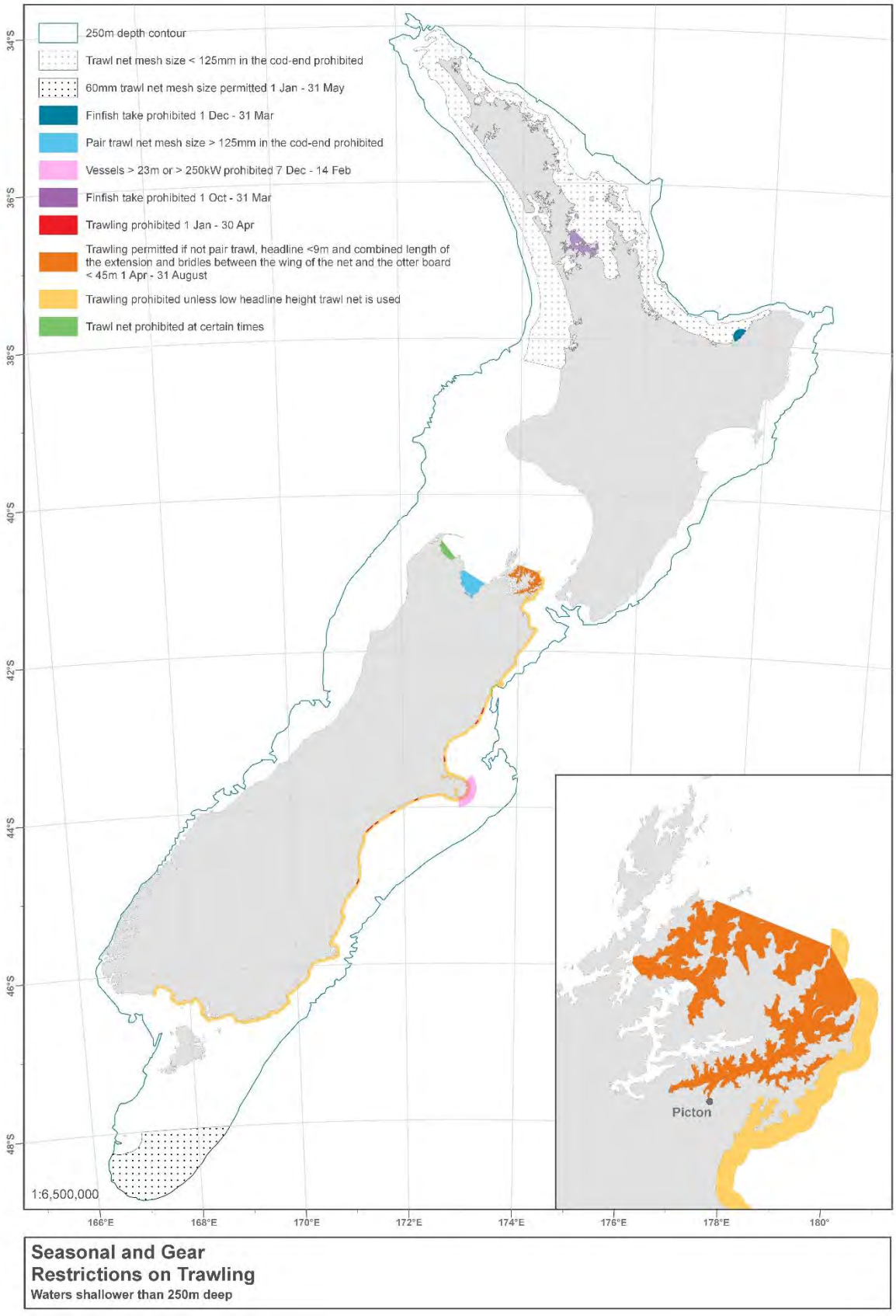


Figure 11.3: Areas where gear and seasonal restrictions apply to the use of trawl gear, in waters shallower than 250 m depth.

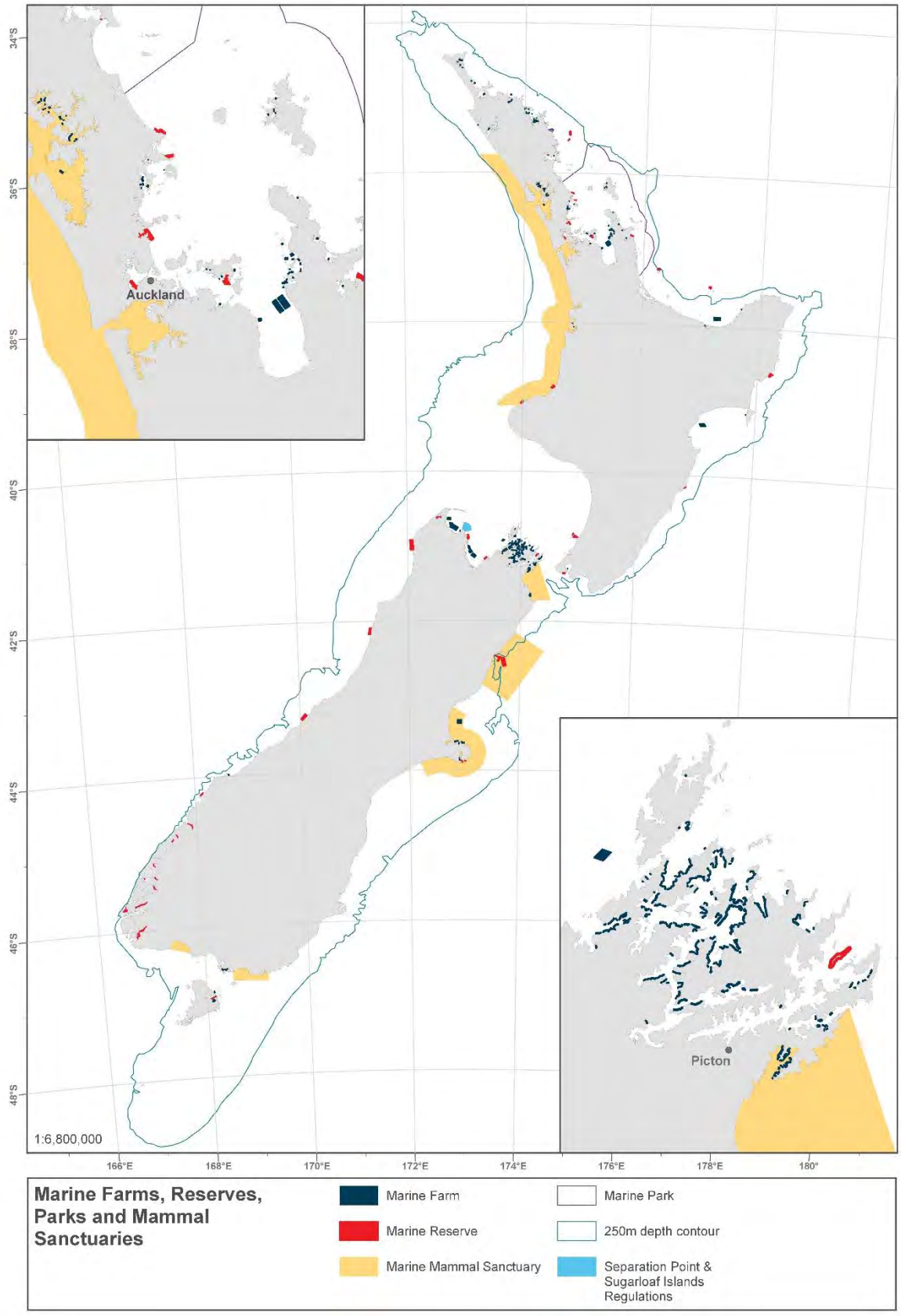


Figure 11.4: Points indicative of locations of marine reserves and marine farms, Separation Point and Sugar Loaf Islands closed areas, marine mammal sanctuaries, and marine parks, in waters shallower than 250 m depth.

Benthic habitats provide shelter and refuge for juvenile fish and the associated fauna can be the prey of demersal fish species. Towed fishing gears (particularly trawl doors), affect benthic habitats and organisms but the level of effect will depend on the type of trawl doors and ground gear used, and the physical and biological characteristics of the seabed habitats in the fishing grounds. The effects are difficult to assess because of the complexity of benthic communities and their temporal and spatial variability, and interpretation can also be complicated by environmental gradients or change. For reasons of accessibility, cost, and tractability, most research on seabed disturbance caused by human activities worldwide has been carried out in coastal systems, and our understanding of the effects of physical disturbance in the sparse but highly diverse communities of the deep sea has developed only recently. The reviews above broadly indicate that numerical abundance of many invertebrate declines (sometimes substantially) after mining, trawling, or other major disturbance. Trawling and dredging can re-suspend sediment and can, depending on sediment and local currents, alter sediment characteristics. Physical effects include furrows and berms from trawl doors, furrows from the bobbins and rock hoppers, and sediment resorting, but the magnitude of these effects depends on sediment type, currents, and wave action (if any). Bottom trawling can also alter natural sediment fluxes and reduce organic carbon turnover (Pusceddu et al. 2014), the depth of the oxic layer in sediments (Churchill 1989, Warnken et al. 2003, Bradshaw et al. 2012), and the shape of the upper continental slope (Puig et al. 2012), reducing morphological complexity and benthic habitat heterogeneity. The mixing of sediments and overlying water can alter the chemical makeup of the sediment and have considerable effects in deep, stable waters (Rumohr 1998). Chemical release from the sediment can also be changed, as shown for phosphate in the North Sea (ICES 1992, noting lower fluxes were observed after trawling events). Trawling can alter benthic communities, reduce total biomass of benthic species, and increase predation by scavengers. Sites subject to greater natural disturbance are generally thought to be less susceptible to change from bottom contact fishing (but see Schratzberger et al. 2009 who concluded that common anthropogenic disturbances differ fundamentally from natural disturbance). There has been less work on the effects of other methods of catching demersal fish or crustaceans that do not involve deliberately towing or dragging fishing gear across the seabed, but some of these methods can have non-negligible effects (e.g., Sharp et al. 2009, Williams et al. 2011).

Studies of recovery dynamics are rarer still, but a return to pre-disturbance levels after bottom-contact fishing can take up to several years, even in some sites subject to considerable natural disturbance (see Kaiser et al. 2006 for a summary). In shallow regions with mobile sediments, the effects are generally difficult to detect and recovery can be rapid (e.g., Jennings et al. 2005). Examining epifauna, Lambert et al. (2014) estimated recovery from scallop dredging to take from less than 1 year to over 10 years, depending on functional group, with faster recovery in areas with faster tidal currents, and large-bodied species recovering faster when conspecifics were abundant locally. Hard-bottom fauna is predicted to recover most slowly and Williams et al. (2010) concluded that hard-bottom fauna on Australasian seamounts did not show signs of recovery within 5–10 years. Recovery rate is typically correlated with the spatial extent of a disturbance event (e.g., Hall 1994, Kaiser et al. 2003; see also Figure 11.5) and the effects of some 'catastrophic' natural disturbance events, such as large-scale marine mudslides, can be detected for hundreds of years, even for taxa thought to be robust to physical disturbance such as nematodes (Hinz et al. 2008).

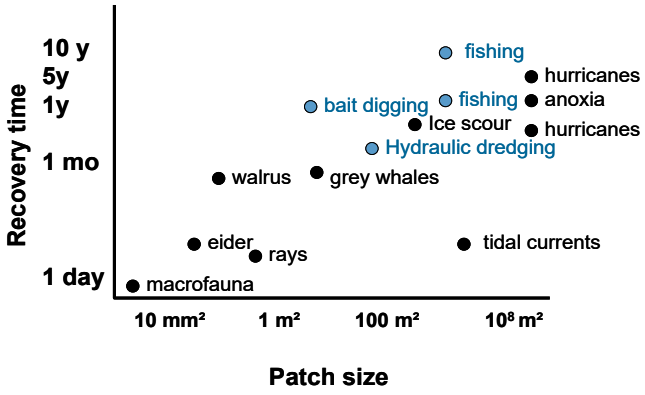


Figure 11.5: General relation between the spatial extent of disturbance events and the time taken to recover from such events in marine systems (after Kaiser et al. 2003). Blue dots signal human impacts, including fishing in habitats of different abilities to recover, and black dots signal natural disturbance.

Rice (2006) summarised the findings of five major reviews of the effects of mobile bottom-contacting fishing gears on benthic species, communities, and habitats. In this 'review of reviews' Rice (2006) summarised the findings of the multiple working groups that contributed to the reviews as follows:

Rice's (2006) conclusions about the effects on habitats of mobile bottom fishing gears were that they can:

- Damage or reduce structural biota (all reviews, strong evidence or support).
- Damage or reduce habitat complexity (all reviews, variable evidence or support).
- Reduce or remove major habitat features such as boulders (some reviews, strong evidence or support).
- Alter seafloor structure (some reviews, conflicting evidence for benefits or harm).

Other emergent conclusions on habitat effects included:

- There is a gradient of effects, with greatest effects on hard, complex bottoms and least effect on sandy bottoms (all reviews, strong support, with qualifications).
- There is a gradient of effects, with greatest effects on low energy environments and least (often negligible) effect on high-energy environments (all reviews, strong support).
- Trawls and mobile dredges are the most damaging of the gears considered (three of the reviews considered other gears; all drew this conclusion, often with qualifications).

Mobile bottom gears affect benthic species and communities in that they:

- Can change the relative abundance of species (all reviews, strong evidence or support).
- Can decrease the abundance of long-lived species with low turnover rates (all reviews, moderate to strong evidence or support).
- Can increase the abundance of short-lived species with high turnover rates (all reviews, moderate to occasionally strong evidence or support).
- Affect populations of surface-living species more often and to greater extents than populations of burrowing species (all reviews, weak to occasionally strong evidence or support).
- Have lesser effects in high-energy or frequent natural disturbance environments than in low energy environments where natural disturbances are uncommon (four reviews (the other did not address the factor), strong evidence or support).
- Affect populations of structurally fragile species more often and to greater extents than populations of 'robust' species (all reviews, variable evidence and support).
- Temporarily increase the abundance of scavengers in areas where bottom trawls have been used (three reviews, variable support or evidence, all argue for the effects being transient).
- Increase the rates of nutrient cycling or sedimentation in areas where bottom trawls have been used (two reviews, mixed views on magnitude of effects and conditions under which they occur).

Considerations in the application or adoption of mitigation measures:

- The effect of mobile fishing gears on benthic habitats and communities is not uniform. It depends on:
  - The features of the seafloor habitats, including the natural disturbance regime (all reviews, strong evidence or support);
  - The species present (all reviews, strong evidence or support, though not mentioned by NMFS panel);
  - The type of gear used and methods of deployment (all reviews, moderate to strong evidence or support);
  - The history of human activities, particularly past fishing, in the area of concern (all reviews, strong evidence or support).
- Recovery time from trawl-induced disturbance can take from days to centuries, and depends on the same factors as listed above (all reviews, strong evidence or support).
- Given the above considerations, the effect of mobile bottom gears has a monotonic relationship with fishing effort, and the greatest effects are caused by the first few fishing events (all reviews, moderate to strong evidence or support).
- Application of mitigation measures requires case specific analyses and planning; there are no universally appropriate fixes (three reviews, moderate to strong evidence or support. The issue of implementing mitigation was not addressed in the FAO review. It was also stressed in the US National Academy of Sciences review and discussed in the ICES review that extensive local data are not necessary for such case-specific planning. The effects of mobile bottom gears on seafloor habitats and communities are consistent enough with well-established ecological theory, and across studies,

that cautious extrapolation of information across sites is legitimate).

Rice (2006) concluded *'These overall conclusions on impacts and mitigation measures, and recommendations for management action form a coherent and consistent whole. They are relevant to the general circumstances likely to be encountered in temperate, sub-boreal, and boreal seas on coastal shelves and slopes, and probably areas … beyond the continental shelves. They allow use of all relevant information that can be made available on a case by case basis, but also guide approaches to management in areas where there is little site-specific information.'*

Since Rice's (2006) paper, Kaiser et al. (2006) published a meta-analysis of 101 separate manipulative experiments that confirms many of Rice's findings. Shellfish dredges have the greatest effect of the various mobile bottom fishing gears, biogenic habitats are the most sensitive to such disturbance (especially for attached fauna on hard substrates) and unconsolidated, coarse sediments (e.g., sands) are the least sensitive. Kaiser et al. (2006) concluded that recovery from disturbance events can take months to years, depending on the combination of fishing method and benthic habitat type. This meta-analysis of manipulative experiments was an important development, reinforcing the inferences drawn from multiple mensurative observations at much larger scale ('fisheries scale') in New Zealand (e.g., Thrush et al. 1998, Cryer et al. 2002) and overseas (e.g., Craeymeersch et al. 2000, McConnaughey et al. 2000, Bradshaw et al. 2002, Blyth et al. 2004, Tillin et al. 2006, Hiddink et al. 2006). This is a powerful combination that implies substantial generality of the findings.

The international literature is, therefore, clear that bottom (demersal) trawling and shellfish dredging are likely to have largely predictable and sometimes substantial effects on benthic community structure and function. However, the positive or negative consequences for ecosystem processes such as production had not been addressed until more recently (e.g., Jennings et al. 2001a, Reiss et al. 2009, Hiddink et al. 2011). It has been mooted that frequent disturbance should lead to the dominance of smaller species with faster life histories and that, because smaller species are more productive than larger ones, system productivity and production should increase under trawling disturbance. However, when this proposition has been tested, it has not been supported by data in real fishing situations (e.g., Hermsen et al. 2003, Reiss et al. 2009) and where overall productivity has been assessed, it decreases with increasing trawling disturbance.

For example, Veale et al. (2000) examined spatial patterns in the scallop fishing grounds in the Irish Sea and found that total abundance, biomass, and secondary production (including that of most individual taxa examined) decreased significantly with increasing fishing effort. Echinoids, cnidarians, prosobranch molluscs, and crustaceans contributed most to the differences. Jennings et al. (2001a) showed that, in the North Sea, trawling led to significant decreases in infaunal biomass and production in some areas even though production per unit biomass rose with increased trawling disturbance. The expected increase in relative production did not compensate for the loss of total production that resulted from the depletion of large-bodied species and individuals. Hermsen et al. (2003) found that mobile fishing gear disturbance had a conspicuous effect on benthic megafaunal production on Georges Bank, and cessation of such fishing led to a marked increase in benthic megafaunal production, dominated by scallops and urchins. Hiddink et al. (2006) estimated that more than half of the southern North Sea was trawled sufficiently frequently to depress benthic biomass by 10% or more, and that 27% was in a state where benthic production was depressed by 10% or more. They estimated that recovery from this situation would take 2.5–6 years or more once fishing effort had been eliminated. They further estimated that fishing reduced benthic biomass and production by 56% and 21%, respectively, compared with an unfished situation. Reiss et al. (2009) found that, although sediment composition was the most important driver of benthic community structure in their North Sea study area, the intensity of fishing effort was also important and reductions in the secondary production of the infaunal community could be detected even within this heavily fished region.

The types of models developed by Hiddink et al. (2006, 2011; but see also Ellis & Pantus 2001 and Dichmont et al. 2008) can be used to assess the likely performance of different management approaches or levels of fishing intensity. Such management-strategy-evaluation (MSE) methods involve specifying management objectives, performance measures, a suite of alternative management strategies, and evaluating these alternatives using simulation (Sainsbury et al. 2000). For instance, the early study by Ellis & Pantus (2001) assessed the effect of trawling on marine benthic communities by combining an implementation of the spatial and temporal behaviour of

the local fishing fleet with realistic ranges for the removal and recovery of benthic organisms. The model was used to compare the outcomes of two radically different management approaches, spatial closures and reductions in fishing effort. From a New Zealand perspective, Mormede & Dunn (2013) developed a simple spatially explicit population model as a tool to assist Ecological Risk Assessments, and Lundquist et al. (2010, 2013) used a more sophisticated spatially explicit landscape mosaic model with variable connectivity between patches to assess the implications of different spatial and temporal patterns of disturbance in the model landscape. They found that the scale of the disturbance regime (which could be trawling or any other physical disturbance) and the dispersal processes interact, and that the scales of these processes greatly influenced changes in the structure and diversity of the model community, and that recovery across the mosaic depended strongly on dispersal. System stability also decreased as dispersal distance decreased. Patterns of abundance of different species groups observed across gradients of fishing pressure were in general agreement with model predictions.

11.3 STATE OF KNOWLEDGE IN NEW ZEALAND

To understand the effects of mobile bottom fishing methods on benthic habitats, it is necessary to have knowledge of:

- the distribution of such habitats,
- the extent to which mobile bottom fishing methods are used in each habitat (the overlap),
- the consequences of any such disturbance (potentially in conjunction with other disturbances or stressors), and
- the nature and speed of recovery from the disturbance.

These components will be dealt with in turn.

11.3.1 DISTRIBUTION OF HABITATS

Mapping of benthic habitats at the large scales inherent in fisheries management is expensive and time-consuming so the New Zealand government commissioned an environmental classification to provide a spatial framework that subdivided the TS and EEZ into areas having similar environmental and biological character. This Marine Environment Classification (MEC) was launched in 2005 (Snelder et al. 2004, 2005, 2006) using available physical and chemical predictors, because environmental pattern was thought to be a reasonable surrogate for biological pattern. The authors suggested that the MEC provided managers with a useful spatial framework for broad-scale management, but cautioned that the full utility and limitations would become clear only as the MEC was applied to real issues. They described the MEC as a tool to organise data, analyses and ideas, and as only one component of the information that would be employed in any analysis. The 20-class version (Figure 11.6, Table 11.1) has been the most widely cited, although additional classification levels provide more detail that is significantly correlated with biological layers. The 2005 MEC was not optimised for any specific ecosystem component but was 'tuned' against data for demersal fish, phytoplankton, and benthic invertebrates. It performed least well as a classification of benthic invertebrates and, at the 20-class level, grouped most of the Chatham Rise and Challenger Plateau into a single class. Although separation of these two areas was evident as the MEC was driven to larger numbers of classes, their inclusion within a single class in the 20-class classification was considered counter-intuitive because their productivity and fisheries are known to be very different.

This disquiet with the predictions of the original MEC for benthic habitat classes led to the development of alternatives that might perform better for benthic systems. First of these was a classification optimised for demersal fish (Leathwick et al. 2006). Several variants of this classification outperformed the original MEC for demersal fish, particularly at lower levels of classification detail and it was adopted by the Ministry for the Environment for their indicators related to bottom trawling and their 2010 Environmental Snapshot where the trawl footprint is compared with putative habitats (Ministry for the Environment 2010, see also: https://www.mfe.govt.nz/environmental-reporting/marine/fishing-activity-indicator/fishing-activity-seabed-trawling.html).

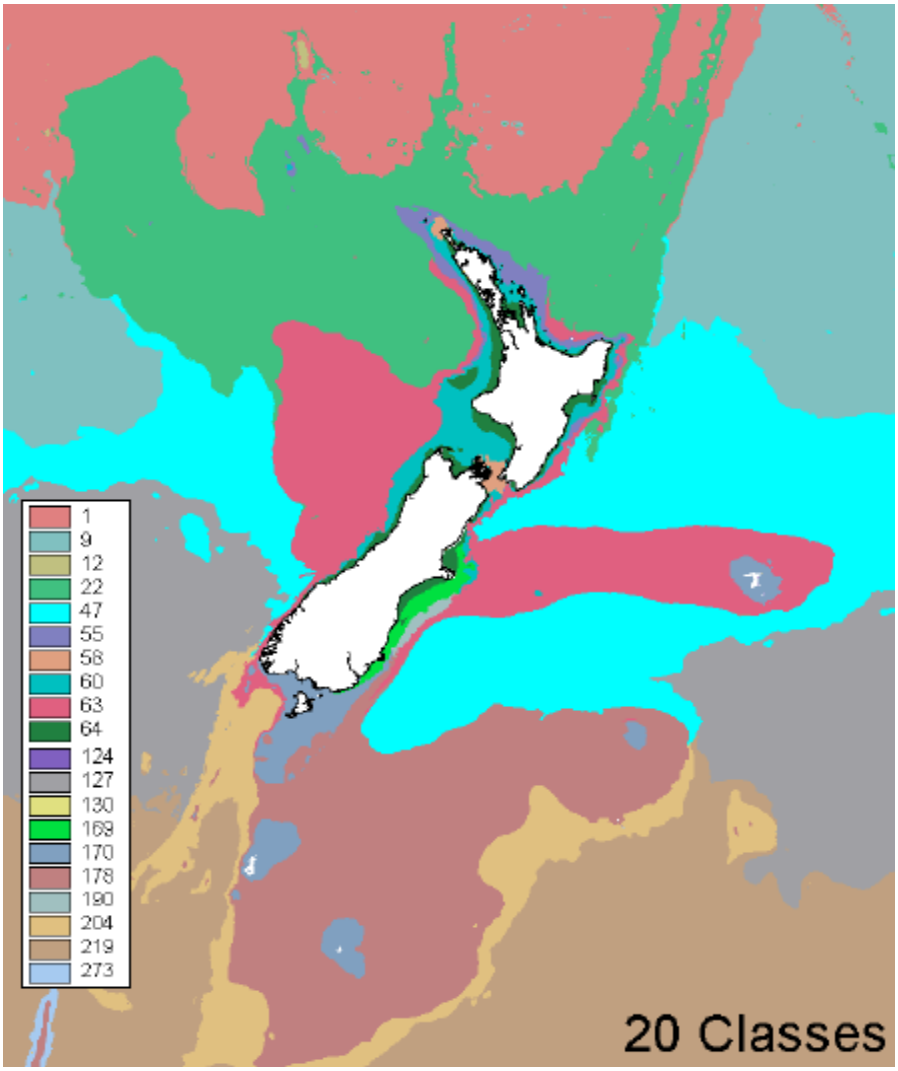


Figure 11.6: The 20-class version of the 2005 general purpose Marine Environment Classification (MEC, from Snelder et al. 2005). The class numbers are nominal; for attributes of each class at this level, see Table 11.1.

Based partly on this experience, the Ministry of Fisheries commissioned a Benthic-Optimised Marine Environment Classification, BOMEC (Leathwick et al 2012). Many more physical, chemical, and biological data layers were available for the development and tuning of this classification than for the 2005 MEC. Especially relevant for benthic invertebrates was the inclusion of a layer for sediment grain size (notably absent from the MEC). Generalised Dissimilarity Modelling (GDM; Ferrier et al. 2002, 2007, Leathwick et al. 2011) was used to define the classification because this approach is well suited to the sparse and unevenly distributed biological data available. The BOMEC classes (15-class level version shown in Figure 11.7) were strongly driven by depth, temperature, and salinity into five major groups: inshore and shelf; upper slope; northern mid-depths; southern mid-depths; and deeper waters (generally beyond the fishing footprint, down to 3000 m, the limit of the analysis). Waters deeper than 3000 m could be considered an additional class. The 15-class BOMEC levels were used in conjunction with a broad sediment type classification and broad depth bands to identify 112 benthic habitats shallower than 250 m (Figure 11.8) (Baird et al. 2015).

Recent testing (Bowden et al. 2011) has indicated that the BOMEC out-performs the original MEC at predicting benthic habitat classes on and around the Chatham Rise, but that none of the available classifications is very good at predicting the abundance and composition of benthic invertebrates at the fine scale of the sampling undertaken (tens of metres to kilometres). This, in conjunction with the findings of Leathwick et al. (2006), reinforces the role of environmental classifications as broad-scale predictors of general patterns at broad scale (tens to hundreds of

kilometres) when more specific biological information is not available.

Where broad-scale classification methods are not applicable, other approaches have been taken. The trawl fisheries for orange roughy, oreos, and cardinalfish take place to a large extent on seamounts or other features (Clark & O'Driscoll 2003, O'Driscoll & Clark 2005). These features are often geographically small and, in common with other, localised habitats such as vents, seeps, and sponge beds, do not appear on broad-scale habitat maps (e.g., at EEZ scale) and cannot realistically be predicted by broad-scale environmental classifications. Many features have been extensively mapped in recent years (e.g., Rowden et al. 2008), and seamount classifications based on biologically-referenced physical and environmental 'proxies' have also been developed, in New Zealand waters by Rowden et al. (2005), and globally by Clark et al. (2010a, 2010b). Davies & Guinotte (2011) developed a method of predicting the framework-forming (i.e., physically structuring) coldwater corals that are a focus for benthic biodiversity in deepwater systems. MPI and MBIE funded a range of projects to inform the spatial management of the South Pacific Region and support the role of New Zealand in SPRFMO. The project "Predicting the occurrence of vulnerable marine ecosystems for planning spatial management in the South Pacific Region" is led by NIWA in collaboration with Victoria University of Wellington and the Marine Conservation Institute (USA). This has involved a major compilation of data on VME species distribution (including corals) throughout the SPRFMO area, as well as the EEZ. Modelling has been completed for the EEZ and general New Zealand region (Anderson et al. 2016a), as well as validation analyses of a wider regional model based on a survey of the Louisville Seamount Chain in 2014 (Anderson et al. 2016b). Work continues worldwide, including in New Zealand, on the development of sampling, analytical, and modelling techniques to provide cost-effective assessments of the distribution of marine habitats at a range of scales. Bowden et al. (2015) provide a desk top assessment of future options for monitoring deepwater benthic communities, and conclude that photographic approaches sampling mega-epifauna are likely to be the most cost effective and relevant for detecting ecological effects at the scale of deep sea fisheries. Such sampling could be added to existing surveys, but would require dedicated time. Opportunistic sampling from trawl surveys or observer data cannot be relied upon to provide representative samples of the benthic community.

Table 11.1: Average values for each of the eight defining environmental variables in each class of the 20-class level of the MEC classification. After Snelder et al. (2005).

Class	Area (km^2)	Depth	Slope	Orbital velocity	Radiation mean	SST amplitude	SST gradient	SST winter	Tidal current	2-class level	4-class level	9-class level
1	88,503	-3001	1.4	0	17.5	2.3	0.01	19.5	0.06	Oceanic	Subtropical	Deep
22	53,368	-1879	1.5	0	15.4	2.4	0.01	16.3	0.11			
9	64,306	-5345	1.4	0	14.8	2.6	0.01	16.1	0.03			Abyssal
47	60,053	-2998	1.0	0	12.1	2.4	0.01	11.6	0.07		Shelf and subtropical front	Central
55	2,213	-334	1.6	0	15.5	2.4	0.02	15.1	0.20			
63	26,626	-754	0.9	0	12.8	2.4	0.02	12.1	0.18			
178	39,360	-750	0.4	0	9.5	1.3	0.01	7.6	0.15			Southern
127	60,884	-4830	0.5	0	10.7	1.7	0.01	10.0	0.05		Sub-Antarctic	
204	18,277	-2044	3.0	0	9.2	0.9	0.01	8.0	0.08			
273	805	-2550	9.1	0	8.4	1.4	0.03	4.4	0.05			
219	93,982	-4779	0.6	0	8.9	1.0	0.01	6.7	0.04			
12	149	-94	0.9	113	17.8	2.3	0.01	19.3	0.30	Coastal		Northern
58	394	-117	0.7	57	14.7	2.2	0.03	13.0	1.09			Central
60	4,084	-112	0.3	21	14.4	2.5	0.02	13.2	0.26			
64	2,689	-38	0.3	272	14.2	2.9	0.02	12.6	0.19			
124	68	-8	0.4	836	13.4	2.3	0.02	12.7	0.00			
130	14	-10	0.4	353	14.1	2.4	0.09	11.9	0.21			
169	932	-66	0.2	113	12.4	2.7	0.04	9.9	0.21			
190	339	-321	1.9	3	12.3	2.3	0.06	9.4	0.10			
170	5,208	-129	0.3	99	10.2	1.3	0.02	9.3	0.55			Southern

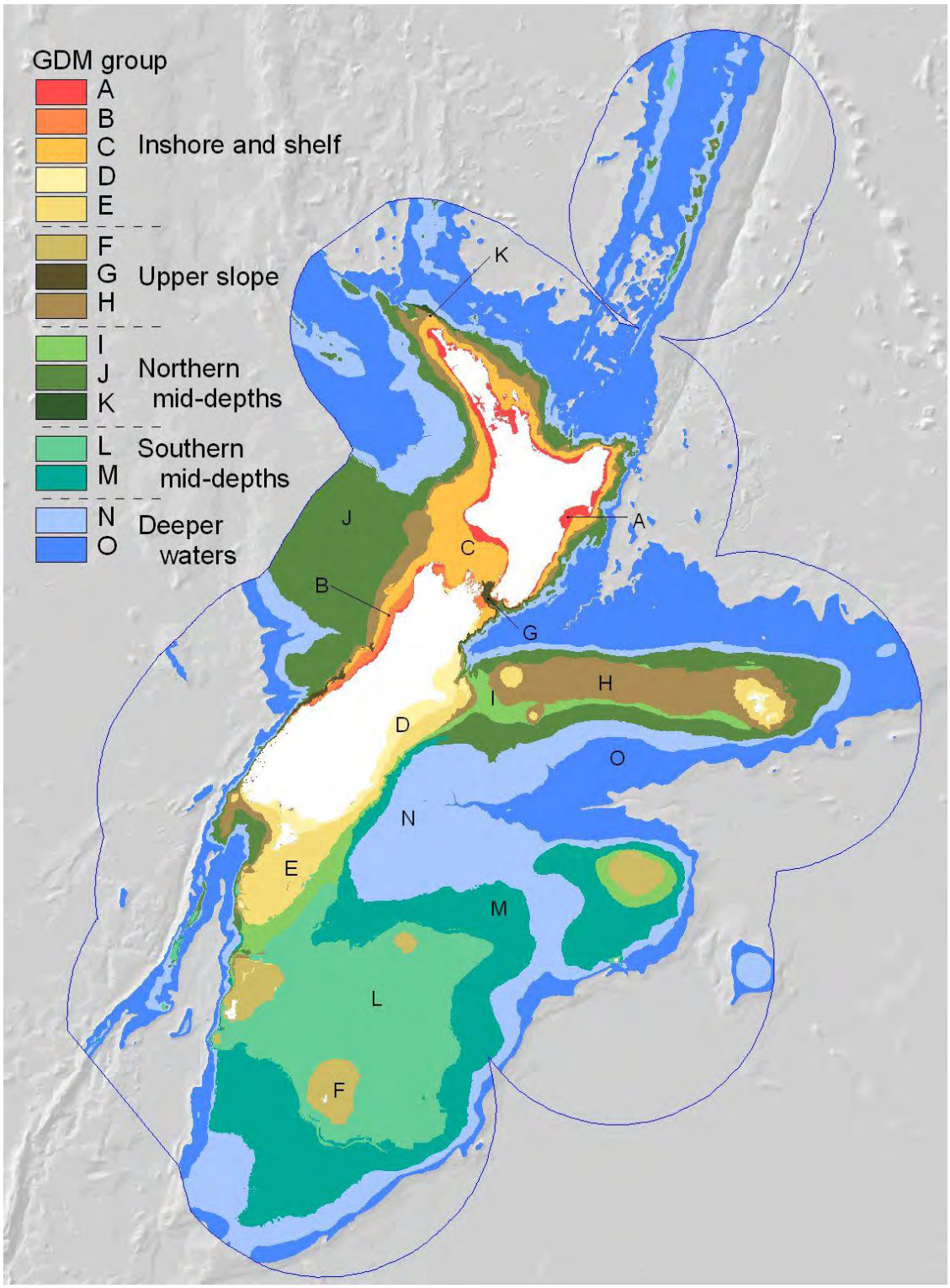


Figure 11.7: Map of the distribution of Benthic-Optimised Marine Environment Classification (BOMEC) classes defined by multivariate classification of environmental data transformed using results from GDM analyses of relationships between environment and species turnover averaged across eight taxonomic groups of benthic species. From Leathwick et al. (2012).

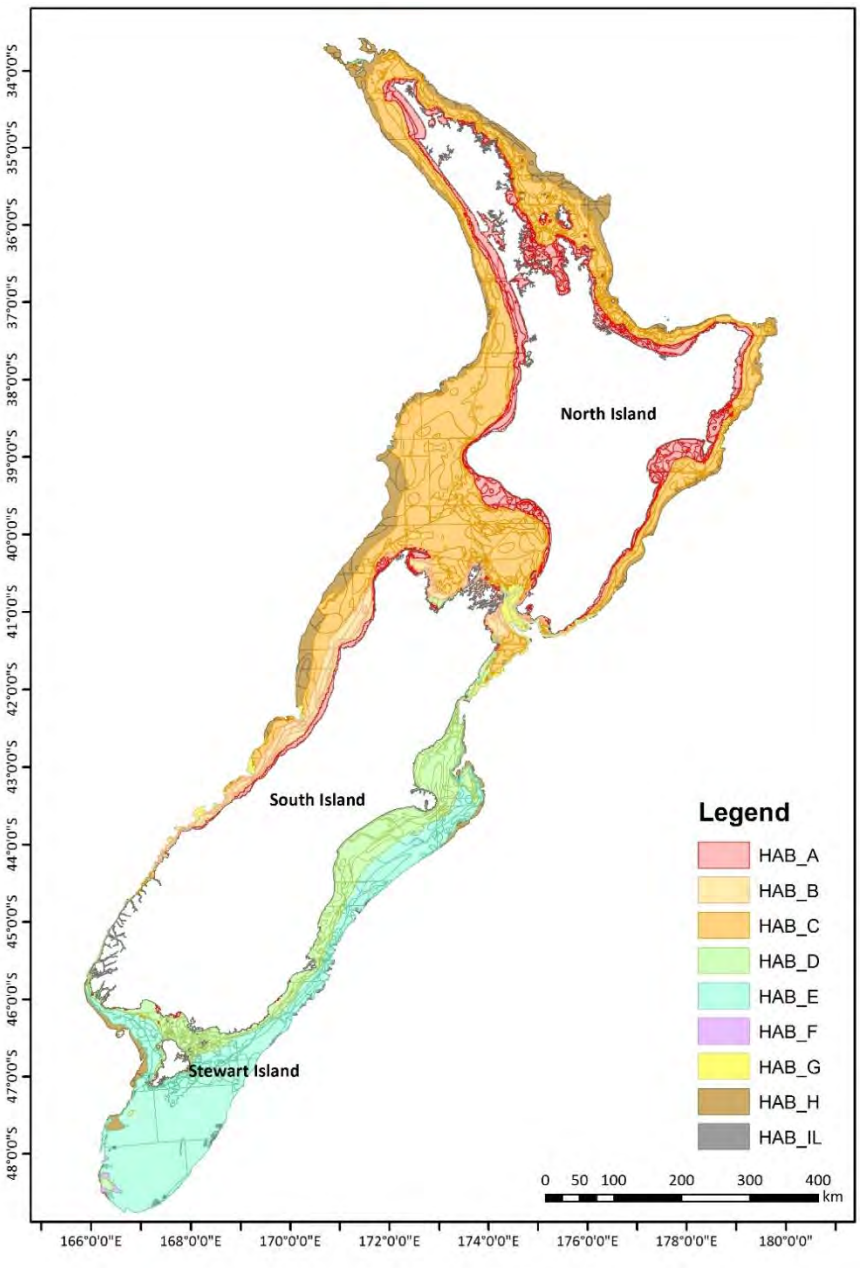


Figure 11.8: The broad habitat definitions based on the BOMEC classes, with divisions indicating areas of different sediment, depth zone, and statistical area in waters shallower than 250 m depth. From Baird et al. (2015).

11.3.2 DISTRIBUTION OF FISHING

Since 1989–90, mobile bottom fishing has been reported on one of three standardised reporting forms (Table 11.2). Trawl Catch Effort and Processing Returns (TCEPRs) contain detailed spatial and other information for each trawl tow, whereas Catch Effort and Landing Returns (CELRs) include only summarised information for each day's fishing, with very limited spatial resolution. Since 2007–08, Trawl Catch and Effort Returns (TCERs) have been available for smaller, predominantly inshore trawlers. These include spatial and other information for each trawl tow but in less detail than on TCEPRs. Between 1989–90 and 2004–05, only about 25% of all mobile bottom fishing events were reported on TCEPRs. Another 25% were bottom trawls reported on CELRs, and the remaining 50% were dredge tows for shellfish reported on CELRs. The distribution of trawling reported on CELRs is not the same as that reported on TCEPRs; the smaller trawlers using CELRs were much more likely than the larger boats to fish close to the coast and target inshore species such as flatfish, red cod, tarakihi, and red gurnard (collectively 73% of all trawl tows reported on CELRs).

Table 11.2: Attributes, usage, and resolution of spatial reporting required on Trawl Catch Effort and Processing Returns (TCEPRs), Trawl Catch and Effort Returns (TCERs), Catch Effort and Landing Returns (CELRs) and Electronic Reporting (ER).

	Trawl catch and effort reporting forms			
	CELR	TCEPR	TCER	ER
Year of introduction	1988-89	1989-90	2007–08	2017-18
Vessels using	Trawlers not using TCER or TCEPR Shellfish dredgers	All trawlers >28 m Other vessels as directed Other vessels optional	All trawlers 6–28 m unless exempted	All trawlers >28 m To be phased in for all other vessels from January 2019
Trawl tow reporting	Daily summary, number of tows, gear, fishery area	Tow by tow, start and finish locations, speed, depth, gear, duration	Tow by tow, start location, speed, depth, gear, duration	Tow by tow, start and finish locations, speed, depth, gear, duration, lost gear, mitigation
Spatial resolution	Statistical reporting area (optionally lat/long)	1 minute (lat/long)	1 minute (lat/long)	4 decimal places of a degree (lat/long)

Baird et al. (2002) and Baird et al. (2011) described the distribution and frequency of reported fishing by mobile bottom fishing gear (dredge, Danish seine, bottom trawl, bottom pair trawl, and mid-water trawl in contact with the bottom) in New Zealand's TS and EEZ during the 1990s and up to 2004–05, respectively, for all commercial targets. This work was updated to 2011–12 by Black & Tilney (2017) for deepwater target species effort reported on TCEPR. These reports showed that fishing was highly heterogeneous (spatially), but had considerable consistency among years; sites that were fished heavily in one year were likely to be fished heavily in other years and vice versa. A similar but more detailed analysis was conducted for the Chatham Rise and subantarctic areas by Baird et al. (2006). Tows reported on TCEPRs were included in the main spatial analysis, but some additional analysis was possible using tows reported on CELRs. Until 2006–07, many inshore vessels used CELRs and these comprised a substantial proportion of reported trawling, even for some 'deepwater' species. For instance, Cryer & Hartill (2002) estimated that, in the Bay of Plenty in the 1990s, 78%, 75% and 39% of trawl tows targeting tarakihi, gemfish and hoki, respectively, were reported on CELR forms. Since 2007–08, almost all trawling effort has been reported on TCEPR or TCER forms.

Baird & Wood (2018) updated the three annual measures of fishing effort for deepwater fishstocks: the number of tows, the aggregate swept area (using assumed door spreads), and the coverage ('footprint') of the total trawl contact (Figure 11.9). Trawls were represented spatially as tracklines between the reported start and finish positions buffered by the assumed door spread to generate trawl polygons. The aggregate swept area for a year is the sum of the areas of the polygons and the 'footprint' is the estimated area of the seabed that is covered by the polygons overlaid. The estimated swept areas and footprint do not account for any modification that might occur alongside the trawl path as represented by the swept area polygon (e.g., by suspended sediments transported by currents away from the trawl track). Baird & Wood (2018) produced maps of the footprint, for each of the deepwater Tier 1 target species or species groups and for the Tier 1 and Tier 2 target species combined, summary maps of the aggregated swept area, and various tables and figures describing trends. The annual number of trawls peaked in 1997–98 at 57 195 tows (aggregate swept area about 166 092 km^2). In 2015–16, 25 572 tows were reported on TCEPRs (about 78 392 km^2). The intensity of bottom-contacting trawling in the main fishery areas is shown in Figure 11.9 for the combined data for the 1989–90 to 2015–16 fishing years.

Baird & Wood (2018) used reported tows on small topographic features that are a focus for orange roughy and cardinalfish fisheries by defining polygons for these tows as radii around the reported start position with the area swept estimated from the reported duration and speed of the

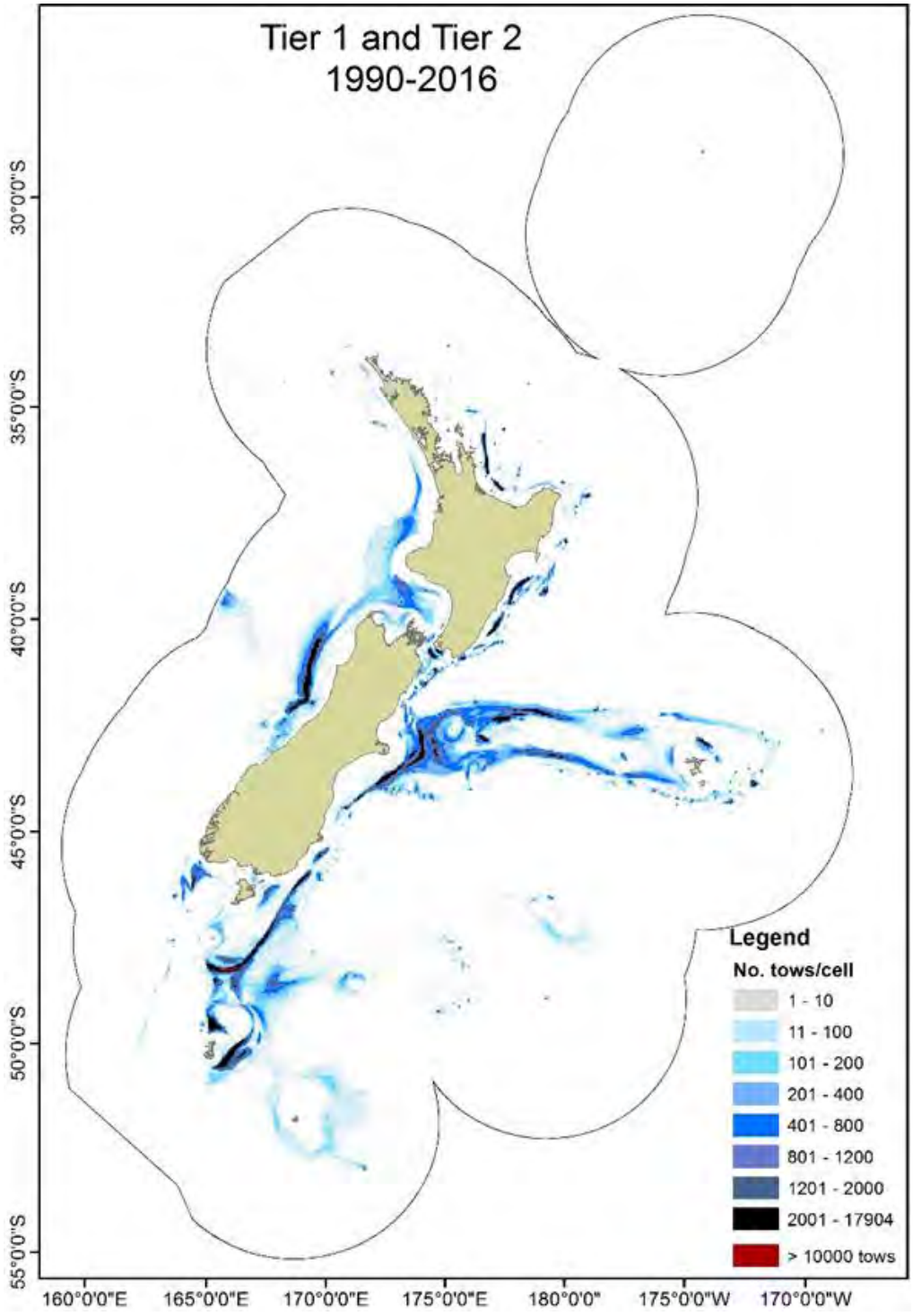


Figure 11.9: Map from Baird & Wood (2018) showing the frequency of bottom-contacting trawling effort reported on TCER and TCEPR forms 1989–90 to 2015–16. The colour scale indicates the frequency of bottom-contact trawling estimated by Baird & Wood for each 5 × 5 km cell, all deepwater Tier 1 and Tier 2 target fishstocks combined (i.e., the most frequently fished 25 km² cells had over 17 000 tows recorded over 27 years).

tow. These short tows do not appear to contribute substantially to broad-scale plots like Figure 11.9, yet can represent intense fishing effort on particular, small seamount features (e.g., Rowden et al. 2005, O'Driscoll & Clark 2005).

Previous trawl footprint analyses (Baird et al. 2011, Black et al. 2013) have recognised that they underestimated trawl effort in inshore areas through exclusion of data recorded on CELR forms. Baird et al. (2015) analysed a combined data set of TCEPR and TCER data for the area shallower than 250 m for 2007–08 to 2011–12 (Figure 11.10).

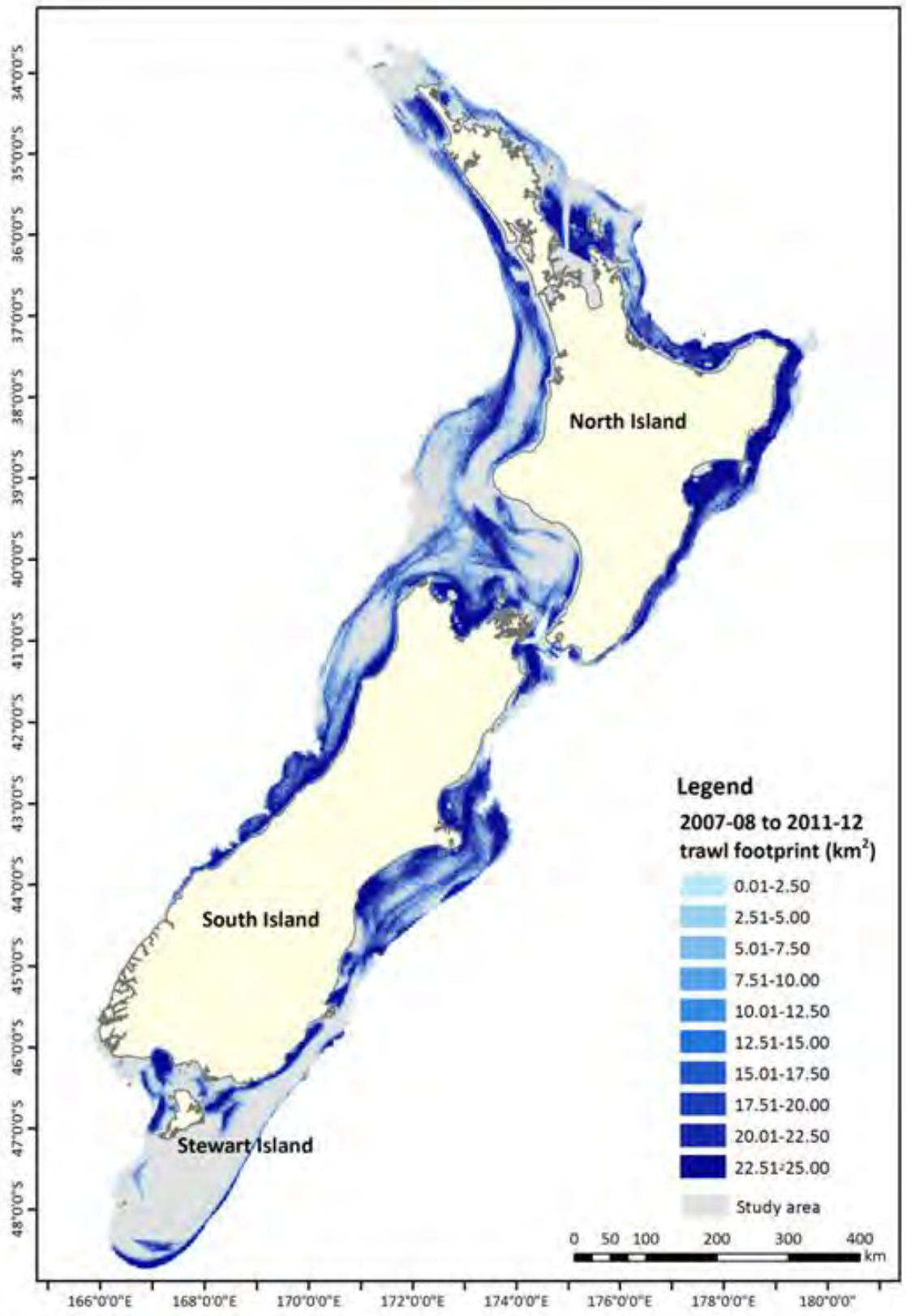


Figure 11.10: Total trawl cell based for footprint for the area shallower than 250 m for 2007–08 to 2011–12 combined (Baird et al. 2015).

Trawl effort for all target species are included in the data shown in Figure 11.11. After the peak of over 140 000 reported trawl tows in 1996–97 and 1997–98 (Figure 11.11) when slightly over half of all tows were reported on TCEPRs, overall trawling effort declined to less than 80 000 tows per year by 2013–14, only about 40% of which is reported on TCEPRs (virtually all other tows are reported on TCERs).

The most recent footprint was generated from the bottom-contacting trawl effort for deepwater fishstocks for 2007–08 to 2016–17, as reported on TCERs and TCEPRs (Baird & Mules 2018; Figure 11.12). This 10 year footprint contacted about 11% of the area shallower than 1600 m and 13% of the area that is open to bottom trawling and shallower than 1600 m. The 1989–90 to 2015–2016 footprint contacted about 20% of the area shallower than 1600 m and 23% of the area that is open to bottom trawling and shallower than 1600 m. Figure 11.13 shows the most recently determined footprint extent for deepwater stocks for the period 2007–08 to 2016–17 compared to the footprint extent determined for the period 1989–90 to 2015–16. Baird & Mules (2018) found that the decreasing trend in the annual footprint extent, after a peak period from 1998–2003 (Baird & Wood 2018), has continued (Figure 11.12).

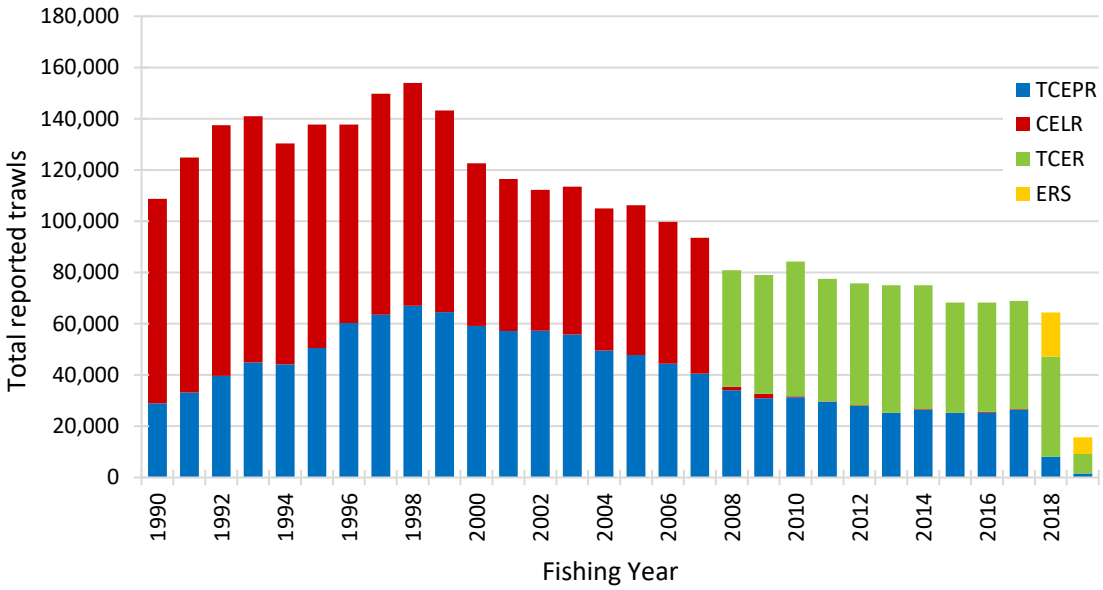


Figure 11.11: The number of bottom trawl tows reported on Trawl Catch Effort and Processing Returns (TCEPR), Catch Effort and Landing Returns (CELR), Trawl Catch and Effort Returns (TCER) and Electronic Reporting (ERS) between the 1989–90 (1990) and 2018–19 (2019) fishing years. Data for the 2018–19 year may be incomplete. [Updated for AEBAR 2018].

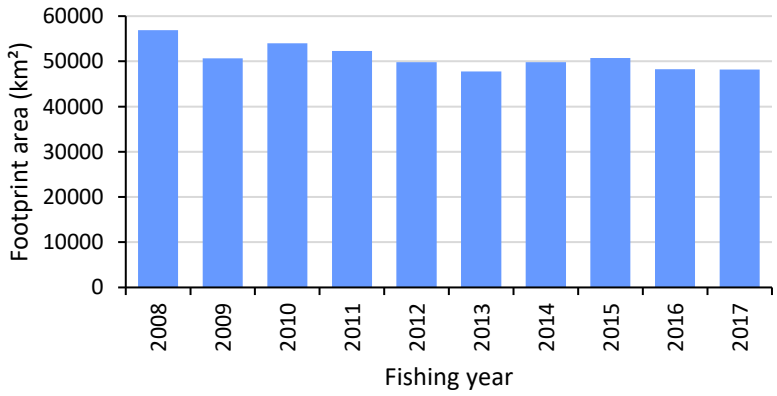


Figure 11.12: Annual footprint (km²) for bottom-contacting trawling for deepwater fishstocks, from TCERs and TCEPRs, between the 2007–08 (2008) and 2016–17 (2017) fishing years. [Updated for AEBAR 2018].

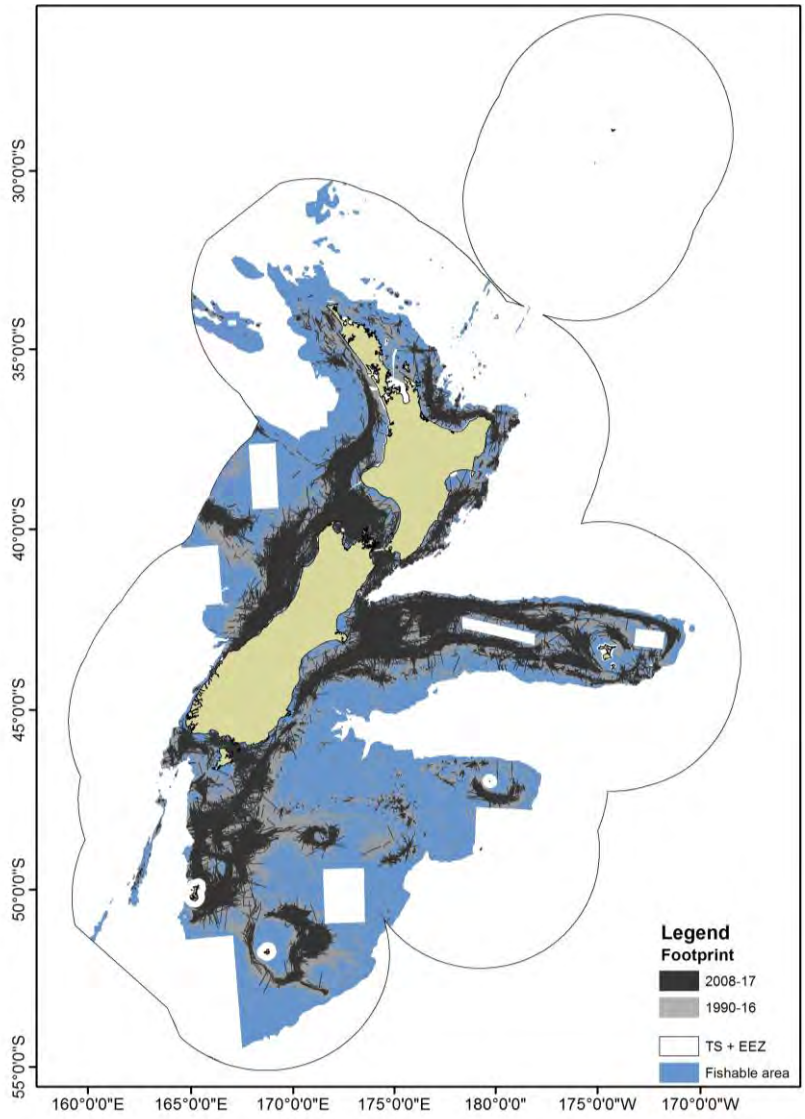


Figure 11.13: Comparison of the extents of the 2007–08 to 2016–17 and the 1989–90 to 2015–16 trawl footprints for Tier 1 and Tier 2 targets combined, relative to the 'fishable' area (depths to 1600 m in waters open to bottom trawling). [Updated for AEBAR 2018].

Dredging for shellfish (oysters and scallops) is conducted in a number of specific areas that have separate, smaller statistical reporting areas (Figure 11.14). Over the 30 year dataset, there were approximately 2 million scallop dredge tows in the four main scallop fisheries and over 1 million oyster dredge tows in the two dredge oyster fisheries. These data are collected on CELRs, usually at the spatial scale of a scallop or oyster fishery area and the data have been summarised as the number of dredge tows. No estimates of the area swept by these dredges have been made, but the number of reported tows has declined markedly since the early 1990s (Figure 11.15).

Our knowledge of the distribution of mobile bottom fishing effort within our TS and EEZ is, by international standards, very good; since 2007–08 we have had tow-by-tow reporting of almost all trawling with a spatial precision of about 1 n. mile. The distribution of dredge tows for shellfish is not reported with such high precision, but records kept by fishers in industry logbooks are often much more detailed than the Fisheries New Zealand statutory returns, and have sometimes been used to support spatial analyses that would not have been possible using the statutory returns (e.g., Tuck et al. 2006 for project ZBD2005/15 on the Coromandel scallop fishery and Michael et al. 2006 for project ZBD2005/04 on the Foveaux Strait oyster fishery). These studies indicate the value of records with higher spatial precision.

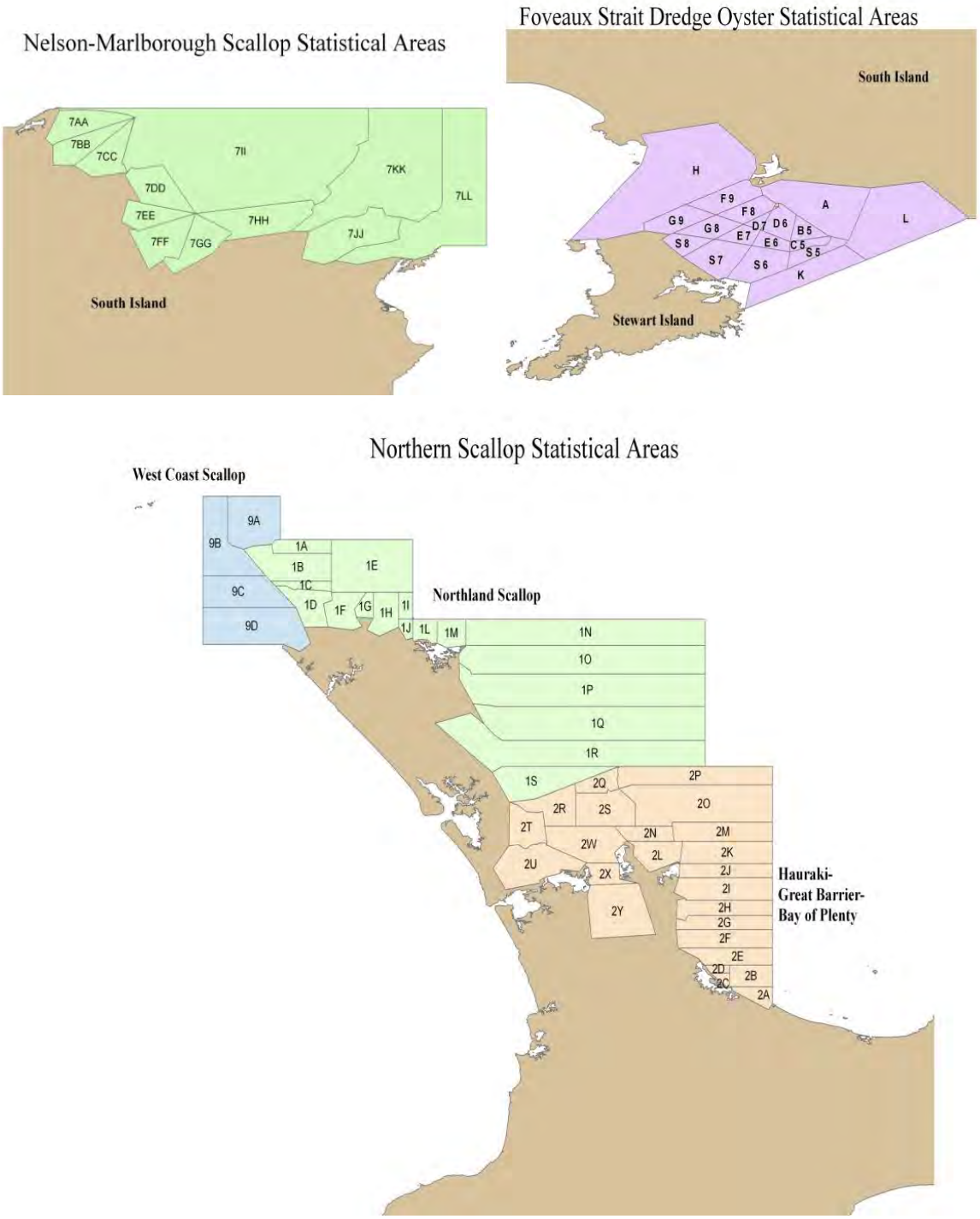


Figure 11.14: Maps taken from Baird et al. (2011) of statistical reporting areas for the main oyster and scallop dredge fisheries (scales differ). Note that these reporting areas are generally much smaller than the General Statistical Areas used for finfish reporting.

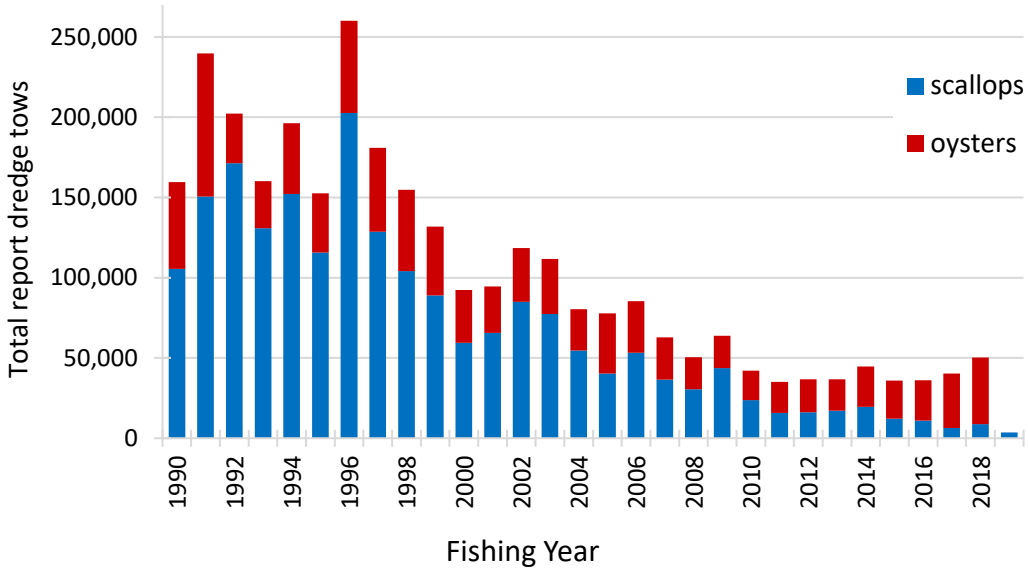


Figure 11.15: The number of dredge tows for scallop or oysters reported on Catch Effort and Landing Returns (CELR) between the 1989–90 (1990) and 2018–19 (2019) fishing years. Data for the 2018-2019 year will be incomplete. [Updated for AEBAR 2018].

11.3.3 OVERLAP OF FISHING AND PREDICTED HABITAT CLASSES

Tuck et al. (2014) reviewed a wide range of ecosystem indicators for deepwater fisheries, and concluded that in relation to benthic impact of fishing, indices of fishing footprint and fishing intensity by habitat and gear or fishery were likely to be the most useful. Baird & Mules (2018) overlaid the 2007–08 to 2016–17 deepwater Tier 1 and Tier 2 fishstock footprint on the 15-class BOMEC to estimate the proportion of each class that had been trawled (and reported on TCERs and TCEPRs) in the 10 year period (Figure 11.16). They found that the size of the footprint and the proportion of each class trawled varied substantially between habitat classes (Table 11.3, Figure 11.17). Class O is the largest BOMEC class but has almost no reported fishing effort; this class is mainly beyond trawlable depths. Conversely, class I is one of the smaller classes but has a larger trawl footprint that overlays 74% of the total class area. Two contrasting classes, together with their trawl footprints, are shown in Figure 11.18, based on analysis up to 2015–16 (Baird & Wood 2018). The 1989–90 to 2015–16 trawl footprint overlaps about 13% of the 2.6 million km^2 of seafloor covered by the BOMEC, about 8% of the 4.1 million km^2 of seafloor within the New Zealand EEZ boundary (i.e., including the Territorial Sea), or about 23% of the area open to trawling in depths down to 1600 m. However, these overlays and that for some individual BOMEC classes (particularly coastal classes A–E) do not represent the total bottom-contacting trawl footprint because the analysis was restricted to trawl effort for deepwater Tier 1 and Tier 2 target fishstocks, and CELR data between 1989–90 and 2006–07 were omitted. For example, this trawl footprint included only the 'deepwater' fishstocks for species such as barracouta and ling, and inshore species such as snapper and tarakihi were not included.

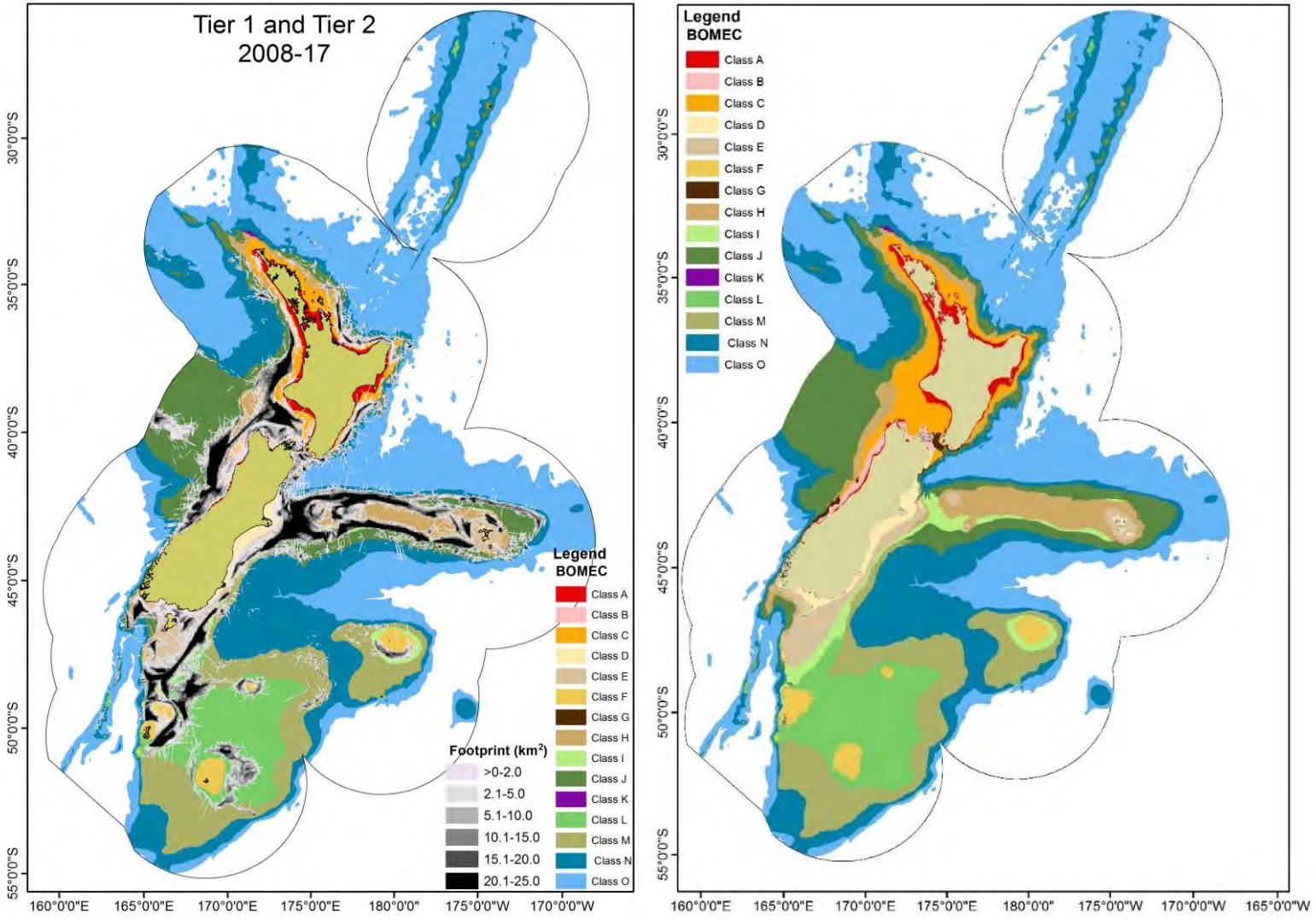


Figure 11.16: The distribution of the 2008–17 deepwater Tier 1 and Tier 2 fishstocks trawl footprint (left) (Baird & Mules (2018)) relative to the 15 BOMEC classes (to depths of 3000 m) (right), (after Leathwick et al. (2012).)
Note: previous versions have taken a longer prior time window (from 1989–90 until the latest fishing year).

Table 11.3: Estimated area of each BOMEC class (within the outer boundary of the EEZ) and fishing footprint from TCER and TCEPR deepwater Tier 1 and Tier 2 target fishstocks over the fishing years 2007–08 to 2016–17 (Baird & Mules 2018). Note: previous versions have taken a longer prior time window (from 1989–90).

BOMEC class	Total area (km²)	Area open to bottom-fishing (km²)	Footprint area (km²)	Footprint area (% of total)	Footprint area (% area open to bottom-fishing)
A*	27 557	19 764	364.1	1.3	1.8
B*	12 420	11 984	2 663.2	21.4	22.2
C*	89 710	87 914	22 638.7	25.2	25.8
D*	27 268	25 786	1 754.1	6.4	6.8
E*	60 990	60 211	12 764.0	20.9	21.2
F	38 608	30 931	3 787.0	9.8	12.2
G	6 342	6 033	2 117.4	33.4	35.1
H	138 550	129 323	39 171.8	28.3	30.3
I	52 224	51 910	26 802.3	51.3	51.6
J	311 361	277 138	35 637.7	11.4	12.9
K	1 290	1 290	0.0	0.0	0.0
L	198 577	175 509	23 720.5	11.9	13.5
M	233 825	183 402	4 574.1	2	2.5
N	493 034	388 647	3 398.7	0.7	0.9
O	935 315	598 607	566.7	0.1	0.1
Total	2 627 073	2 048 448	179 960.5	6.9	8.8

* The trawl footprint and proportion overlapped in coastal classes A–E will be grossly underestimated because the data included here are for deepwater fishstocks and therefore do not include the inshore fishstocks.

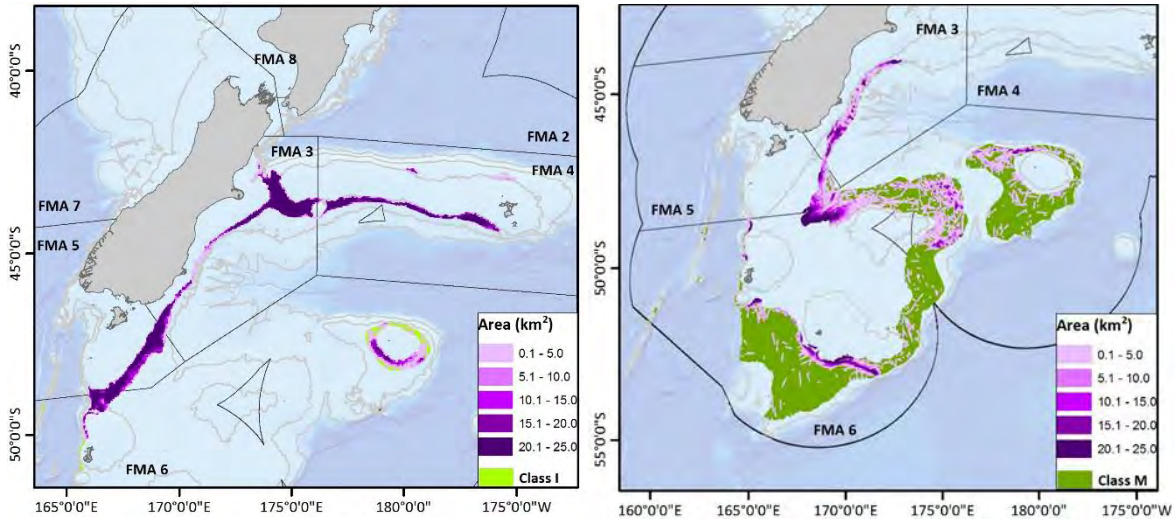


Figure 11.17: Maps created from Baird & Wood (2018) showing BOMEC classes I (left) and M (right) overlaid with the footprint of deepwater Tier 1 and Tier 2 fishstock trawls on or near the seafloor reported on TCER and TCEPR forms between 1989-90 and 2015-16 for each 25-km² cell. Grey contour lines indicate depths of 500 m, 1000 m and 1500 m.

Baird et al. (2015) overlaid the combined TCER and TCEPR 2007–08 to 2011–12 fishing years trawl footprint on a classification for benthic habitats shallower than 250 m (presented in figure 18 of that report). As with the offshore data, the size of the footprint and proportion of each class trawled varied between habitat classes (Table 11.4) and ranged from 21% (class F) to 76% (class B). Over the 2007–08 to 2011–12 period, the trawl footprint overlays 48% of the area shallower than 250 m.

Table 11.4: Areas* of the separate habitat classes and areas† of the five-year trawl footprint in each habitat class, for the BOMEC classes, depth zones, and sediment types, and the percentage of the five-year trawl footprint in each class (see Baird et al. 2015 for more details). *Note: the trawl footprint is based on trawling out to 250 m.

Habitat class descriptors	Habitat class area (km²)	Total footprint (km²)	% area with trawl contact
BOMEC class			
A	27 375.2	14 047.1	52.1
B	12 318.8	9 322.3	75.7
C	89 560.4	47 120.0	52.6
D	25 513.1	16 344.7	64.4
E	47 186.8	13 890.6	29.6
F	381.7	73.2	21.1
G	3 898.4	1 902.9	50.8
H	25 204.4	9 228.1	36.8
I	473.2	340.8	72.0
J	133.9	31.3	30.0
L	188.9	121.5	67.6
Depth zone			
< 50 m	50 781.3	29 529.4	58.8
50–100 m	63 493.9	37 375.2	59.1
100–250 m	117 959.8	45 517.9	38.7
Sediment type			
Sand	104 830.2	54 851.9	52.4
Mud	72 518.1	41 001.8	57.0
Gravel	18 530.0	9 339.3	50.6
Sandy mud	203.4	203.4	99.98
Calcareous sand	6 763.7	2 080.7	31.3
Calcareous gravel	29 389.5	4 945.4	17.0
All	232 235.0	112 422.5	48.4

* The area measures for the habitat classes include any seafloor closed to trawling.

† The area measures for the five-year footprint represent 98.8% of the total footprint.

11.3.4 STUDIES OF THE EFFECTS OF MOBILE BOTTOM FISHING METHODS IN NEW ZEALAND

The widespread nature of bottom trawling suggests that fishing is the main anthropogenic disturbance agent to the seabed throughout most of New Zealand's EEZ. Wind waves are certainly very widespread, but both field studies and modelling (Green et al. 1995) suggest that erosion of the seabed deeper than 50 m by waves occurs only very rarely in the New Zealand EEZ. Despite their widespread distribution at the surface, therefore, wind waves are not a dominant feature of the long-term disturbance regime throughout most of the EEZ. In some places, especially in the coastal zone and in areas close to headlands, straits, or islands, currents and tides may dominate the natural disturbance regime and a community adapted to this type of disturbance will have developed. However, over most of the EEZ between about 100 and 1000 m depth, especially in areas where there are few strong currents, fishing is probably the major broad-scale disturbance agent.

Several studies have been conducted since 1995 in New Zealand, focusing on the effects of various dredge and trawl fishing methods on a variety of different habitats in several geographical locations (Table 11.5). Despite the diversity of these studies, and their different depths, locations, and habitat types, the results are consistent with the global literature on the effects of mobile bottom fishing gear on benthic communities. Generally, there are decreases in the density and diversity of benthic communities and, especially, the density of large, structure-forming epifauna, and long-lived organisms along gradients of increasing fishing intensity. Large, emergent epifauna such as sponges and framework-forming corals that provide structured habitat for other fauna are particularly noted as being susceptible to disturbance by mobile bottom fishing methods (Cranfield et al. 1999, 2001, 2003, Cryer et al. 2000), especially on hard (non-sedimentary) seabeds (Clark & Rowden 2009, Clark et al. 2010a, 2010b, Williams et al. 2011). Even though large emergent fauna seem most susceptible, effects have also been shown in the sandy or silty sedimentary systems usually considered to be most resistant to disturbance (Thrush et al. 1995, 1998, Cryer et al. 2002). Also reflecting the international literature is a substantial variation in the extent to which individual New Zealand studies have shown clear effects. For instance, in Foveaux Strait, Cranfield et al. (1999, 2001, 2003) inferred

substantial changes in the benthic system caused by over 130 years of oyster dredging, but Michael et al. (2006) did not support such conclusions in the same system. Subsequent review of these studies found much common ground but no overall consensus on the long-term effects of dredging on the benthic community of the strait.

These studies have focused predominantly on changes in patterns in biodiversity associated with trawling and/or dredging and less work has been done to assess changes in ecological process or to estimate the rate of recovery from fishing. Projects that have started on recovery rates are focused on relatively few habitats and primarily those that are known to be sensitive to physical disturbance, including by trawling or dredging (e.g., seamounts, project ENV2005/16, and areas of high current and natural biogenic structure, projects ENV9805, ENV2005/23 and BEN2009/02). Thus, the understanding of the consequences of fishing (or of ceasing to fish) for sustainability, biodiversity, ecological integrity and resilience, and fish stock productivity in the wide variety of New Zealand's benthic habitats remains incomplete. Reducing this uncertainty would allow the testing of the utility and likely long-term productivity of a variety of management strategies, and enable a move towards a regime that maximises value to the nation consistent with the MPI 'Our Strategy' document (http://www.mpi.govt.nz/about-mpi/our-strategy).

An expert-based assessment of 65 threats to 62 marine habitats from saltmarsh to the abyss (MacDiarmid et al. 2012) concluded that only 7 of the 20 most important threats to New Zealand marine habitats were directly related to human activities within the marine environment. The most important of these was bottom trawling (ranked third-equal most important), but invasive species, coastal engineering, and aquaculture were also ranked highly. However, the two top threats, five of the top six threats, and over half of the 26 top threats stemmed largely or completely from human activities external to the marine environment (the most important being ocean acidification, rising sea temperatures, and sedimentation resulting from changes in land use). The assessment suggested that the number and severity of threats to marine habitats declines with depth, particularly deeper than about 50 m. Shallow coastal habitats face up to 52 non-trivial threats whereas most deepwater habitats are threatened by fewer than five. Coastal and estuarine reef, sand, and mud habitats were considered to be the most threatened habitats whereas slope and deepwater habitats were among the least threatened.

Table 11.5: Summary of studies of the effects of bottom trawling and dredging in New Zealand waters. [Continued on next page]

Location	Approach	Key findings	References
Mercury Islands sandy sediments. Scallop dredge	Experimental	Density of common macrofauna at both sites decreased as a result of dredging at two contrasting sites; some populations were still significantly different from reference plots after three months.	Thrush et al. 1995
Hauraki Gulf various soft sediments. Bottom trawl and scallop dredge.	Observational, gradient analysis	Decreases in the density of echinoderms, longlived taxa, epifauna, especially large species, the total number of species and individuals, and the Shannon-Weiner diversity index with increasing fishing pressure (including trawl and scallop dredge). Increases in the density of deposit feeders, small opportunists, and the ratio of small to large heart urchins.	Thrush et al. 1998
Bay of Plenty continental slope. Scampi and other bottom trawls.	Observational, multiple gradient analyses	Depth and historical fishing activity (especially for scampi) at a site were the key drivers of community structure for large epifauna. The Shannon-Weiner diversity index generally decreased with increasing fishing activity and increased with depth. Many species were negatively correlated with fishing activity; fewer were positively correlated (including the target species, scampi).	Cryer et al. 1999 Cryer et al. 2002
Foveaux Strait, sedimentary and biogenic reef. Oyster dredge.	Observational, various	Interpretations of the authors differ. Cranfield et al.'s papers concluded that dredging biogenic reefs for their oysters damages their structure, removes epifauna, and exposes associated sediments to resuspension such that, by 1998, none of the original bryozoan reefs remained. Michael et al. concluded that there are no experimental estimates of the effect of dredging in the strait or on the cumulative effects of fishing or regeneration, that environmental drivers should be included in any assessment, and that the previous conclusions cannot be supported.	Cranfield et al. 1999, 2001, 2003 Michael et al. 2006

Location	Approach	Key findings	References
		The authors agree that biogenic bycatch in the fishery has declined over time in regularly fished areas, that there may have been a reduction in biogenic reefs in the strait since the 1970s, and that simple biogenic reefs appear able to regenerate in areas that are no longer fished (dominated by byssally attached mussels or reef-building bryozoans). There is no consensus that reefs in Foveaux Strait were (or were not) extensive or dominated by the bryozoan *Cinctopora*.	
Spirits Bay, sedimentary and biogenic areas. Scallop dredge.	Observational, gradient analysis	In 1999, depth was found to be the most important explanatory variable for benthic community composition but a coarse index of dredge fishing intensity was more important than substrate type for many taxonomic groups. Sponges seemed most affected by scallop dredging, and samples taken in an area once rich in sponges had few species in 1999. This area had probably been intensively dredged for scallops. Analysis of historical samples of scallop survey bycatch showed a marked decline in sponge species richness between 1996 and 1998. In 2006, significant differences were identified between areas within which fishing was or was not allowed. Species contributing to these differences included those identified as being most vulnerable to the effects of fishing. These differences could not be attributed specifically to fishing because of interactions with environmental gradients and uncertainty over the history of fishing. No significant change between 1999 and 2006 was identified. In 2010, analysis of both epifaunal and infaunal community data identified change since 2006, and significant depth, habitat and fishing effects. The combined fishing effects accounted for 15–30% of the total variance (about half of the explained variance). Individual species responses to fishing were examined, and those identified as most sensitive to fishing in this analysis had previously been categorised as sensitive on the basis of life history characteristics within the 2006 study.	Cryer et al. 2000 Tuck et al. 2010 Tuck & Hewitt 2013
Tasman and Golden Bays. Bottom trawl, scallop and oyster dredge	Observational, gradient analysis	A gradient analysis was adopted to investigate the importance of the different factors affecting epifaunal and infaunal communities in Tasman and Golden Bays. Fishing was consistently identified as an important factor in explaining variance in community structure, with recent trawl and scallop effort being more important than other fishing terms. Important environmental variables included maximum current speed, maximum wave height, depth, % mud, and salinity. Fishing accounted for 31–50% of the explained variance in epifaunal and infaunal community composition, species richness, and Shannon-Weiner diversity. Overall, models explained 30–54% of variance, and additional spatial patterns identified in the analysis explained a further 5–16% of variance.	Tuck et al. 2017
South Canterbury Bight. Bottom trawl	Observational, gradient analysis	A gradient analysis was adopted to investigate the importance of the different factors affecting epifaunal and infaunal communities in the South Canterbury Bight. Both fishing effort and environmental variables were identified as being important in explaining the patterns in the community data observed, although fishing effort accounted for only a relatively small component of the overall variance (5 – 9%). The important environmental variables for both infaunal and epifaunal community analysis, included sediment grain size and organic carbon parameters, wave height parameters, chlorophyll-*a*, and distance from earthquake epicentres (although it must be remembered that this parameter was correlated with distance north). In addition to the weak (but mostly significant) fishing effects detected in relation to species based community and univariate measures, functional trait effects were also detected, with the predicted factor ceiling response identified for long-lived, sedentary, habitat-forming species, and a significant negative effect of fishing identified on this functional trait group.	Tuck et al. 2017

Location	Approach	Key findings	References
Graveyard complex 'seamounts', northern Chatham Rise. Orange roughy bottom trawl.	Observational, multiple analyses	From surveys in 2001 and 2006, substrate diversity and the amount of intact coral matrix were lower on fished seamounts. Conversely, the proportions of bedrock and coral rubble were higher. No change in the megafaunal assemblage consistent with recovery over 5–10 years on seamounts where trawling had ceased. Some taxa had significantly higher abundance in later surveys. This may be because of their resistance to the direct effects of trawling, their protection in natural refuges, or because these taxa represent the earliest stages of seamount recolonisation.	Clark & Rowden 2009, Clark et al. 2010a, 2010b Williams et al. 2010

11.3.5 CURRENT RESEARCH

Project BEN2017-01 provides for an annual assessment of the 'footprint' of middle depth and deepwater trawl fishstocks, including the overlap of the footprint with various depth ranges and habitat classes. Inshore fisheries, including shellfish dredge fisheries, are not covered under this project, so the focus is on offshore fisheries and habitats. Project BEN2018-01 will provide an assessment of the footprint of inshore trawl and dredge fisheries and an update of the middle depth and deepwater trawl footprint.

Project ZBD2012-03 will use data collected from recent Oceans Survey 20/20 sampling on the Chatham Rise to determine whether there are quantifiable effects of variations in seabed trawling intensity on benthic communities, and also conduct seabed mapping and photographic surveys in previously unsampled areas on the central crest of the rise.

Project BEN2014-02 provided additional support to work started under FRST-funded research on underwater topographic features from 1999. This programme, and its MBIE successor ('Vulnerable Deep-Sea Communities' (CO1X0906)) developed a fishing impact recovery comparison based on repeated towed camera surveys on six of the Graveyard Knolls on the northern flank of Chatham Rise. These knolls cover conditions where trawling has ceased, where trawling is still active, or knolls which have been untrawled. Surveys were carried out in 2001, 2006, 2009, and 2015, with support from MFish, NIWA, the cross-departmental Oceans Survey 20/20 programme, and in 2015 from MPI under BEN2014-02. Results from this time series have recently been published (Clark et al. 2019).

Univariate community metrics of biodiversity (abundance, species richness, diversity) were almost always higher for untrawled Ghoul and Gothic knolls than the other four. Multivariate community analyses of each knoll at each time-step showed a similar pattern, with the untrawled Gothic and Ghoul knolls having similar levels and patterns of community structure at one end of the ordination space, the persistently heavily trawled Graveyard seamount at the other end along with the previously heavily trawled and now closed Morgue knoll, and intermittently trawled knolls lying in-between (Figure 11.18). This ordination matches the gradient in commercial fishing effort. Community structure on Graveyard knoll was more consistent than on the other knolls, with persistently lower faunal richness, possibly due to a regular 're-setting' of the community by disturbance from trawling.

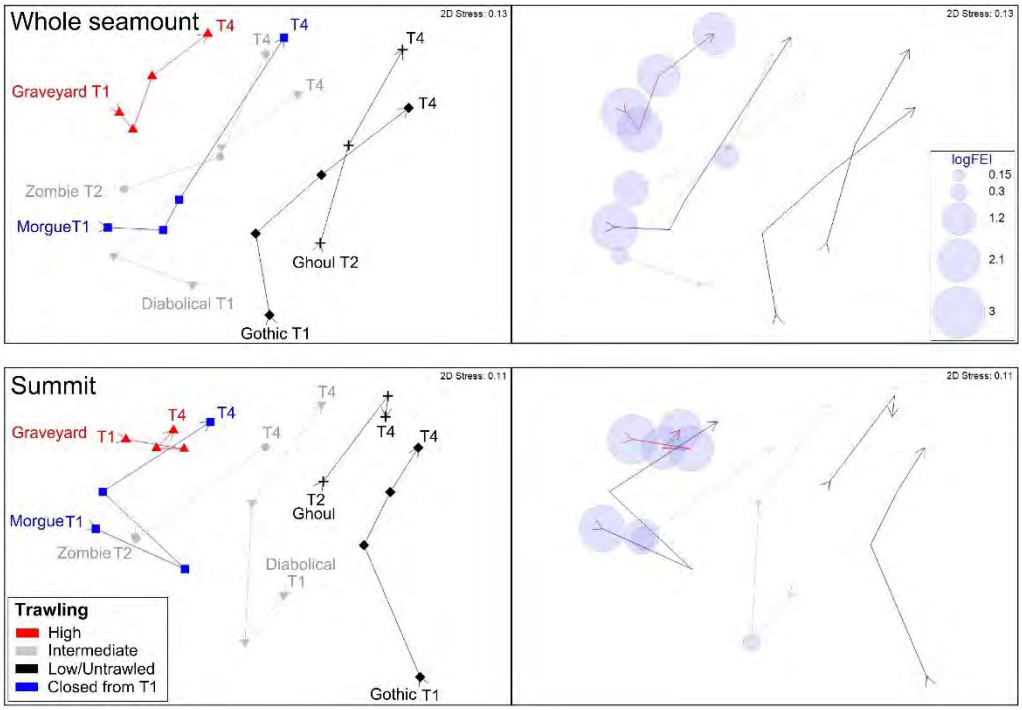


Figure 11.18: Ordinations (nMDS) illustrating benthic community similarities over time steps T1 to T4 and in relation to relative trawl history (see legend): Top panel, ordination using all images ('whole seamount'); bottom panel, only the summit sector. Right windows show trawling intensity as the Fishing Effects Index (FEI) superimposed as bubble plots for each seamount at each time step. From Clark et al. 2019.

The time series of surveys indicates low resilience of benthic communities on the knolls to the effects of bottom trawling. There is no evidence that benthic communities on Morgue knoll are recovering following its closure to fishing in 2001. Intact scleractinian coral 'reef' is or was a "climax habitat" on the Graveyard Knolls, and levels of this habitat on Morgue knoll remain much lower than those on the untrawled knolls (Figure 11.19).

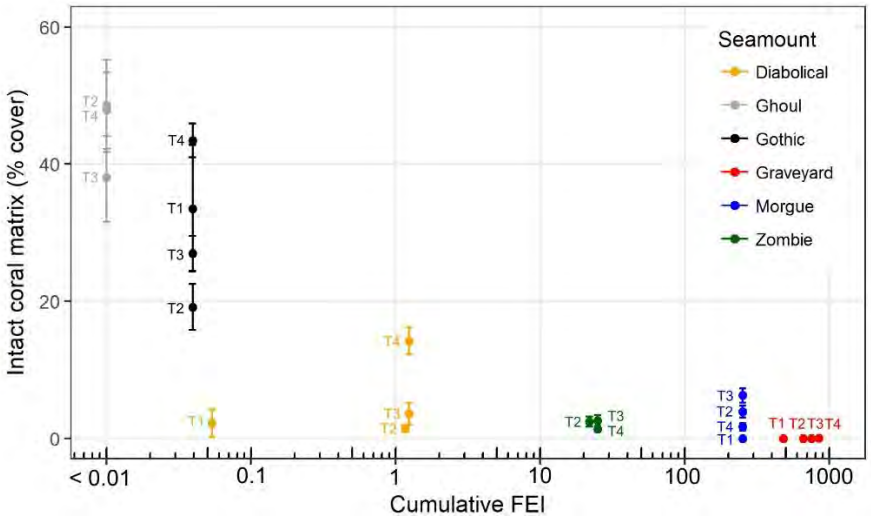


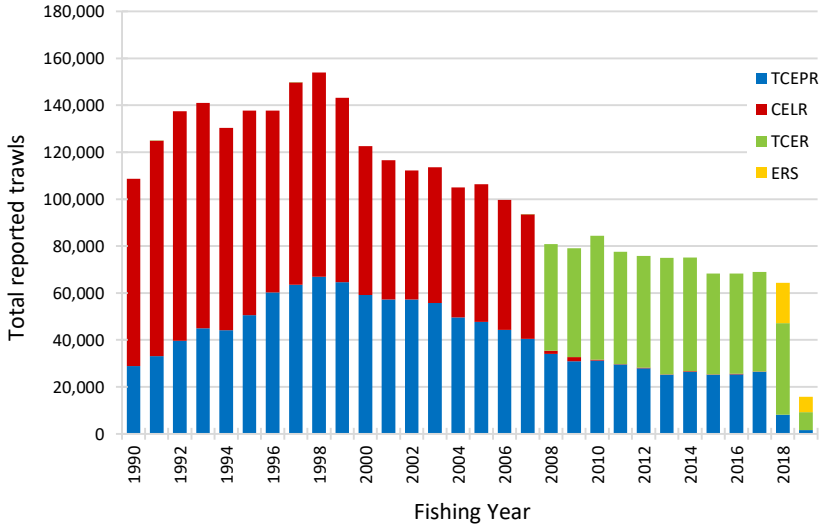
Figure 11.19: Intact coral matrix, as mean percent cover measured in individual seabed photographs, on six seamounts at each of four survey times (2001, 'T1'; 2006, 'T2'; 2009, 'T3' and 2015, 'T4') in relation to cumulative trawling impact (Fishing Effects Index, FEI, all years up to and including the year of each survey). From Clark et al. 2019 .

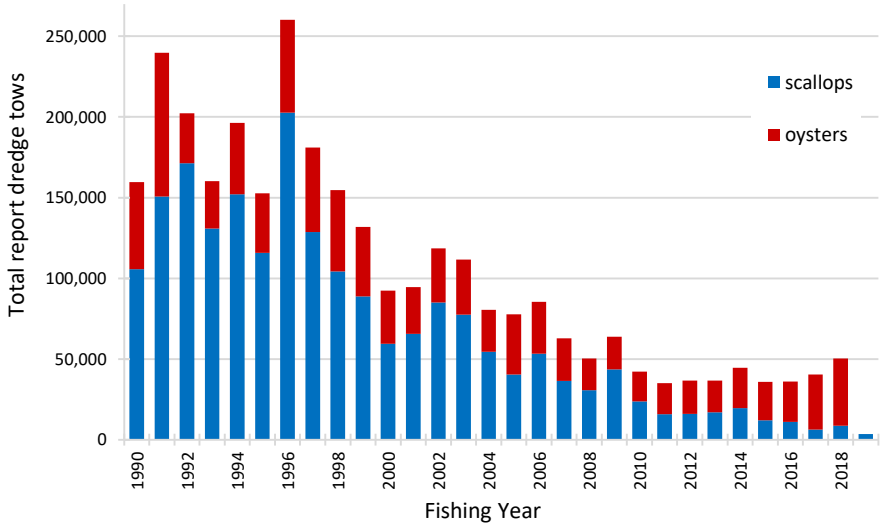
Another linkage with the MBIE Vulnerable Deep-Sea Communities' project is an ongoing evaluation (under NIWA funding) of the relative vulnerability of benthic communities in several deep-sea habitats (e.g., seamounts, canyons, continental slope, hydrothermal vents, seeps) and their risk from bottom trawling. The importance of fishing effort as a factor influencing benthic invertebrate community composition has been examined for epi-megafauna from towed camera data (Bowden et al. 2016) and epibenthic sled tows (Rowden et al. 2016), and from multicorer samples for macro-infauna and meiofauna (Leduc et al. 2016, Rosli et al. 2016). Data are currently being compiled for carrying out an overall risk assessment of the relative susceptibility and recoverability of communities in the different habitats based on ecological traits.

Benthic community data from surveys conducted in 2006, 2010 and 2017 was analysed to investigate the effects of fishing in the Spirits Bay area, and recovery following the closure of areas within this region (under projects BEN2014-03 and ZBD2017-05). Multivariate and univariate analyses of epifaunal and infaunal community data from the Spirits Bay area consistently identified year, habitat and depth effects, but scallop and trawl fishing were also retained in minimum adequate models (accounting for a median level of 20% of the total variance, and up to 50% of the explained variance), with effects still detectable 7–9 years after fishing in some analyses. The effects detected were independent of similarity measure, analysis approach or data set used, and as we might expect, the effects of fishing were weaker in analyses of more recent survey data, where recent fishing effort was lower. Species sensitivities, categorised on the basis of morphology and life history characteristics, were consistent with species responses to fishing terms within the modelled analysis, and most of the most sensitive species were only found in areas with no recent fishing history.

11.4 INDICATORS AND TRENDS

Annual number of tows	2017–18 fishing year: 65 133 trawl tows 50 288 shellfish dredge tows
Trend in number of tows	Trawl and dredge effort stable or decreasing in recent years [Updated for AEBAR 2018]:





Cumulative overlap of TCEPR trawl footprint with BOMEC habitat classes for 2007–08 to 2016–17

BOMEC class	Area (km²)	Footprint area (km²)	Footprint area (%)
A*	27 557	364.1	1.3
B*	12 420	2 663.2	21.4
C*	89 710	22 638.7	25.2
D*	27 268	1 754.1	6.4
E*	60 990	12 764.0	20.9
F	38 608	3 787.0	9.8
G	6 342	2 117.4	33.4
H	138 550	39 171.8	28.3
I	52 224	26 802.3	51.3
J	311 361	35 637.7	11.4
K	1 290	0.0	0.0
L	198 577	23 720.5	11.9
M	233 825	4 574.1	2
N	493 034	3 398.7	0.7
O	935 315	566.7	0.1
Total	2 627 073	179 960.5	6.9

Cumulative overlap of trawl footprint shallower than 250 m with BOMEC habitat classes, depth zones and sediment types for 2007–08 to 2011–12

Habitat class descriptors	Habitat class area (km²)	Total footprint (km²)	% area with trawl contact
BOMEC class			
A	27 375.2	14 047.1	52.1
B	12 318.8	9 322.3	75.7
C	89 560.4	47 120.0	52.6
D	25 513.1	16 344.7	64.4
E	47 186.8	13 890.6	29.6
F	381.7	73.2	21.1
G	3 898.4	1 902.9	50.8
H	25 204.4	9 228.1	36.8
I	473.2	340.8	72.0
J	133.9	31.3	30.0
L	188.9	121.5	67.6
Depth zone			
< 50 m	50 781.3	29 529.4	58.8
50–100 m	63 493.9	37 375.2	59.1
100–250 m	117 959.8	45 517.9	38.7
Sediment type			
Sand	104 830.2	54 851.9	52.4
Mud	72 518.1	41 001.8	57.0
Gravel	18 530.0	9 339.3	50.6
Sandy mud	203.4	203.4	99.98
Calcareous sand	6 763.7	2 080.7	31.3
Calcareous gravel	29 389.5	4 945.4	17.0
All	232 235.0	112 422.5	48.4

11.5 REFERENCES

Anderson, O F; Guinotte, J; Rowden, A A; Tracey, D M; Mackay, K; Clark, M R (2016a) Habitat suitability models for predicting the occurrence of vulnerable marine ecosystems in the seas around New Zealand. *Deep Sea Research I 115:* 265–292.

Anderson, O F; Guinotte, J M; Rowden, A A; Clark, M R; Mormede, S; Davies, A J; Bowden, D A (2016b) Field validation of habitat suitability models for vulnerable marine ecosystems in the South Pacific Ocean: implications for the use of broad-scale models in fisheries management. *Ocean and Coastal management 120*: 110–126.

Auster, P J; Langton, R W (1999) The effects of fishing on fish habitat. *American Fisheries Society Symposium 22*: 150–187.

Barnes, P W; Thomas, J P (Eds.) (2005) Benthic habitats and effects of fishing. *American Fisheries Society Symposium 41*. American Fisheries Society, Bethesda, MD.

Baird, S J; Bagley, N W; Wood, B A; Dunn, A; Beentjes, M (2002) The spatial extent and nature of mobile bottom fishing methods within the New Zealand EEZ, 1989–90 to 1998–99. Final Research Report for project ENV2000/05, Objective 1. (Unpublished report held by Fisheries New Zealand, Wellington.)

Baird, S J; Hewitt J; Wood B A (2015) Benthic habitat classes and trawl fishing disturbance in New Zealand waters shallower than 250m. *New Zealand Aquatic Environment and Biodiversity Report No. 144.* 184 p.

Baird, S J; Mules, R (2018) Extent of bottom contact by New Zealand commercial trawl fishing for deepwater Tier 1 and Tier 2 target species determined using CatchMapper software, 2008–17. Draft New Zealand Aquatic Environment and Biodiversity Report prepared for BEN201701. 87 p.

Baird, S J; Wood, B A (2018) Extent of bottom contact by New Zealand commercial trawl fishing for deepwater Tier 1 and Tier 2 target fishstocks, 1989–90 to 2015–16. *New Zealand Aquatic Environment and Biodiversity Report No. 193*. 102 p.

Baird, S J; Wood, B A; Bagley, N W (2011) Nature and extent of commercial fishing effort on or near the seafloor within the New Zealand 200 n. mile Exclusive Economic Zone 1989/90 to 2004/05. *New Zealand Aquatic Environment and Biodiversity Report No. 73*. 144 p.

Baird, S J; Wood, B A; Clark, M R; Bagley, N W; McKenzie, A (2006) Description of the spatial extent and nature of disturbances by bottom trawls in Chatham Rise and Southern Plateau fisheries. Final Research Report for project ENV2003/03. 139 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Black, J; Wood, R; Berthelsen, T; Tinley, R (2013) Monitoring New Zealand's trawl footprint for deepwater fisheries: 1989–90 to 2009–10. *New Zealand Aquatic Environment and Biodiversity Report No. 110*. 61 p.

Black, J; Tilney, R (2017) Monitoring New Zealand's trawl footprint for deepwater fisheries: 1989/90 to 2011/12 and 1989/90 to 2012/13. *New Zealand Aquatic Environment and Biodiversity Report No.*176. 65 p.

Blyth, R E; Kaiser, M J; Edwards-Jones, G; Hart, P J B (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology 41*: 951–961.

Bowden, D A; Clark, M R; Hewitt, J E; Rowden, A A; Leduc, D; Baird, S J (2015) Designing a programme to monitor trends in deep-water benthic communities. *New Zealand Aquatic Environment and Biodiversity Report No. 143.* 61 p.

Bowden, D A; Compton, T J; Snelder, T H; Hewitt, J E (2011) Evaluation of the New Zealand Marine Environment Classifications using Ocean Survey 20/20 data from Chatham Rise & Challenger Plateau. *New Zealand Aquatic Environment and Biodiversity Report No. 77*.

Bowden, D A; Rowden, A A; Leduc, D; Beaumont, J; Clark, M R (2016) Deep-sea seabed habitats: do they support distinct mega-epifaunal benthic communities that have different vulnerabilities to anthropogenic disturbance? *Deep Sea Research I. 107*: 31–47.

Bradshaw, C; Tjensvoll, I; Sköld, M; Allan, I J; Molvaer, J; Magnusson, J; Naes, K; Nilsson, H C (2012) Bottom trawling resuspends sediment and releases bioavailable contaminants in a polluted fjord. *Environmental Pollution 170*: 232–241.

Bradshaw, C; Veale, L O; Brand, A R (2002) The role of scallop-dredge disturbance in long-term changes in Irish Sea benthic communities: a re-analysis of an historical dataset. *Journal of Sea Research 47*: 161–184.

Brodie, S; Clark, M (2003) The New Zealand Seamount Management Strategy – steps towards conserving offshore marine habitat. *In*: Beumer, J P; Grant, A; Smith, D C (Eds.), Aquatic Protected Areas: what works best and how do we know? Proceedings of the World Congress on Aquatic Protected Areas, Cairns, Australia, August 2002. pp 664–673. Australian Society of Fish Biology, Australia.

Churchill, J H (1989) The effect of commercial trawling on sediment resuspension and transport over the Middle Atlantic Bight Continental Shelf. *Continental Shelf Research 9*: 841–864.

Clark, M R; Bowden, D A; Baird, S J; Stewart, R (2010a) Effects of fishing on the benthic biodiversity of seamounts of the 'Graveyard' complex, northern Chatham Rise. *New Zealand Aquatic Environment and Biodiversity Report No. 46*. 40 p.

Clark, M.R.; Bowden, D.A.; Rowden, A.A.; Stewart, R. (2019). Little evidence of benthic community resilience to bottom trawling on seamounts after 15 years. *Frontiers in Marine Science 6:63*. doi: 10.3389/fmars.2019.00063

Clark, M R; Koslow, J A (2007) Impacts of fisheries on seamounts. Chapter 19. *In*: Pitcher, T J; Morato, T; Hart, P J B; Clark, M R; Haggan, N; Santos, R S (Eds.), Seamounts: ecology, fisheries, and conservation. pp 413–441. *Blackwell Fisheries and Aquatic Resources Series 12*. Blackwell Publishing, Oxford. 527 p.

Clark, M; O'Driscoll, R (2003) Deepwater fisheries and aspects of their impact on seamount habitat in New Zealand. *Journal of Northwest Atlantic Fishery Science 31*: 441–458.

Clark, M R; Rowden, A A (2009) Effect of deepwater trawling on the macro-invertebrate assemblages of seamounts on the Chatham Rise, New Zealand. *Deep Sea Research I 56*: 1540–1554.

Clark, M R; Watling, L; Rowden, A A; Guinotte, J M; Smith, C R (2010b) A global seamount classification to aid the scientific design of marine protected area networks. *Journal of Ocean and Coastal Management 54*: 19–36.

Craeymeersch, J A; Piet, G J; Rijnsdorp, A D; Buijs, J (2000) Distribution of macrofauna in relation to the microdistribution of trawling effort. *In*: Kaiser, M J; de Groot, S J (Eds.), The effects of fishing on nontarget species and habitats: biological, conservation, and socio-economic issues. p 187. Blackwell Scientific, Oxford, UK.

Cranfield, H J; Carbines, G; Michael, K P; Dunn, A; Stotter, D R; Smith, D J (2001) Promising signs of regeneration of blue cod and oyster habitat changed by dredging in Foveaux Strait, southern New Zealand. *New Zealand Journal of Marine and Freshwater Research 35*: 897–908.

Cranfield, H J; Manighetti, B; Michael, K P; Hill, A (2003) Effects of oyster dredging on the distribution of bryozoans biogenic reefs and associated sediments in Foveaux Strait, southern New Zealand. *Continental Shelf Research 23*: 1337–1357.

Cranfield, H J; Michael, K P; Doonan, I J (1999) Changes in the distribution of epifaunal reefs and oysters during 130 years of dredging for oysters in Foveaux Strait, southern New Zealand. *Aquatic Conservation of Marine and Freshwater Ecosystems 9*: 461–483.

Cryer, M; Coburn, R; Hartill, B H; O'Shea, S; Kendrick, T; Doonan, I (1999) Scampi stock assessment for 1998 and an analysis of the fish and invertebrate bycatch of scampi trawlers. *New Zealand Fisheries Assessment Research Document 99/4*. 74 p. (Unpublished document Fisheries New Zealand, Wellington.)

Cryer, M; Hartill, B H (2002) Relative impacts on benthic invertebrate community structure of trawl fisheries for scampi and finfish in the Bay of Plenty. Final Research Report for Ministry of Fisheries research project ENV2000/05, Objective 2. 28 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Cryer, M; Hartill, B H; O'Shea, S (2002) Modification of marine benthos by trawling: toward a generalization for the deep ocean? *Ecological Applications 12*: 1824–1839.

Cryer, M; O'Shea, S; Gordon, D P; Kelly, M; Drury, J D; Morrison, M A; Hill, A; Saunders, H; Shankar, U; Wilkinson, M; Foster, G (2000) Distribution and structure of benthic invertebrate communities between North Cape and Cape Reinga Final Research Report for Ministry of Fisheries research project ENV9805, Objectives 1–4. (Unpublished report held by Fisheries New Zealand, Wellington.)

Davies, A J; Guinotte, J M (2011) Global habitat suitability for framework-forming cold-water corals. *PLoS One 6 (4)*: e18483.

Dayton, P K; Thrush, S F; Agardy, M T; Hofman, R J (1995). Environmental effects of fishing. *Aquatic Conservation: Marine and Freshwater Ecosystems 5*: 205–232.

Dayton, P K; Thrush, S F; Coleman, F C (2002) The Ecological Effects of Fishing in Marine Ecosystems of the United States. Pew Oceans Comm, Arlington, VA.

Dichmont, C M; Deng, A; Punt, A E; Ellis, N; Venables, W N; Kompas, T; Ye, Y; Zhou, S; Bishop, J (2008) Beyond biological performance measures: bringing in economics and the effects of trawling on the benthos. *Fisheries Research 94*: 238–250.

Ellis, N; Pantus, F (2001) Management strategy modelling: tools to evaluate trawl management strategies with respect to impacts on benthic biota within the Great Barrier Reef Marine Park area. Cleveland, Qld. CSIRO Div. Marine Research. Parts 1 & 2. 132 p. Retrieved from http://www.cmar.csiro.au/e-print/open/ellisn_2001.pdf.

Ferrier, S; Drielsma, M; Manion, G; Watson, G (2002) Extended statistical approaches to modelling spatial pattern in biodiversity in north-east New South Wales: II. Community-level modelling. *Biodiversity and Conservation 1*: 2309–2338.

Ferrier, S; Manion, G; Elith, J; Richardson, K (2007) Using generalised dissimilarity modelling to analyse and predict patterns of beta-diversity in regional biodiversity assessment. *Diversity and Distributions 13*: 252–264.

Green, M O; Vincent, C E; McCave, I N; Dickson, R R; Rees, J M; Pearsons, N D (1995) Storm sediment transport: observations from the British North Sea shelf. *Continental Shelf Research 15*: 889–912.

Hall, S J (1994) Physical disturbance and marine benthic communities: life in unconsolidated sediments. *Oceanography and Marine Biology Annual Review 32*: 179–239.

Hall, S J (1999) The effects of fishing on marine ecosystems and communities. Blackwell Scientific, Oxford, UK.

Helson, J; Leslie, S; Clement, G; Wells, R; Wood, R (2010) Private rights, public benefits: Industry-driven seabed protection. *Marine Policy 34*: 557–566.

Hermsen, J M; Collie, J S; Valentine, P C (2003) Mobile fishing gear reduces benthic megafaunal production on Georges Bank. *Marine Ecology Progress Series 260*: 97–108.

Hiddink, J G; Jennings, S; Kaiser, M J; Queiros, A M; Duplisea, D E; Piet, G J (2006) Cumulative impacts of seabed trawl disturbance on benthic biomass, production, and species richness in different habitats. *Canadian Journal of Fisheries and Aquatic Sciences 63*: 721–736.

Hiddink, J G; Johnson, A F; Kingham, R; Hinz, H (2011) Could our fisheries be more productive? Indirect negative effects of bottom trawl fisheries on fish condition. *Journal of Applied Ecology 48*: 1–9.

Hinz, H; Hiddink, J G; Forde, J; Kaiser, M J (2008) Large-scale responses of nematode communities to chronic otter-trawl disturbance. *Canadian Journal of Fisheries and Aquatic Sciences 65*: 723–732.

ICES (1992) Report of the Study Group on Ecosystem Effects of Fishing Activities. ICES CM:1992/G11. 144 p.

ICES (2000a) Report of the Advisory Committee on the Marine Environment 2000. *ICES Cooperative Research Report 241*.

ICES (2000b) Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2000/ACME:02.

Jennings, S; Dinmore, T A; Duplisea, D E; Warr, K J; Lancaster, J E (2001a) Trawling disturbance can modify benthic production processes. *Journal of Animal Ecology 70*: 459–475.

Jennings, S; Freeman, S; Parker, R; Duplisea, D E; Dinmore, T A (2005) Ecosystem Consequences of Bottom Fishing Disturbance. *American Fisheries Society Symposium 41*: 73–90.

Jennings, S; Kaiser, M J (1998) The effects of fishing on marine ecosystems. *Advances in Marine Biology and Ecology 34*: 201–352.

Jones, J B (1992) Environmental impact of trawling on the seabed: a review. *New Zealand Journal of Marine and Freshwater Research 26*: 59–67.

Kaiser, M J; Clarke, K R; Hinz, H; Austen, M C V; Somerfield, P J; Karakassis, I (2006) Global analysis of the response and recovery of benthic biota to fishing. *Marine Ecology Progress Series 311*: 1–14.

Kaiser, M J; Collie, J S; Hall, S J; Jennings, S; Poiner, I R (2003) Impacts of fishing gear on marine benthic habitats. Conference on Responsible Fisheries in the Marine Ecosystem. Reykjavik, Iceland, 1–4 October 2001.

Kaiser, M J; de Groot, S J (2000) Effects of fishing on non-target species and habitats: biological, conservation and socio-economic issues. Blackwell, Oxford. 399 p.

Lambert, G I; Jennings, S; Kaiser, M J; Davies, T W; Hiddink, J G (2014) Quantifying recovery rates and resilience of seabed habitats impacted by bottom fishing. *Journal of Applied Ecology 51 (5)*: 1326–1336. doi:10.1111/1365-2664.12277.

Leduc, D; Rowden, A A; Clark, M R; Bowden, D A; Thurber, A R (2016) Limited differences among habitats in deep-sea macro-infaunal communities off New Zealand: implications for their vulnerability to anthropogenic disturbance. *Marine Ecology 37*: 845–866.

Leathwick, J; Francis, M; Julian, K (2006) Development of a demersal fish community classification for New Zealand's Exclusive Economic Zone. *NIWA Client Report HAM2006-062*, prepared for Department of Conservation, Hamilton.

Leathwick, J R; Rowden, A; Nodder, S; Gorman, R; Bardsley, S; Pinkerton, M; Baird, S J; Hadfield, M; Currie, K; Goh, A (2012) A Benthic–optimised Marine Environment Classification (BOMEC) for New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 88*. 54 p.

Leathwick, J R; Snelder, T; Chadderton, W L; Elith, J; Julian, K; Ferrier, S (2011) Use of generalised dissimilarity modelling to improve the biological discrimination of river and stream classifications. *Freshwater Biology 56*: 21–38.

Lindeboom, H; deGroot, S J (1998) The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems. Netherlands Institute of Sea Research, Texel, The Netherlands.

Lokkeborg, S (2005) Impacts of trawling and scallop dredging on benthic habitats and communities. *FAO Technical Paper 472*. FAO, Rome.

Lundquist, C J; Pritchard M; Thrush S F; Hewitt J; Greenfield B L; Halliday J; Lohrer A M (2013) Bottom disturbance and seafloor community dynamics: Development of a model of disturbance and recovery dynamics for marine benthic ecosystems. *New Zealand Aquatic Environment and Biodiversity Report No. 118*. 63 p.

Lundquist, C J; Thrush, S F; Coco, G; Hewitt, J E (2010) Interactions between disturbance and dispersal reduce persistence thresholds in a benthic community. *Marine Ecology Progress Series 413*: 217–228.

McConnaughy, R A; Mier, K L; Dew, C B (2000) An examination of chronic trawling effects on soft-bottom benthos of the eastern Bering Sea. *ICES Journal of Marine Science 57*: 1377–1388.

MacDiarmid, A; McKenzie, A; Sturman, J; Beaumont, J; Mikaloff-Fletcher, S; Dunne, J (2012) Assessment of anthropogenic threats to New Zealand marine habitats. *New Zealand Aquatic Environment and Biodiversity Report No. 93*. 255 p.

Michael, K P; Kroger, K; Richardson, K; Hill, N (2006) Summary of information in support of the Foveaux Strait Oyster Fishery Plan: the Foveaux Strait ecosystem and effects of oyster dredging. Final Research Report for Ministry of Fisheries project ZBD2005/04. (Unpublished report held by Fisheries New Zealand, Wellington.)

Ministry for the Environment (2010) Fishing Activity: Fish Stocks Environmental Snapshot. Wellington.

Mormede, S; Dunn, A (2013) An initial development of spatially explicit population models of benthic impact to inform Ecological Risk Assessments in New Zealand deepwater fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 106*. 20 p.

National Marine Fisheries Service (2002) Workshop on the effects of fishing gear on marine habitats off the northeastern United States. 23–25 October 2001. Northeast Fisheries Science Center Reference Document 02-01.

National Research Council (2002) Effects of trawling and dredging in seafloor habitat. National Academy Press, Washington DC.

O'Driscoll, R L; Clark, M R (2005) Quantifying the relative intensity of fishing on New Zealand seamounts. *New Zealand Journal of Marine and Freshwater Research 39*: 839–850.

Puig, P; Canals, M; Company, J B; Martin, J; Amblas, D; Lastras, G; Palanques, A; Calafat, A (2012) Ploughing the deep sea floor. *Nature 489*: 286–289.

Pusceddu, A; Bianchelli, S; Martín, J; Puig, P; Palanques, A; Masqué, P; Danovaro, R (2014) Chronic and intensive bottom trawling impairs deep-sea biodiversity and ecosystem functioning. *Proceedings of the National Academy of Sciences 111 (24)*: 8861–8866. doi:10.1073/pnas.1405454111.

Reiss, H; Greenstreet, S P R; Siebe, K; Ehrich, S; Piet, G J; Quirijns, F; Robinson, L; Wolff, W J; Kronke, I (2009) Effects of fishing disturbance on benthic communities and secondary production within an intensively fished area. *Marine Ecology Progress Series 394*: 201–213.

Rice, J (2006) Impacts of mobile bottom gears on seafloor habitats, species, and communities: a review and synthesis of selected international reviews. Canadian Science Advisory Secretariat Research Document 2006/057. 35 p. Retrieved from http://www.dfo-mpo.gc.ca/CSAS/Csas/DocREC/2006/RES2006_057_e.pdf.

Rosli, N; Leduc, D; Rowden, A A; Clark, M R; Probert, P K; Berkenbusch, K; Neira, C (2016) Differences in meiofauna communities with sediment depth are greater than habitat effects on the New Zealand continental margin: implications for vulnerability to anthropogenic disturbance. *Peer J 4*: e2154; doi 10.7717/peerj.2154. : e2154; doi 10.7717/peerj.2154. 39p.

Rowden, A A; Clark, M R; Wright, I C (2005) Physical characterisation and a biologically focused classification of seamounts in the New Zealand region. *New Zealand Journal of Marine and Freshwater Research 39*: 1039–1059.

Rowden, A A; Oliver, M D; Clark, M R; MacKay, K (2008) New Zealand's 'SEAMOUNT' database: recent updates and its potential use for ecological risk assessment. *New Zealand Aquatic Environment and Biodiversity Report No. 27*. 49 p.

Rowden, A A; Leduc, D; Clark, M R; Bowden, D A (2016) Habitat differences in deep-sea megafaunal communities off New Zealand: Implications for vulnerability to anthropogenic disturbance and management. *Frontiers in Marine Science 3 (241*). 19 p. DOI: 10.3389/fmars.2016.00241.

Rumohr, H (1998) Information on impact of trawling on benthos in Kiel Bay. Annex to Eighth Report of the Benthos Ecology Working Group. ICES, CM: 1989/L19, 80.

Sainsbury, K J; Punt, A E; Smith, A D M (2000) Design of operational management strategies for achieving fishery ecosystem objectives. *ICES Journal of Marine Science 57*: 731–741.

Schratzberger, M; Lampadariou, N; Somerfield, P J; Vandepitte, L; Vanden Berghe, E (2009) The impact of seabed disturbance on nematode communities: linking field and laboratory observations. *Marine Biology 156*: 709–724.

Sharp, B; Parker, S; Smith, N (2009) An impact assessment framework for bottom fishing methods in the CCAMLR Convention area. *CCAMLR Science 16*: 195–210.

Snelder, T H; Leathwick, J R; Dey, K L; Rowden, A A; Weatherhead, M A; Fenwick, G D; Francis, M P; Gorman, R M; Grieve, J M; Hadfield, M G; Hewitt, J E; Richardson, K M; Uddstrom, M J; Zeldis, J R (2006) Development of an ecological marine classification in the New Zealand region. *Environmental Management 39*: 12–29.

Snelder, T H; Leathwick, J R; Dey, K L; Weatherhead, M A; Fenwick, G D; Francis, M P; Gorman, R M; Grieve, J M; Hadfield, M G; Hewitt, J E; Hume, T; Richardson, K M; Rowden, A A; Uddstrom, M J; Wild, M; Zeldis, J R (2005) The New Zealand Marine Environment Classification. Ministry for the Environment, Wellington.

Snelder, T H; Leathwick, J R; Image, K; Weatherhead, M A; Wild, M (2004) The New Zealand Marine Environment Classification. *NIWA Client Report CHC2004-071*. 86 p.

Thrush, S F; Dayton, P K (2002) Disturbance to marine benthic habitats by trawling and dredging – implications for marine biodiversity. *Annual Reviews in Ecology and Systematics 33*: 449–473.

Thrush, S F; Hewitt, J E; Cummings, V J; Dayton, P K (1995) The impact of habitat disturbance by scallop dredging on marine benthic communities: What can be predicted from the results of experiments? *Marine Ecology Progress Series 129*: 141–150.

Thrush, S F; Hewitt, J E; Cummings, V J; Dayton, P K; Cryer, M; Turner, S J; Funnell, G A; Budd, R G; Milburn, C J; Wilkinson, M R (1998) Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. *Ecological Applications 8*: 866–879.

Tillin, H M; Hiddink, J G; Jennings, S; Kaiser, M J (2006) Chronic bottom trawling alters the functional composition of benthic invertebrate communities on a sea-basin scale. *Marine Ecology Progress Series 318*: 31–45.

Tuck, I D; Drury, J; Kelly, M; Gerring, P (2010) Designing a programme to monitor the recovery of the benthic community between North Cape and Cape Reinga. *New Zealand Aquatic Environment and Biodiversity Report No. 53*. 79 p.

Tuck, I D; Hewitt, J E (2013) Monitoring change in benthic communities in Spirits Bay. *New Zealand Aquatic Environment and Biodiversity Report No. 111*. 52 p.

Tuck, I D; Hewitt, J E; Handley, S; Lundquist, C L (2017) Assessing the effects of fishing on soft sediment habitat, fauna and processes. *New Zealand Aquatic Environment and Biodiversity Report No. 178*. 147 p.

Tuck, I D; Parkinson, D M; Dey, K; Oldman, J; Wadwha, S (2006) Information on benthic impacts in support of the Coromandel scallop fisheries plan. Final Research Report for Ministry of Fisheries research project ZBD2005/15. (Unpublished report held by Fisheries New Zealand, Wellington.)

Tuck, I D; Pinkerton, M H; Tracey, D M; Anderson, O A; Chiswell, S M (2014) Ecosystem and Environmental Indicators for Deepwater Fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 127*. 149 p.

Veale, L O; Hill, A S; Hawkins, S J; Brand, A R (2000) Effects of long-term physical disturbance by commercial scallop fishing on subtidal epifaunal assemblages and habitats. *Marine Biology 137*: 325–337.

Warnken, K W; Gilla, G A; Dellapennaa, T M; Lehmana, R D; Harperb, D E; Allison, M A (2003) The effects of shrimp trawling on sediment oxygen consumption and the fluxes of trace metals and nutrients from estuarine sediments. *Estuarine, Coastal and Shelf Science 57*: 25–42.

Watling, L; Norse, E A (1998) Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. *Conservation Biology 12*: 1180–1197.

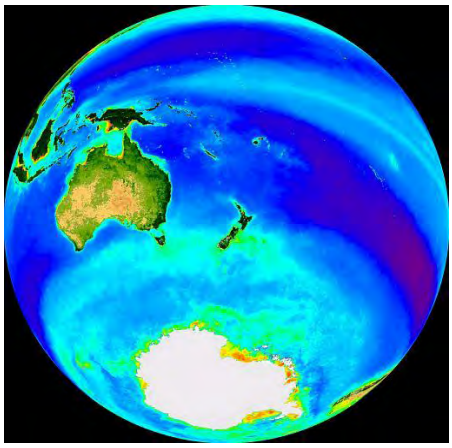
Williams, A; Dowdney, J; Smith, A D M; Hobday, A J; Fuller, M (2011) Evaluating impacts of fishing on benthic habitats: a risk assessment framework applied to Australian fisheries. *Fisheries Research 112*: 154–167.

Williams, A; Schlacher, T A; Rowden, A A; Althaus, F; Clark, M R; Bowden, D A; Stewart, R; Bax, N J; Consalvey, M; Kloser, R J (2010) Seamount megabenthic assemblages fail to recover from trawling impacts. *Marine Ecology 31 (supp.1)*: 183–199.

THEME 4: ECOSYSTEM EFFECTS

AEBAR 2018: *Snapshot* of Chapter 12- Oceans

Ocean Variability and Change



Trends, extremes, decadal cycles

1. THE ISSUE

Ultimately, the ongoing productivity of fisheries is dependent a productive ecosystem and conditions that support biological productivity. Climate and oceanographic variability and long-term environmental changes are therefore of key relevance to fisheries productivity and the carrying capacity of the broader marine environment. Setting the scene essentially provides context.

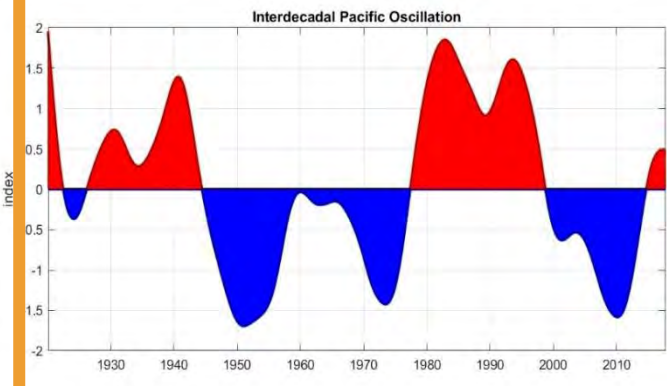
Understanding the trends and cycles observed in the ocean allows improved understanding of the links between observed fisheries patterns and drivers of biological processes. It also allows for the exploration of likely future scenarios for New Zealand fisheries as climate change and global warming continue to progress.

The cumulative effects of ocean climate change and other anthropogenic stressors on aquatic ecosystems (productivity, structure and function) are likely to be high, and increasingly seen in the next 20–30 years.

2. New Zealand seas

- New Zealand straddles 25° of latitude and 30° of longitude in the SW Pacific Ocean from the tropics to the Southern Ocean, with a large area, ~6 million km^2, of jurisdiction.
- Essentially the direction of flow is from west to east across the Tasman Sea that has a relatively slow circulation, and then around northern and southern New Zealand to the more dynamic eastern side, bordering the Pacific Ocean.
- Primary productivity is higher than most of Australasia, but lower than coastal upwelling systems around the rest of the world.
- The greatest productivity is across the Chatham Rise associated with the Subtropical Front and mixing of water masses either side of the front.
- Some long-term trends in the marine environment available at a national scale are incorporated in the Environmental Reporting system developed by MfE and Statistics NZ.

3. Inter-decadal cycles since 1900



Positive and negative phases of the Interdecadal Pacific Oscillation index bring different ocean conditions to New Zealand waters. Under positive phases (red), La Nina conditions tend to prevail, and reduced westerly flow results in less upwelling on the west coast, and warmer air and sea temperatures.

Under negative IPO conditions, El Nino conditions tend to prevail with increased westerly winds and higher upwelling. The abundance of some fishstocks reflects these cycles.

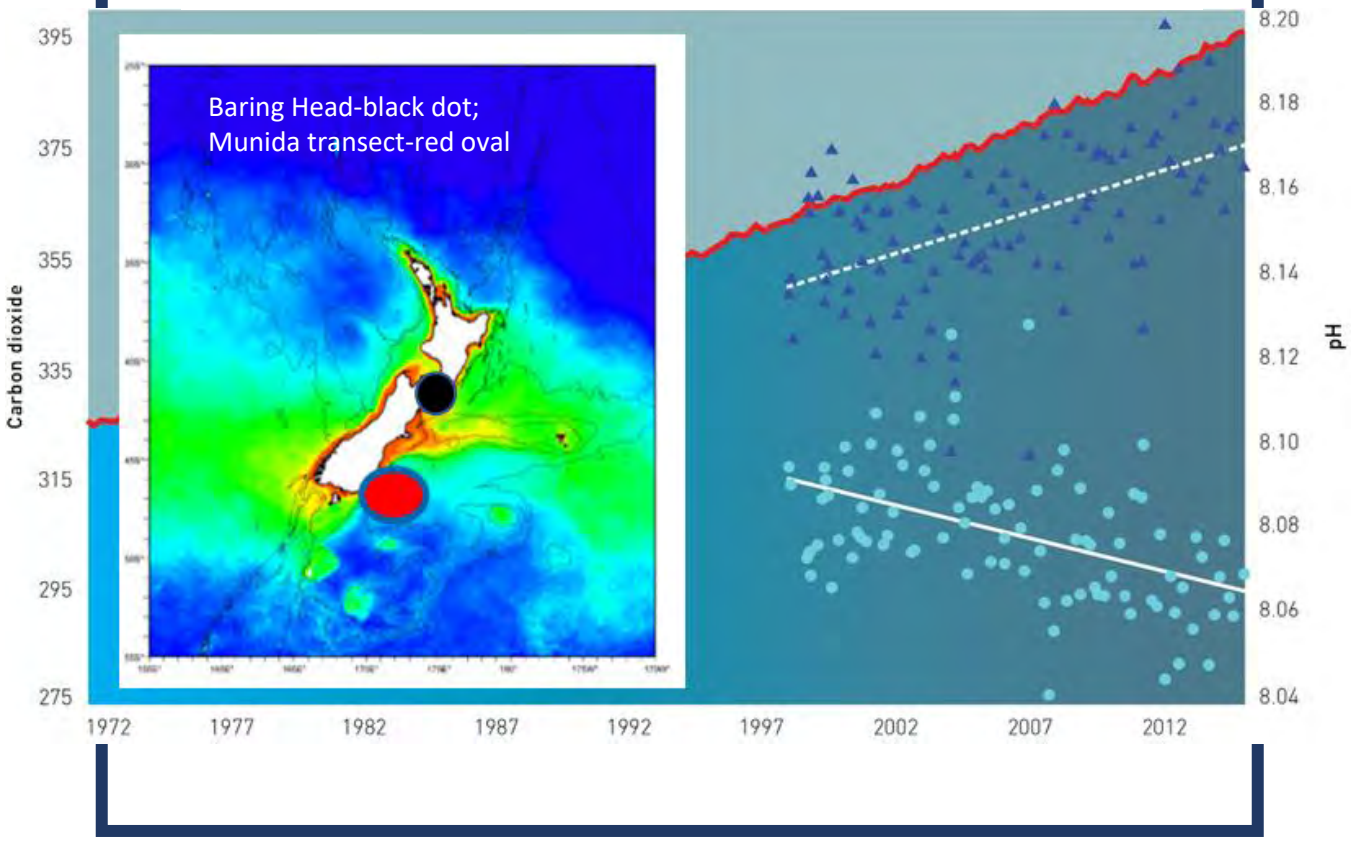
4. New Zealand's seas are changing

Like the rest of the world, our ocean is showing measurable effects of climate change and global warming. Sea temperatures are increasing, ocean acidification is increasing, storm frequencies are higher and more intense and the knock-on effects to fish and biodiversity are evident in some areas. Extreme events such as marine heatwaves are likely to occur more often.

5. CARBON DIOXIDE AND OCEAN ACIDIFICATION. Increased levels of atmospheric CO_2, sea-surface pCO_2 and pH in New Zealand open waters are following the same trends reported in the northern hemisphere.

There are a large number of international studies that have identified a range of direct and indirect effects of ocean acidification across a broad range of marine phylogenetic groups from bacteria to fish, and a number of ecosystems from coastal to deep sea. Of particular concern in New Zealand is the potential effects on shellfish fisheries, aquaculture and deep sea corals.

6. Time series of atmospheric CO_2 at Baring Head (red line), and Munida surface water pCO_2 (dark blue triangles) and pH (light blue) circles)



7. Looking ahead with modelling

- The highest projected regional warming occurs in the East Australian Current, and in subantarctic waters south of Chatham Rise. As the present warming rate in in the south-west Tasman Sea is similar to that projected for waters around NZ, this region may provide an analogue for future changes in NZ waters.
- The depth of the surface mixed layer will decrease across much of the NZ area, except for some subantarctic water regions. The resulting increase in light exposure for plankton may be beneficial to productivity and food webs in subantarctic waters where nutrients are plentiful, but may be deleterious in warmer subtropical waters that are oligotrophic (i.e. low nutrient).
- Model projections for food supply to fisheries on the Chatham Rise are conflicting and need to be resolved. The Chatham Rise is identified as one of the more vulnerable areas in New Zealand, yet it may be one of the areas least affected by OA changes.

12 NEW ZEALAND'S CLIMATE AND OCEANIC SETTING

Status of chapter	This chapter has been fully updated for AEBAR 2018.
Scope of chapter	This chapter provides context within which to consider interactions between the environment and the seafood sector. It provides an overview of primary productivity, oceanography, bentho-pelagic coupling, ocean acidification and oceanic climate trends in the Southwest Pacific region.
Area	New Zealand regional setting.
Focal localities	All New Zealand waters.
Key issues	• Climate and oceanographic variability and long-term changes are of relevance to fisheries and the broader marine environment. • Allows improved understanding of the links between observed patterns and drivers of biological processes. • Allows for testing of likely future scenarios. • New Zealand trends of increasing air and sea temperatures and ocean acidification are consistent with global trends.
Emerging issues	• New Zealand's oceanic climate is changing. • Causal mechanisms that link the dynamics of a variable marine environment to variations in biological productivity, particularly of fisheries and biodiversity, are not well understood in New Zealand or internationally, but are the subject of multiple studies. • Cumulative effects of ocean climate change and other anthropogenic stressors on aquatic ecosystems (productivity, structure and function) are likely to be high, and seen in the next 20–30 years. • Some long-term trends in the marine environment available at a national scale are incorporated in the new Environmental Reporting system being developed by MfE and Statistics New Zealand. • There is a growing recognition that stressors will act both individually and interactively, confounding predictions of the net effects of climate change. • Improved scenario setting and the need for risk evaluation. • The first regime shift in IPO since most fisheries monitoring began occurred in 2000, which is likely to result in fewer El Niño events for a 20–30 year period, which in turn is likely to impact fish productivity. • 2018 saw a very strong marine heat wave (MHW) in the Tasman Sea. MHWs would be expected to become more frequent in a warming world.
Fisheries New Zealand research (current)	ZBD2013-02 *VME connectivity*; ZBD2014-01 *Live DW coral experiment phase 2*; ZBD2013-08 *NZ-Ross sea connectivity Humpback whales*; ZBD2014-10 *BPA biodiversity*; ZBD2014-03 *Sublethal effects of environment change on fish populations*; ZBD2014-04 *Isoscapes for trophic studies*; ZBD2014-05 *Ocean acidification*; ZBD2014-06 *Macroalgae mapping and potential as national scale indicators*; ZBD2014-07 *Southern coralline algae shellfish habitat*; ZBD2014-09 *Climate change risks and opportunities*; ZBD2013-03 *Continuous plankton recorder (16849)*; ZBD2013-06 *Shell generation*; ZBD2012-03 *CRise benthos (16589).*
Government and other research	NIWA Coast & Oceans Centre; NIWA Climate and Atmosphere Centre; University of Otago-NIWA shelf carbonate geochemistry & bryozoans; Munida time-series transect; Physical Oceanography research; Geomarine Services-foraminiferal record of human impact; Regional Council monitoring programmes; Statistics New Zealand Environmental Domain review; Department of Conservation the impacts of climate change on marine protected species. Relevant global climate programmes: Argo; Southern Ocean Observing System. Dragonfly science: Fast-forward fish: resilience of exploited marine populations to a changing ocean.

Related chapters/issues	This chapter provides background environmental information relevant to all chapters, but particularly to the chapters: Biodiversity; Trophic and ecosystem-level effects.

Note: This chapter has been fully revised and updated for 2018 AEBAR.

12.1 CONTEXT

This chapter summarises information on oceanography and ocean acidification around New Zealand, and climate trends in the Southwest Pacific region. This information provides context to understand the interactions between the environment and seafood productivity in the region. Climate and oceanographic conditions play an important role in driving the productivity of our oceans and the abundance and distribution of our fishstocks and fisheries. The most recent analysis of trends in climate and oceanographic variables relevant to fisheries management in New Zealand is given in Hurst et al. (2012).

New Zealand is part of a large submerged continent (Figure 12.1). The Territorial Sea (TS, extending from mean low water shore line to 12 nautical miles), Exclusive Economic Zone (EEZ, extending from 12 nautical miles to 200 miles offshore), and the extended continental shelf (ECS) combine to produce one of the largest areas of marine jurisdiction in the world, an area of almost 6 million square kilometres (Figure 12.1). New Zealand waters straddle more than 25 degrees of latitude from warm, salty subtropical waters at 30°S to cooler, fresher subantarctic waters at 56°S, and 30 degrees of longitude from 161°E in the Tasman Sea to 171°W in the west Pacific Ocean. New Zealand's coastline, with its numerous embayments, is long, with estimates ranging from 15 000 to 18 000 km, depending on the measurement method (Gordon et al. 2010).

New Zealand lies across an active subduction zone in the western Pacific plate; tectonic activity and volcanism have resulted in diverse and varied seascapes within the EEZ. The undersea topography comprises a relatively narrow band of continental shelf down to 200 m water depth, extensive continental slope areas from 200 to 1000 m, extensive abyssal plains, submarine canyons and deep-sea trenches, ridge systems and numerous seamounts and other underwater topographic features such as hills and knolls. There are three significant submarine plateaus, Challenger Plateau, Campbell Plateau, and Chatham Rise.

The physical oceanography of the deep seas around New Zealand has recently been reviewed by Chiswell et al. (2015). Measurements from platforms including satellites, drifting and profiling floats, moorings and oceanographic voyages have provided a wealth of new observations over the last 30 years and analysis of these observations has substantially improved our understanding of the oceanography. Chiswell et al. (2015) summarise and integrate earlier research through a series of schematics of the ocean currents around New Zealand. Their surface currents are shown in Figure 12.2.

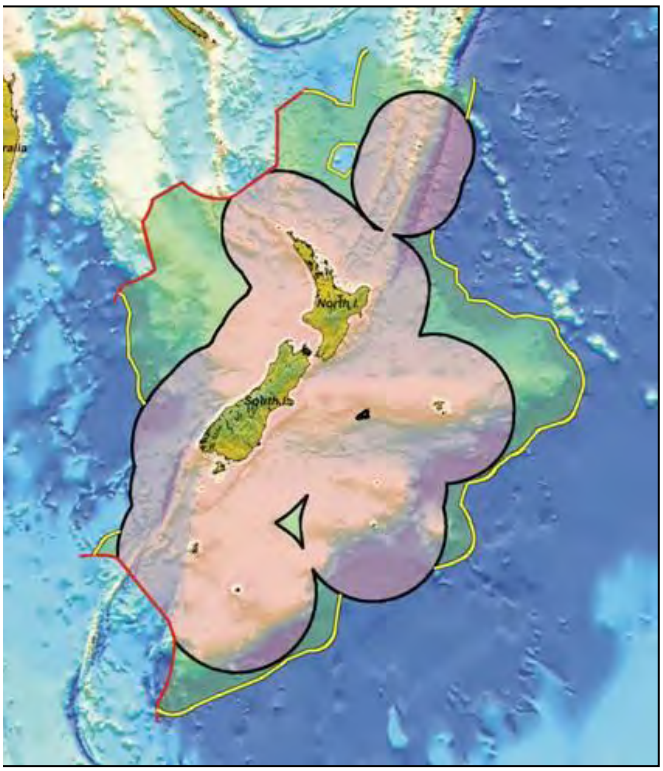


Figure 12.1: New Zealand land mass area 250 000 km²; EEZ and territorial sea area (pink) 4 200 000 km²; extended continental shelf extension area (light green) 1 700 000 km²; Total area of marine jurisdiction 5 900 000 km². The black line shows the boundary of the New Zealand EEZ, the yellow line indicates the extension to New Zealand's legal continental shelf, and the red line the agreed Australia/New Zealand boundary under UNCLOS Article 76. Image courtesy of GNS.

The Tasman Sea, west of New Zealand, is isolated from the South Pacific Gyre by the New Zealand landmass. The South Pacific Western Boundary Current, the East Australian Current (EAC) flows down the east coast of Australia, before separating from the Australian land mass at about 31-32°S (Ridgway & Dunn 2003). Part of the separated flow crosses the Tasman Sea as the Tasman Front (Stanton 1981,

Ridgway & Dunn 2003, Sutton & Bowen 2014) while the remaining flow continues south in the EAC extension. The Tasman Front then feeds the western boundary currents adjacent to New Zealand- the East Auckland Current (EAUC) between North Cape and East Cape and the East Cape Current (ECC) between East Cape and the northern side of Chatham Rise. Sutton & Bowen (2014) found that the Tasman Front is a weaker connection than previously thought between the EAC and EAUC, with the Tasman Front being shallower and transporting less water than the EAC and EAUC.

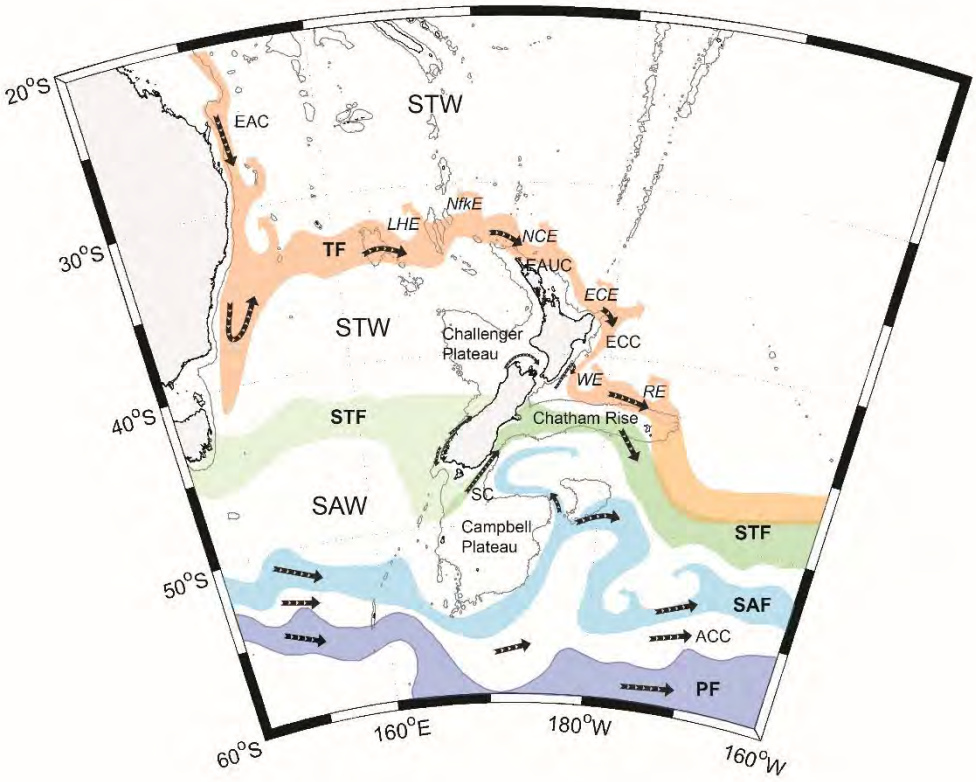


Figure 12.2: Schematic surface circulation around New Zealand based on drifter and hydrographic data. Regions of flow are shown as coloured streams. Colours reflect the temperature of the flows with red being warmest, and dark blue being coldest. The STF in the Tasman Sea is density compensated with little flow, as indicated by the shading. Water Masses are Subtropical Water (STW), Tasman Sea Central Water (TSCW), Subantarctic Water (SAW) and Antarctic Surface Water (AASW). Ocean fronts are Tasman Front (TF), Subtropical Front (STF), Subantarctic Front (SAF) and Polar Front (PF). Ocean currents are East Australia Current (EAC), East Australia Current extension (EACx), East Auckland Current (EAUC), East Cape Current (ECC), d'Urville Current (dUC), Wairarapa Coastal Current (WCC), Westland Current (WC), Southland Current (SC) and Antarctic Circumpolar Current (ACC). Eddies are Lord Howe Eddy (LHE), Norfolk Eddy (NfkE), North Cape Eddy (NCE), East Cape Eddy (ECE), Wairarapa Eddy (WE) and Rekohu Eddy (RE). Adapted from Chiswell et al. (2015).

At the southern limit of the Tasman Sea is the Subtropical Front, which passes south of Tasmania and approaches New Zealand at the latitude of Fiordland (Stanton & Ridgway 1988, Hamilton 2006). Around 165°E the front diverts south across Macquarie Ridge where it has two clear branches (Smith et al. 2013), which continue onto Campbell Plateau where they merge to form the Southland Front along the Otago Coast (Chiswell 1996, Sutton 2003).

The circulation in the central Tasman Sea, east of the influence of the EAC, and between the Tasman Front and Subtropical Front is thought to be relatively slow. Ridgway & Dunn (2003) showed eastward surface flow across the interior of the Tasman Sea sourced from the southernmost limit of the EAC, with the flow separating around Challenger Plateau and, ultimately, New Zealand. Reid's (1986) analysis indicates that a small anticlockwise gyre exists in the western Tasman Sea at 1000–2500 m depth. This gyre is centred at about 35°S, 155°E on the offshore side of the EAC and west of Challenger Plateau. All indications are that

the eastern Tasman region overlying Challenger Plateau is not very energetic.

In contrast, the east coast of both islands and Cook Strait

are highly energetic. Along the north-east coast of the North Island there are two semi-permanent eddies that vary in size and strength, the North Cape and East Cape Eddies (Roemmich & Sutton 1998). The inshore sides of these eddies comprise the EAUC, which flows down the east coast of the North Island to East Cape. Most of the EAUC water continues south in the ECC, with the remainder being split between the East Cape Eddy and the Pacific Ocean. There are several eddies in the East Cape Current region, with the largest known as the Wairarapa Eddy. It sits between the North Island and the northern flanks of Chatham Rise (Chiswell et al. 2015).

Along the south-eastern coast of the South Island there is a narrow band of warm, salty subtropical water on the shelf separated from offshore cold, fresh subantarctic water by the Southland Front (the local manifestation of the Subtropical Front). This front has an associated flow of mainly subantarctic water, called the Southland Current (Sutton 2003) which flows north to the southern flank of Chatham Rise. The Southland Current then turns east along Chatham Rise before turning south at the Chatham Islands and then east into the Pacific Ocean contributing to the South Pacific Current (Figure 12.2, Stramma et al. 1995).

Forcén-Vázquez (2015) showed that water from the Subtropical Front is found over a large fraction of Campbell Plateau and varies significantly from year to year. They also found that water from the Southern Ocean is mixed onto the plateau, giving it an oceanography distinct from the surrounding seas. These waters are well mixed and are known to be iron limited (Boyd et al. 1999). It has been suggested that there are high transfer efficiencies between low and high trophic levels (Bradford-Grieve et al. 2003). Steering of currents along and around plateaus and ridges gives rise to higher ocean productivity than might be expected in the generally oligotrophic western Pacific Ocean (Figure 12.3).

New Zealand net primary productivity levels are high compared with most of Australasia, but lower than most coastal upwelling systems around the world (Field et al. 1998). Chatham Rise has the highest productivity levels in the region, associated with the Subtropical Front and mixing between the different water masses across the front (Figure 12.3). Macronutrient-limited subtropical waters (Bradford-Grieve et al. 1997) mix with high nutrient, low-chlorophyll (NHLC) iron-limited subantarctic waters (Boyd et al. 1999) in the Subtropical Front, effectively removing both the iron and macronutrient restraints. The increased productivity resulting from the mixing is apparent in the high surface chlorophyll a detected with ocean colour remote sensing (Figure 12.3 and Figure 12.4 left panel).

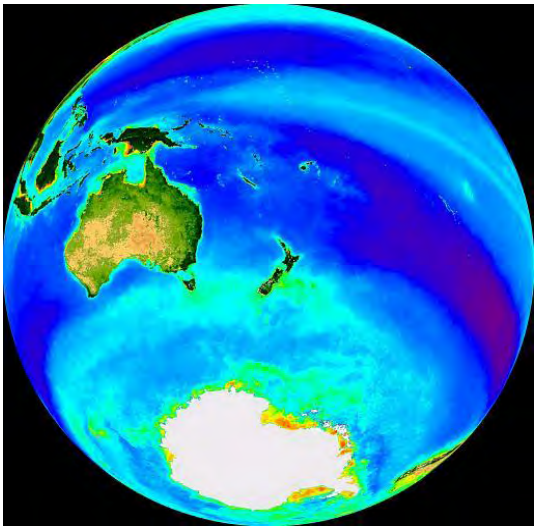


Figure 12.3: SeaWIFS image showing elevated chlorophyll a (green) near New Zealand. Image courtesy of NOAA.

Around the coast turbulence and upwelling play key roles in driving primary productivity (Figure 12.3; Pinkerton et al. 2005). However, care is needed in interpreting remotely-sensed ocean colour in coastal waters as algorithms cannot readily distinguish between primary productivity (from phytoplankton) and sediments in freshwater runoff. Thus, interpretation of coastal relative productivity levels has to be made in conjunction with knowledge of river flow.

High productivity in coastal waters and on ocean plateaus support a range of commercial shellfish and finfish fisheries from the shoreline to depths of about 1500 m. Seamounts, seamount chains and ridge structures can also provide additional localised areas of upwelling and increased productivity sometimes associated with commercial fisheries.

Patterns in surface waters of primary productivity are mirrored to an extent in the amount of 'energy' that sinks to the seafloor (Figure 12.4 right panel). This particulate organic carbon (POC) flux is based on a model which accounts for sinking rates of dead organisms and predation in the water column (Lutz et al. 2007). The POC flux serves

as a potential surrogate for benthic production and indicates where bentho-pelagic coupling may be strong. The highest levels of POC flux coincide to a large extent with surface productivity, with coastal waters (including around the offshore islands) and Chatham Rise having the highest estimated productions (Figure 12.4 right panel).

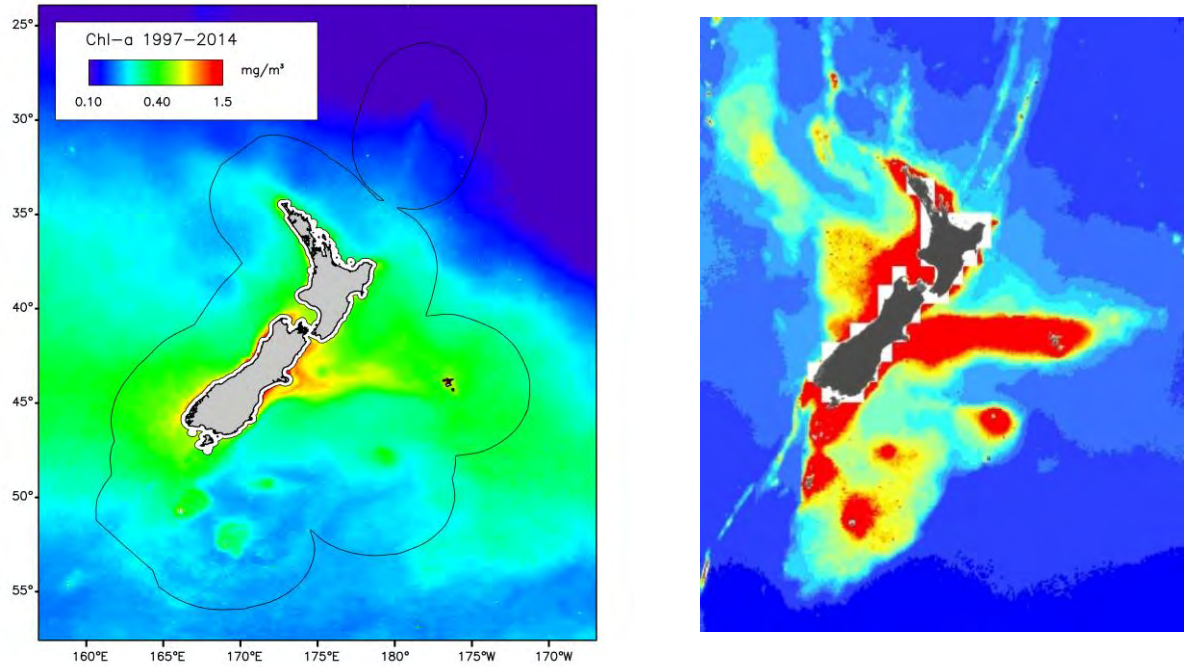


Figure 12.4: [Left panel] Mean annual concentration of chlorophyll-a (the ubiquitous phytoplankton pigment) from satellite ocean colour observations. The image is based on the merged dataset of SeaWiFS and MODIS-Aqua, which covers the period 1997–2014. The boundary of the New Zealand EEZ is also shown. Data in the territorial sea (12 nautical miles from the coast) is excluded because of possible contamination by suspended sediment and river run-off. [Right panel] The relative concentrations of particulate organic carbon (POC) that reach the seafloor. Red shows the highest levels, which are likely to be associated with areas of enhanced benthic productivity (based on the model of Lutz et al. 2007). Images courtesy of NIWA.

12.2 INDICATORS AND TRENDS

12.2.1 SEA TEMPERATURE

Sea surface temperature (SST), sea surface height (SSH), air temperature and ocean temperature to 1000 m depth, are all somewhat correlated over seasonal and inter-annual time scales (Hurst et al. 2012). New Zealand air temperatures have increased by about 1°C since 1900 (Figure 12.5).

Although a linear trend has been fitted to the seven-station temperatures in Figure 12.5, the temperature changes are not uniform over time. For example, marked warming occurred through the periods 1940–60 and 1993–2000. Higher frequency variations can be related to fluctuations in the prevailing north-south airflow across New Zealand (Mullan et al. 2010). Temperatures are higher in years with stronger or more prevalent northerly winds and are lower in years with stronger or more prevalent southerly winds. This is as expected, since southerly winds transport cool air from over the Southern Oceans to New Zealand.

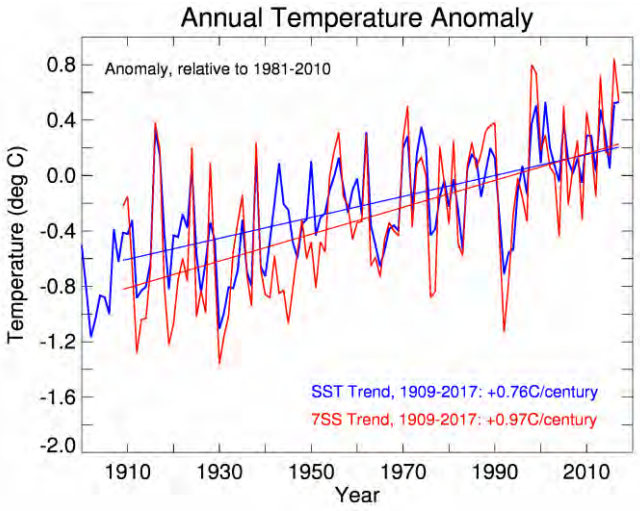


Figure 12.5: Annual time series in New Zealand. NOAA annual mean sea surface temperatures (blue line) averaged over the box outlined in black in Figure 12.6 and NIWA's seven-station annual mean air temperature composite series (red line), expressed as anomalies relative to the 1981–2010 climatological average. Linear trends over the period 1909–2017, in °C/century, are noted under the graph. (Image source: updated from Mullan et al. 2010. NOAA data taken from ERSSTv5 (Huang et al. 2017)).

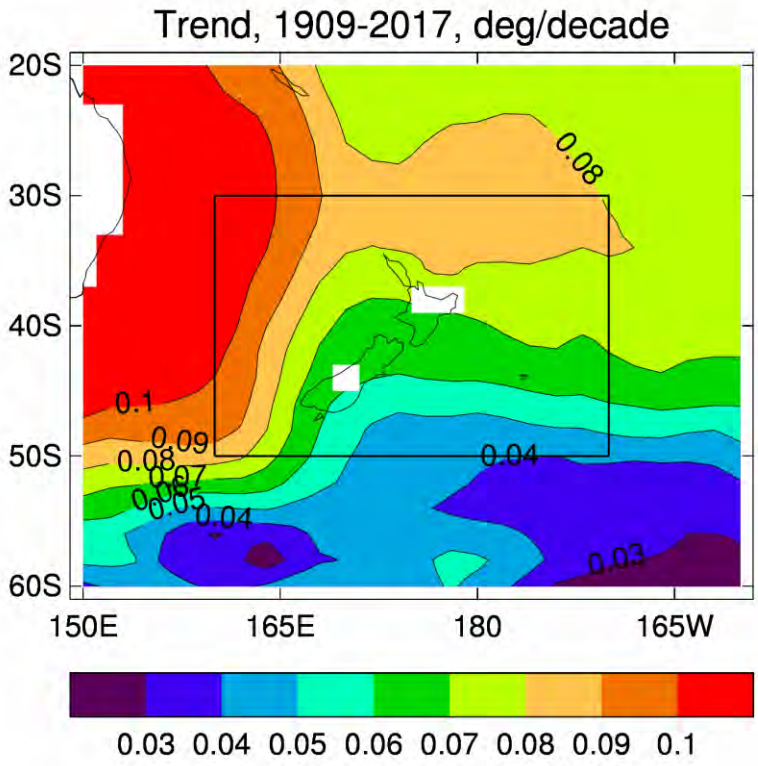


Figure 12.6: Trends in sea surface temperature, in °C/decade over the period 1909–2017, calculated from the NOAA_ERSST_v5 data-set (provided by NOAA's ESRL Physical Sciences Division, Boulder, Colorado, USA, from their web site at http://www.esrl.noaa.gov/psd). The data values are on a 2° latitude-longitude grid. (Image source: updated from Mullan et al. 2010).

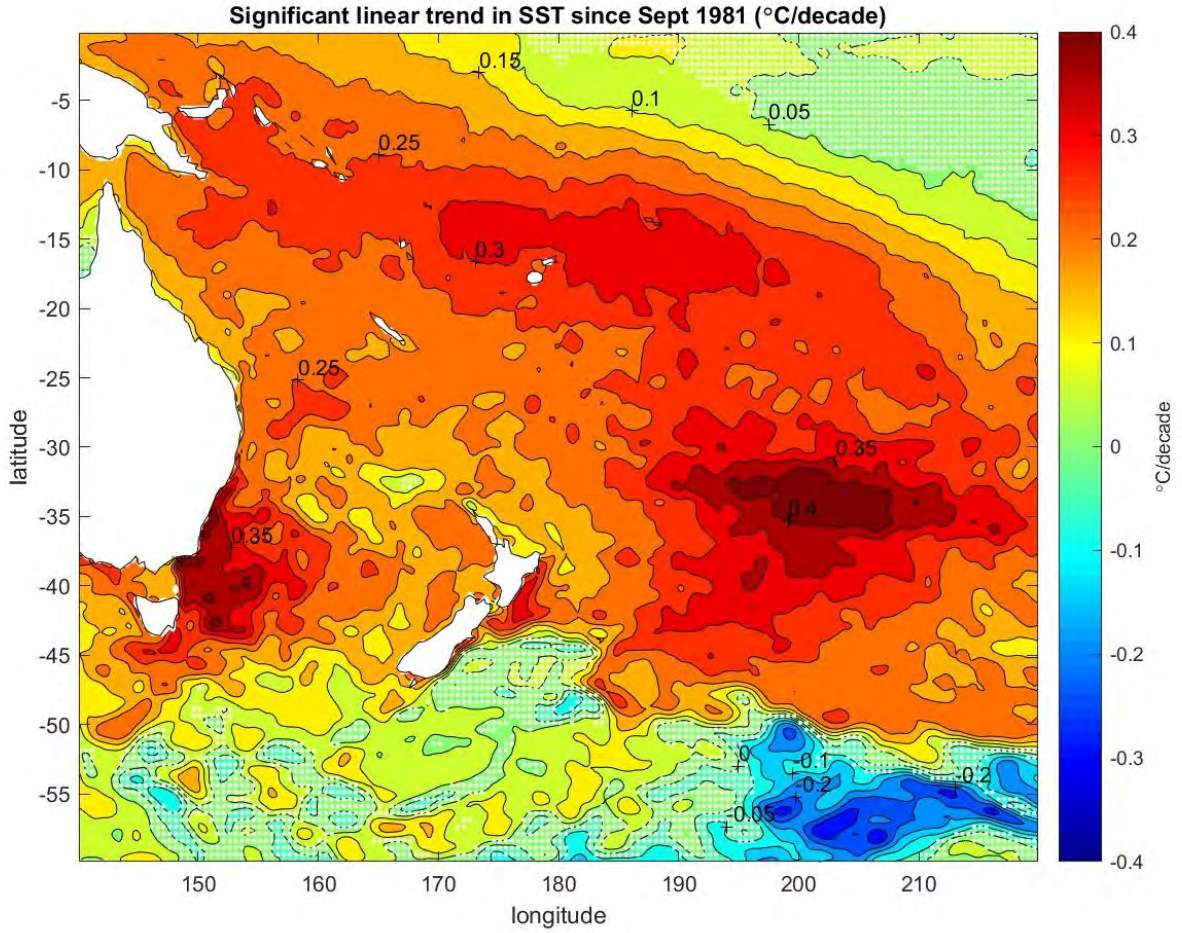


Figure 12.7: Trends in sea surface temperature, in °C/decade over the period 1982–2018. The data are from NOAA based on daily interpolated satellite measurements over a 0.25° grid (http://www.ncdc.noaa.gov/oisst, Reynolds et al. 2007). Areas where the trend is not statistically significant are shaded in white. Modified from Sutton & Bowen 2019.

The unusually steep warming in the 1940–60 period is paralleled by an unusually large increase in northerly winds during this same period (Mullan et al. 2010). On a longer timeframe, there has been a trend towards less northerly (more southerly) winds since about 1960 (Mullan et al. 2010). However, New Zealand temperatures have continued to increase over this time, albeit at a reduced rate compared with earlier in the twentieth century. This is consistent with a warming of the entire Southwest Pacific (Mullan et al. 2010).

Trends in sea surface temperature (SST) in the New Zealand region tend to be slightly smaller than trends in air temperature over land (Figure 12.5). Mullan et al. (2010) describe the pattern of warming in New Zealand as consistent with changes in sea surface temperature and prevailing winds. Their review shows enhanced rates of warming along the East Australian coast and to the east of the North Island, and much lower rates of warming south and east of the South Island (Figure 12.6).

Figure 12.7 shows SST trends since 1982 calculated from daily satellite measurements. These are at higher spatial resolution than Figure 12.6, providing more detail but spanning a shorter period. It is apparent that SSTs are increasing north of about 45°S while they are increasing more slowly, and decreasing in recent decades, east of Otago and south of New Zealand. This regional pattern of cooling (or only slow warming) to the south, and strong warming in the Tasman and western Pacific can be related to increasing westerly winds and their effect on ocean circulation (Mullan et al. 2010, Roemmich et al. 2007, Roemmich et al. 2016). Thompson & Solomon (2002) discuss the increase in Southern Hemisphere westerlies and the relationship to global warming; Roemmich et al. (2007) and Roemmich et al. (2016) describe recent ocean circulation changes with the South Pacific subtropical gyre

spinning up in response to the changing winds and Thompson et al. (2009) discuss the consequent effect on sea surface temperatures in the Tasman Sea.

Coastal SST data, particularly long time series from University of Auckland measurements at Leigh and University of Otago measurements at Portobello, have been used to link changes in the near shore environment with larger-scale climate signals. SST anomalies at Portobello since 1953 and Leigh since 1967 suggest a strong relationship between SST and the southern oscillation index (SOI) (Figure 12.8

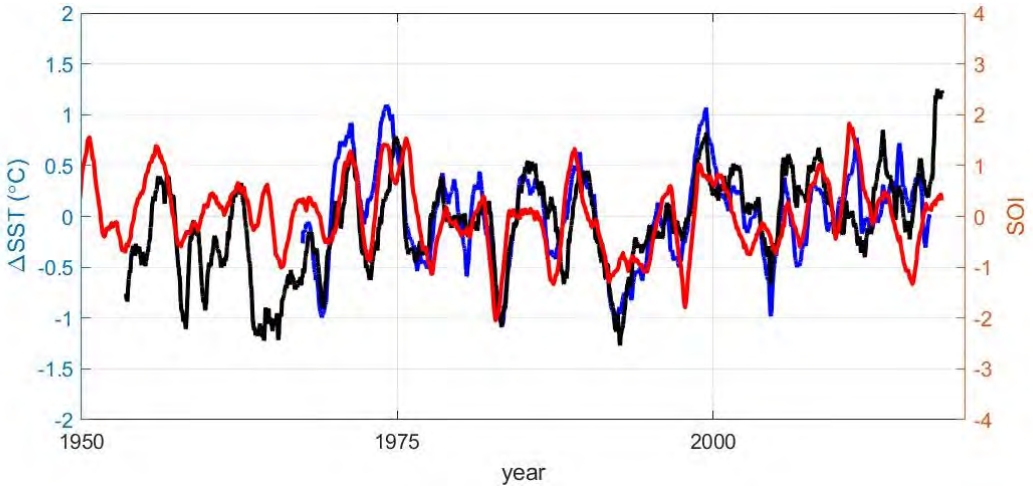


Figure 12.8: Sea surface temperature (SST) anomalies from a 40-year climatology from 1971–2010 of observations at Leigh (blue line), Portobello (black line) and the Southern Oscillation Index (SOI; red). SSTs are de-seasoned and all three time series are smoothed with a one year running mean.

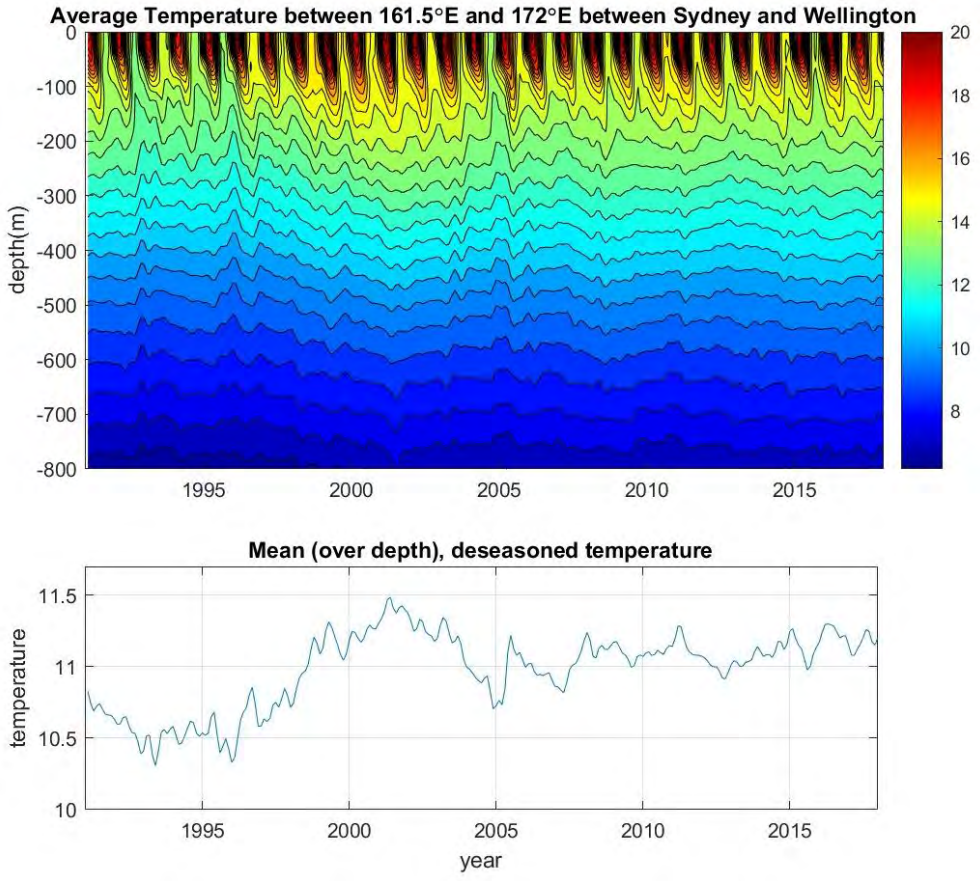


Figure 12.9: Eastern Tasman ocean temperature: along Wellington to Sydney transect: 161.5°E to 172°E, 1991–2018. Coloured scale to the right is temperature (°C). Image updated from Sutton et al. 2005. Bottom panel shows the mean 0-800m temperature with seasons removed.

Temperature fluctuations also occur at depth in the ocean as demonstrated by changes in temperature down to 800 m in the eastern Tasman Sea between 1992 and late 2014.

The ocean temperature between Sydney and Wellington has been sampled about four times per year since 1991 by CSIRO and Scripps Institution of Oceanography (http://www-hrx.ucsd.edu/). Analyses of the subsurface temperature field using these data include Sutton & Roemmich (2001) and Sutton et al. (2005). The temperature as a function of depth and time for the eastern portion (between 161.5°E and 172°E) of this section is shown in Figure 12.9. This eastern Tasman section is close to New Zealand, and has low oceanographic variability meaning that subtle inter-annual changes can be seen. The portion of the transect shown is along a fairly constant latitude and is therefore unaffected by latitudinal temperature and seasonal cycle variation. The lower panel shows the de-seasoned temperature averaged along the transect between the surface and 800 m and from 1991 to the most recent sampling.

The seasonal cycle is clearly visible in the upper 100–150 m. There is a marked warming signal that occurred through the late 1990s, apparent from the isotherms increasing in depth through that time period. This warming was significant in that it extended through the full 800 m of the measurements (effectively the full depth of the eastern Tasman Sea). It also began during an El Niño period when conditions would be expected to be relatively cool. It was thought to be linked to a large-scale warming event centred on 40°S that had hemispheric and perhaps global implications. This warming has been discussed by Sutton et al. (2005) who examined the local signals, Bowen et al. (2006) who studied the propagation of the signal into the New Zealand area, and Roemmich et al. (2007), who examined the broad-scale signal over the entire South Pacific Ocean. Roemmich et al. (2007) hypothesised that the ultimate forcing was an increase in high latitude westerly winds effectively speeding up the entire South Pacific gyre. An update of this analysis has confirmed that the gyre spin-up continued through to 2015 (Roemmich et al. 2016).

Other phenomena have led to periods of warming that are not as yet fully understood. Both stochastic environmental variability and predictable cycles of change influence the productivity and distribution of marine biota in our region.

The summer of 2017/2018 saw very warm SSTs in the Tasman Sea, with SST anomalies reaching 4–5°C above normal. Studies of this event indicate that this anomalous warm event was the result of a prolonged period of calm winds, in turn as a result of a blocking atmospheric high pressure to the east of New Zealand (Salinger at al. 2018; Behrens et al. 2018). Figure 12.10c shows the average vertical temperature anomaly in the eastern Tasman Sea through this time period as measured by Argo floats. This shows an approximately 30 m deep layer of warm water formed in November 2017 and persisting until February 2018 before it was eroded at the surface by weather events. There was no significant change in the total heat content through this period because the anomaly was relatively thin, and there was no discernible air-sea flux signature (Behrens et al. 2018). Longer-term occurrences of anomalously warm and cold events are shown in Figure 12.10 a) and b). Regional average SST anomaly time series are shown for three areas, along with the annually-smoothed time series and associated linear trends (Sutton & Bowen 2019). Time periods where the daily anomalies are more than two standard deviations from the mean are shown in bold. In all regions, the occurrences of warm and cool events are modulated by interannual variability, that is warm extreme events almost exclusively occur during warm periods and cool extreme events during cool periods, consistent with the results of Behrens et al. 2018. This modulation is clear during the early 1990s cool period in subtropical water (the Tasman and northeast regions), particularly so for the northeast, with the bulk of northeast cool events occurring between 1991 and 1995. Beyond this decadal modulation, there is no clear trend in the occurrence of warm or cool events in the northeast or southeast regions. There is a suggestion that the occurrence of warm events in the Tasman region is becoming more frequent, and cool events look to be becoming rarer, with only one Tasman cool event since 2008, as could be expected given the significant warming trend in the eastern Tasman (Figure 12.7).

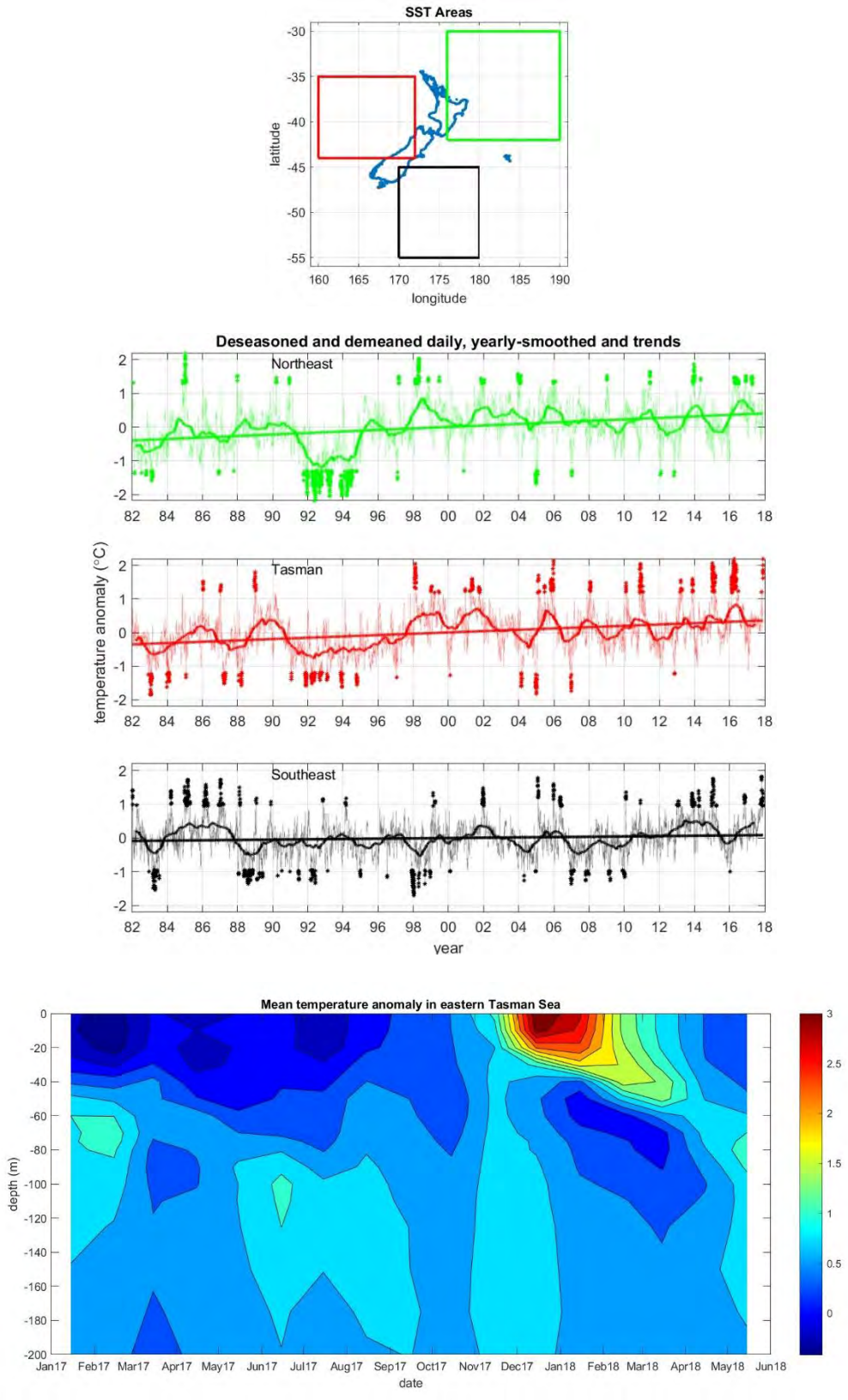


Figure 12.10: Marine heat waves. The top panel shows three regions chosen to represent different regimes. The middle panel shows the mean de-seasoned satellite SST values for these regions, at daily and annually-smoothed resolutions. The linear trends are also shown. Values more than two standard deviations from the mean are shown in bold. (Modified from Sutton & Bowen 2019). The bottom panel shows the mean temperature anomaly in the eastern Tasman Sea through the summer of 2017–2018. (Modified from Salinger et al. 2018)

12.2.2 CLIMATE VARIABLES

The Interdecadal Pacific Oscillation (IPO) is a Pacific-wide reorganisation of the heat content of the upper ocean and represents large-scale, decadal temperature variability with changes in phase over 15–30-year time scales. In the past 100 years, phase changes occurred in 1925, 1947, 1977 and about 2000 (Figure 12.11). The latest shift should result in New Zealand experiencing a period of reduced westerlies, with associated warmer air and sea temperatures and reduced upwelling on western coasts (Hurst et al. 2012).

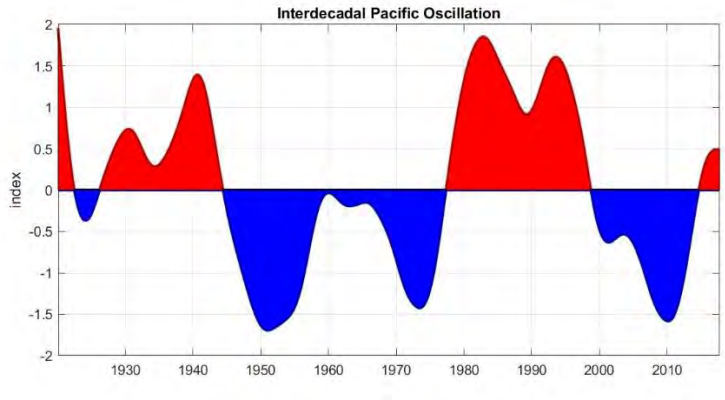


Figure 12.11: Smoothed index of the Interdecadal Pacific Oscillation (IPO) since 1900. (NIWA data).

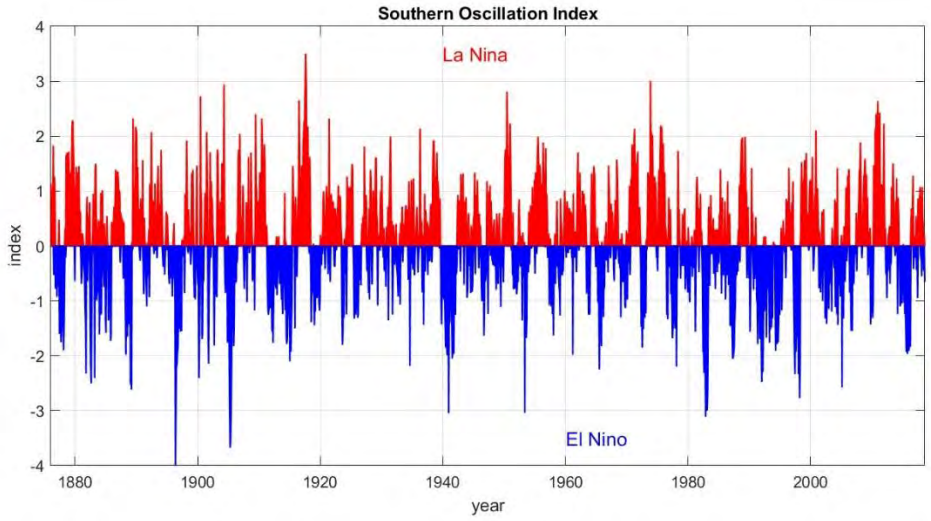


Figure 12.12: Southern Oscillation Index (SOI). Red indicates La Niña and a tendency for warmer temperatures around New Zealand, blue indicates El Niño and a tendency for cooler conditions around New Zealand. (NIWA data).

The El Niño-Southern Oscillation (ENSO) cycle in the tropical Pacific has a strong influence on New Zealand. ENSO is described here by the Southern Oscillation Index (SOI), a measure of the difference in mean sea-level pressure between Tahiti and Darwin. When the SOI is strongly positive (persisting above +1, Figure 12.12), a La Niña event is taking place and New Zealand tends to experience more north-easterlies, reduced westerly winds, milder, more settled, warmer anticyclonic weather and warmer sea temperatures (Hurst et al. 2012). When the SOI is strongly negative (persisting below -1), an El Niño event is taking place and New Zealand tends to experience increased westerly and south-westerly winds, cooler, less settled weather and enhanced along shelf upwelling off the west coast South Island and north-east North Island (Shirtcliffe et al. 1990, Zeldis 2004, Chang & Mullan 2003). The SOI is available monthly from 1876 onwards (Mullan 1995) (Figure 12.12).

12.2.3 WATER CHEMISTRY: OCEAN ACIDIFICATION

The increase in atmospheric carbon dioxide (CO_2) since the industrial revolution has been paralleled by an increase in CO_2 concentrations in the upper ocean (Sabine et al. 2004). The ocean is estimated to hold about 25% of all anthropogenic CO_2 released since 1750 (Le Quéré et al. 2018), with the anthropogenic CO_2 signal apparent to an average depth of about 1000 m and reaching depths of 3000 m in of deep water formation regions (Sabine et al. 2004). The Southern Ocean is the primary route for the entry of anthropogenic CO_2, accounting for about 40% of current oceanic uptake (DeVries 2014)

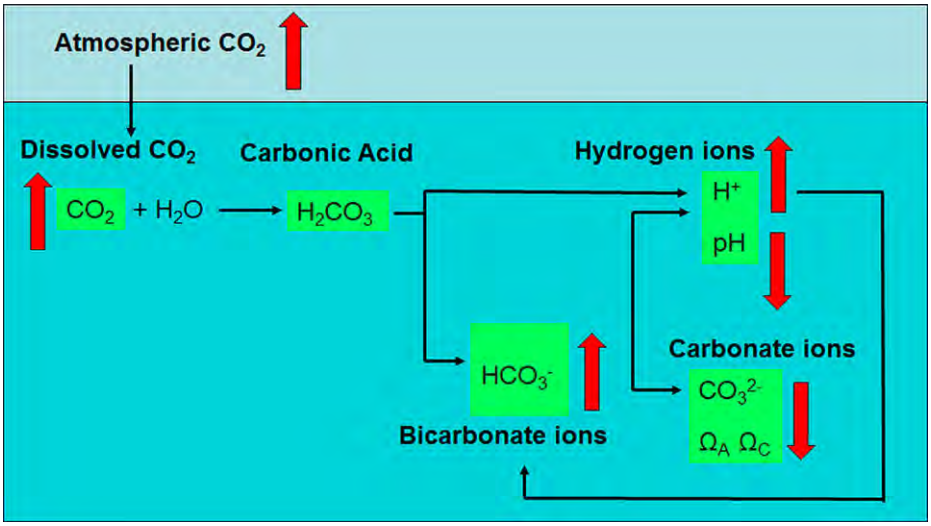


Figure 12.13: Conceptual diagram showing the reactions and chemical species of the marine carbonate system (black arrows), with the direction of the vertical red arrows indicating the net change in response to increasing atmospheric CO_2, and resulting in ocean acidification (Law et al. 2018a).

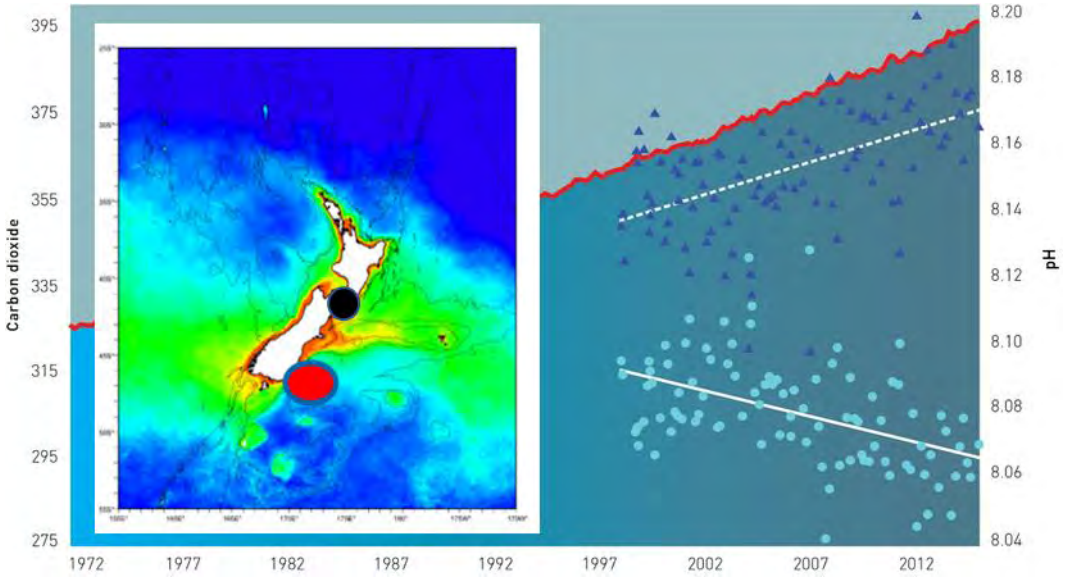


Figure 12.14: Time series of atmospheric CO_2 at Baring Head (location: black dot, Wellington; data: red line, Brailsford et al. 2012), and surface water pCO_2 (data: dark blue triangles) and pH (data: light blue circles) at the subantarctic water station (location: red dot, on the Munida transect off Dunedin) (Law et al. 2018a).

Carbon dioxide absorbed by seawater reacts with the water and carbonate ions to form bicarbonate ions and releases hydrogen ions in the process (Figure 12.13). This reaction raises the acidity and lowers the pH of seawater, in addition to decreasing the amount of carbonate in the seawater. Since the beginning of the industrial era, average surface ocean pH has decreased by 0.1 units, with a further

decrease of up to 0.4 units expected by the end of the century (Rhein et al. 2013). The pH scale is logarithmic, so a 0.4 pH decrease corresponds to a 150% increase in hydrogen ion concentration. Both the predicted pH in 2100 and the rate of change in pH are outside the range experienced by the oceans for at least half a million years.

Recent projections for the open ocean around New Zealand indicate a mean pH decline to 7.935–7.985 by 2050, depending upon the emission scenario. Under a more favourable pathway, RCP4.5, which anticipates zero emissions by mid-century, pH around NZ will fall to a mean of 7.95 by 2100; conversely, using the worst-case scenario of continued emissions, RCP8.5, indicates that ocean pH will have fallen by about 0.33 to about 7.77 by 2100 (Law et al. 2018b). The latter projection is equivalent to an increase in Hydrogen ion concentration (acidity) of 116%. The drop in pH shows little spatial variability in NZ waters as it is primarily driven by atmospheric CO_2.

The Munida Time Series has maintained bimonthly measurements of surface CO_2 and pH at a station in subantarctic waters off the Otago shelf for 20 years and is the longest running ocean acidification time series station in the Southern hemisphere (Law et al. 2018a). The Munida Time Series data show a pH decline in subantarctic surface waters since 1998 (Figure 12.14: Law et al. 2018b), consistent with that expected from equilibrium with atmospheric CO_2, (see Figure 12.14; Law et al. 2018a). Dissolved pCO_2 (partial pressure of CO_2) at the Munida station shows a statistically significant increase of 1.3 µatm yr^{-1} for 1998–2012 (Bates et al. 2014), that is consistent with long-term changes observed at other global time series sites. Surface pH in the open ocean has been determined at seven long-term time series stations, six of which are in the northern hemisphere (Bates et al. 2014; Figure 12.15). All time-series records show long-term trends of increasing pCO_2 and decreasing pH. The significant seasonal variability apparent in the subantarctic waters time series is due to biological uptake of dissolved inorganic carbon and seasonal temperature changes (Brix et al. 2013).

On a global basis, the open ocean shows relatively low spatial and temporal variability in pH relative to coastal waters where pH may vary by up to 1 unit in response to precipitation events and biological activity. A New Zealand coastal ocean acidification observing network (NZOA-ON) has been operating for three years to establish baseline conditions against which to assess future changes in pH. The network maintains pH time series for 15 coastal stations around New Zealand and includes a range of coastal morphotypes and conditions. A variety of stakeholders, including government agencies, councils, industry and citizens, provide alkalinity and DIC samples on a fortnightly basis for analysis at a centralised facility (NIWA/University of Otago Research Centre for Oceanography). Examples of pH variability at three of the NZ-OAON sites are shown in Figure 12.16. The seasonal pH cycle is controlled by the temperature at most sites, with a maximum pH in late winter and minimum in late summer–autumn, as for Wellington Harbour in Figure 12.16. Local factors also influence pH, as at the Chatham Islands where phytoplankton blooms along the Subtropical Front result in periods of elevated pH, and in Jackson Bay (West Coast, South Island) where freshwater input causes sharp pH declines. Freshwater input also supplies excess nutrients and organic matter in some regions, which enhance acidification of coastal waters. An example of this is the Firth of Thames, where seasonal surveys of surface water have shown a pH decline to 7.9 in autumn in the innermost Firth (Zeldis & Swaney 2018; K. Currie, pers. comm.), with subsequent monitoring with SeaFET sensors detecting pH events of less than 7.7 (J. Zeldis, pers. comm.). This pH is lower than that projected for the surface open ocean around New Zealand by the year 2100 (Law et al. 2018b), indicating that some coastal waters already experience periods of intense acidification.

Biological implications of ocean acidification result from increasing hydrogen ion concentration (decreasing pH), decreasing carbonate availability and increasing dissolved pCO_2. There are a large number of international studies that have identified a range of direct and indirect effects of ocean acidification across a broad range of marine phylogenetic groups, from bacteria to whales and a number of ecosystems from coastal to deep sea. A particular concern regarding ocean acidification is that the reduction in carbonate availability may potentially impact organisms that produce shells or body structures of calcium carbonate, resulting in a weakening of shell integrity, redistribution of an organism's metabolic activity and increased physiological stress.

Organisms likely to be affected by ocean acidification include those at the base of the food chain (protozoa, plankton), coralline algae, rhodoliths, shallow and deepwater corals, echinoderms, molluscs, and possibly

cephalopods (e.g., squids) and high-activity pelagic fish (see Feely et al. 2004 and references therein; Orr et al. 2005; Langer et al. 2006). In particular, early life stages are particularly vulnerable in calcifying (carbonate-forming) species across many phyla (e.g., Byrne et al. 2013, Mu et al. 2015, Bylenga et al. 2015). Ocean acidification is a threat to deep-sea habitats such as seamounts, which can support structural reef-like habitat composed of stony corals (Tracey et al. 2011). A shoaling carbonate saturation horizon could push such biogenic structures to the tops of seamounts, or cause widespread die-back (e.g., Thresher et al. 2012). This has important implications for the structure and function of benthic communities. Recent research on NZ cold-water corals identified potential changes in physiology when maintained under lower pH for one year (Gammon et al. 2018). Changes in cold water corals may indirectly impact commercial fisheries for deepwater species such as orange roughy (Clark 1999).

In surface and shallow waters some phytoplankton and sea-grasses may benefit from an increase in dissolved pCO_2 due to increased photosynthesis. Direct effects of acidification on the physiology and development of fish have also been investigated. Adverse effects on physiology development (e.g. Franke & Clemmesen 2011) and behavior modification (Munday et al. 2014) have been documented. Such studies highlight the potential for increasing acidification to impact larval growth and development, with implications for survival and recruitment of both forage fish and fish harvested commercially. However, recent studies showed no adverse effects of low pH on Yellowtail Kingfish (Munday et al. 2015), with warming having a greater impact on their behaviour (Watson et al. 2018).

An assessment of the current knowledge, and vulnerability, of different New Zealand biological groups, species and ecosystems to ocean acidification has been recently published, as summarized in Table 12.1 (Law et al. 2018a). The general findings of the synthesis are that a) calcifying organisms (those with carbonate shells) are vulnerable; b) early-life history stages (eggs and larvae) are more vulnerable than later life history stages, and so are a potential bottleneck to species survival; c) more research is required to determine the response of larger marine fauna (crustacea, fish and higher trophic levels) to ocean acidification (see Table 12.1); the interaction of ocean acidification with other climate stressors and its effect on ecosystem interactions; and the potential for adaptation to ocean acidification.

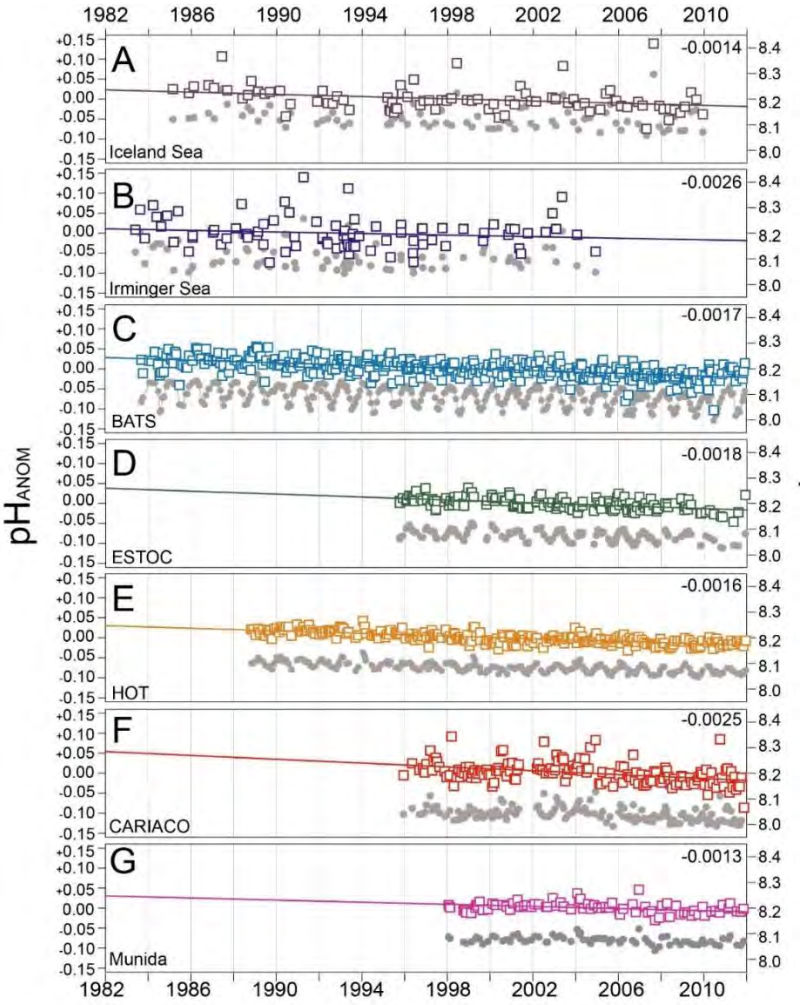


Figure 12.15: Time series of surface ocean pH (grey symbols, right hand axis) and pH anomaly (coloured symbols, left hand axis) at seven time series sites, including the Munida Time Series in New Zealand. Trends (pH change yr^{-1}) are given in the top right of each panel. (A) Iceland Sea, North Atlantic Ocean (purple); (B) Irminger Sea, North Atlantic Ocean (blue); (C) BATS, North Atlantic Ocean (cyan); (D) ESTOC, North Atlantic Ocean (green); (E) HOT, North Pacific Ocean (orange); (F) CARIACO, North Atlantic Ocean (red), and; (G) MUNIDA, South Pacific (pink). Image directly sourced from Bates et al. 2014.

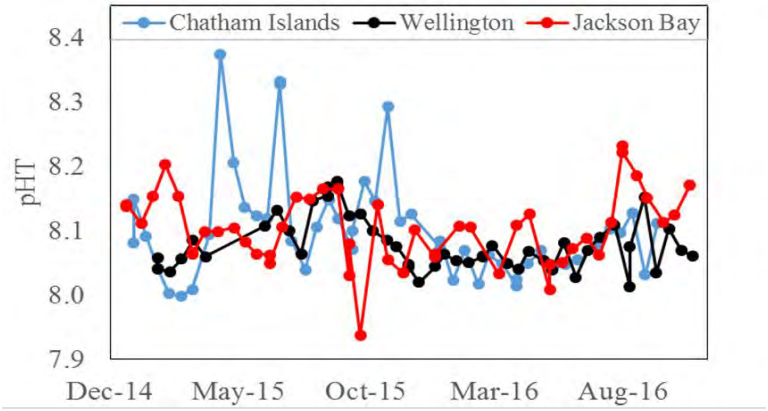


Figure 12.16: pH time series at three coastal stations; Chatham Islands (blue squares and line) Wellington (black circles and line); Jackson Bay (red triangles and line) in the NZOA-ON (Law et al. 2018a).

	Heterotrophs	Primary Producers		Primary and Secondary Consumers						Tertiary Consumers	Higher Trophic
	Bacteria	Phytoplankton	Macroalgae	Bryozoa	Sponges	Cold Water Corals	Crustacea	Molluscs	Echinoderms	Fish	Cetaceans, pinnepids, seabirds
a) Vulnerability to OA in NZ waters	**Low**	**Low-Med**	**Low-Med**	**?**	**?**	**Medium**	**?**	**Med-High**	**Med-High**	**?**	**?**
b) Current knowledge in NZ waters											
Established Response to OA	Medium*+	Medium*	Med-High	-	Low	Low-Med	-	Medium*	Medium+	Low-Med*	-
Mechanistic understanding of response	Low-Med	Low-Med	Medium	-	Low	Low	-	Low-Med*	Low-Med+	Low	-
Indirect/Ecosystem interaction	Medium*+	Low-Med*	Medium*	Low	Low	Low	-	Low	Low-Med	-	-
Interaction with other stressors	Low-Med*+	Med-High*	Medium	Low	Low	Low	-	Medium	Medium+	-	-
Socio-economic/Ecosystem services	Low	Low-Med	Low-Med	Low	-	Low	-	Medium	Low-Med		-
Adaptive capacity	-	-	-	-	-	-	-	-	Low	-	-

Low single study, low confidence; Low-Med 1-2 studies, low-medium confidence; Medium 2-3 studies, medium confidence; Med-High 4+ studies, medium-high confidence

Table 12.1: Qualitative assessment of a) the vulnerability, and b) the current state of knowledge of different aspects of OA research for the major biotic groups in New Zealand waters (Law et al. 2018a). Vulnerability is classified on a Low to High scale, with "?" indicating where vulnerability is currently unknown. Current knowledge is also classified on a Low to High scale, and based upon the number and results of published studies (see key), with "-" indicating that no studies have been carried out. Ongoing New Zealand studies are indicated by "+", with those in the CARIM project indicated by "*" (from Law et al. 2018a).

Projections for a range of climate-related variables in the surface ocean were generated by the *Climate Change: Impacts & Implications* project and published in Law et al. (2018b). The projections reflect the outputs of two Earth System Models (ESMs) identified as the "best" models for simulating current conditions in the surface ocean around New Zealand from analysis by Rickard et al. (2016). Law et al. (2018) provides the projected mean change for the middle and end of the 21st Century, for the RCP4.5 and 8.5 scenarios (as summarised in Table 12.2) and considers the spatial variation of change in waters around New Zealand. Results from this study include:

- Despite being the best models for NZ waters, the two ESMs provide different spatial distributions and magnitudes of change in most parameters (Figure 12.17)
- Projected Sea Surface Temperature increases of +1.6 to +2.95°C by 2100 are comparable with the means projected for both Australian and global waters (Figure 12.17).
- Highest projected regional warming occurs in the East Australian Current, and in subantarctic waters south of Chatham Rise. As the present warming rate in in the south-west Tasman Sea is similar to that projected for waters around NZ, this region may provide an analogue for future changes in NZ waters.

Table 2. Summary table of present-day and projected mean values (Δ = absolute change; % Δ = % change) for the middle and end of the 21st Century for all variables under RCP4.5 and 8.5 for the SWP region using the inner ESMs (R16).

		Present Day Mean	RCP4.5 Mid-Century			RCP4.5 End-Century			RCP8.5 Mid-Century			RCP8.5 End-Century		
Property	Unit		Mean	Δ	% Δ	Mean	Δ	% Δ	Mean	Δ	% Δ	Mean	Δ	% Δ
SST	°C	15.84	16.6	0.8	4.7	16.9	1.1	6.9	16.8	1	6.6	18.3	2.5	15.8
MLD	m	91	86.5	−4.5	−4.9	85.0	−6	−6.6	83.2	−7.8	−8.6	77.0	−14	−15.4
Nitrate	mmol m^{-3}	5.31	5.1	−0.2	−3	5.0	−0.28	−5.3	5.1	−0.2	−3.4	4.9	−0.4	−7.5
Phosphate	mmol m^{-3}	0.5	0.49	−0.01	−2	0.47	−0.03	−5.0	0.48	−0.02	−4.0	0.45	−0.05	−9.0
Silicate	mmol m^{-3}	3.2	3.1	−0.05	−1.5	2.7	−0.5	−15.6	3.0	−0.2	−6.3	2.55	−0.65	−20.3
Fe	mmol m^{-3}	0.13	0.16	0.03	19.2	0.17	0.05	34.62	0.14	0.01	7.7	0.16	0.04	26.9
pH		8.11	7.98	−0.12		7.95	−0.16		7.93	−0.18		7.77	−0.33	
[H+]	µmol kg^{-1}	0.008	0.010	0.003	32.1	0.011	0.003	42.2	0.012	0.004	48.3	0.017	0.009	116.3
Chl-a	mg m^{-3}	0.2	0.19	−0.005	−2.5	0.19	−0.01	−5.0	0.19	−0.01	−5.0	0.185	−0.015	−7.5
NPP	mmol $m^{-2}d^{-1}$	33.52	33.2	−0.35	−1.0	33.1	−0.4	−1.2	33.0	−0.5	−1.5	32.0	−1.5	−4.5
Exp	mg $m^{-2}d^{-1}$	50.2	49	−1.2	−1.3	47.3	−2.8	−4.5	47.5	−2.7	−4.4	43.1	−7.1	−12

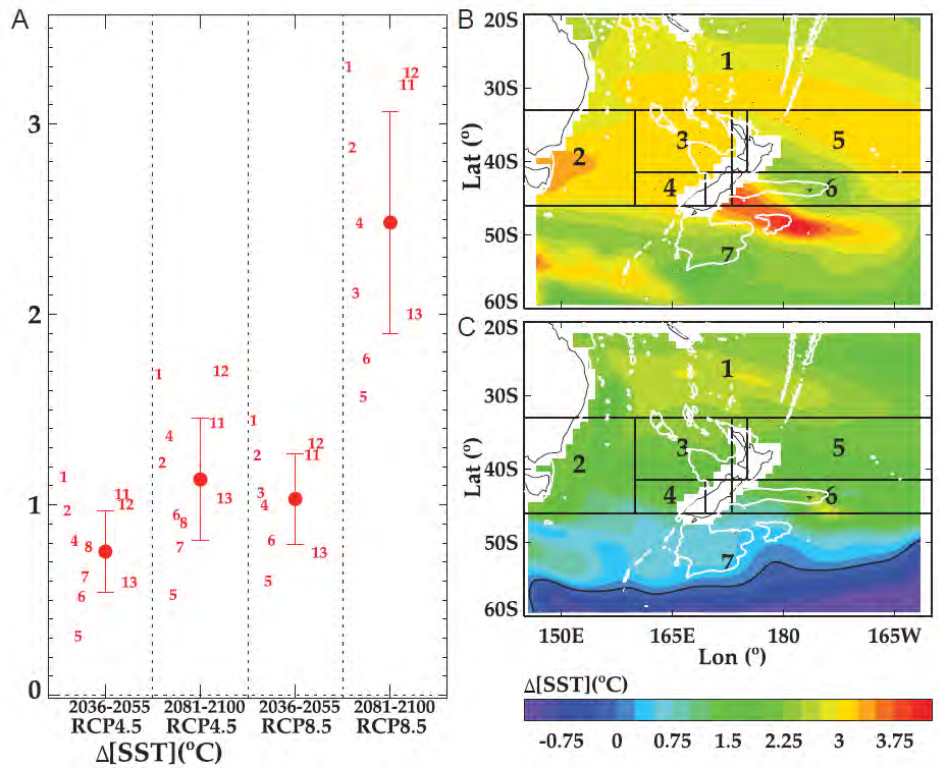


Figure 12.17: Projections of mean change in sea surface temperature (ΔSST, °C, ±1 standard deviation) for Mid and End-Century under RCP4.5 and 8.5 (delineated by vertical dashed lines) from the best ESM subset. The numbers indicate the mean for each individual ESM. The regional variation of ΔSST for the End-Century under RCP8.5 is shown for B, ESM2 and C, ESM5. The regional boxes are indicated by number, the white contours the 1000 m isobath and the black contours indicate zero change (from Law et al. 2018b).

- The depth of the surface mixed layer will decrease across much of the NZ area, except for some subantarctic water regions. The resulting increase in light exposure for plankton may be beneficial to productivity and food webs in subantarctic waters where nutrients are plentiful, but may be deleterious in warmer subtropical waters that are oligotrophic (i.e. low nutrient).
- Surface macronutrients (nitrate, phosphate and silicate) will decline across the NZ region, although silicate may increase in subantarctic and polar waters south of NZ. The projected declines in nitrate and phosphate are greatest east of Chatham Rise and in subantarctic waters south of the Rise. Conversely, dissolved concentrations of the micronutrient iron will increase in surface waters across much of the region.
- Surface chlorophyll-a (the pigment that provides an indicator of phytoplankton biomass) shows only minor changes across much of the region except for a band of frontal waters south-west and south-east of the South Island which shows the largest decline. The regional variations in primary productivity (see Fig. 12.18) are greater, with decreases limited to warmer oligotrophic waters, and most of the region south of 40°S showing an increase in primary production.
- Projected decreases in vertical particle flux exports are 4.5% and 12% under RCP4.5 and 8.5, respectively, for the NZ region by 2100, which is within the range of projections for decreases in global export production under RCP8.5 of 7–18% (Bopp et al. 2013). One of the two models indicates that the decline will be greatest on Chatham Rise, suggesting a significant reduction in food supply for foodwebs and fisheries, whereas the second model suggests a potential increase in flux in this region.

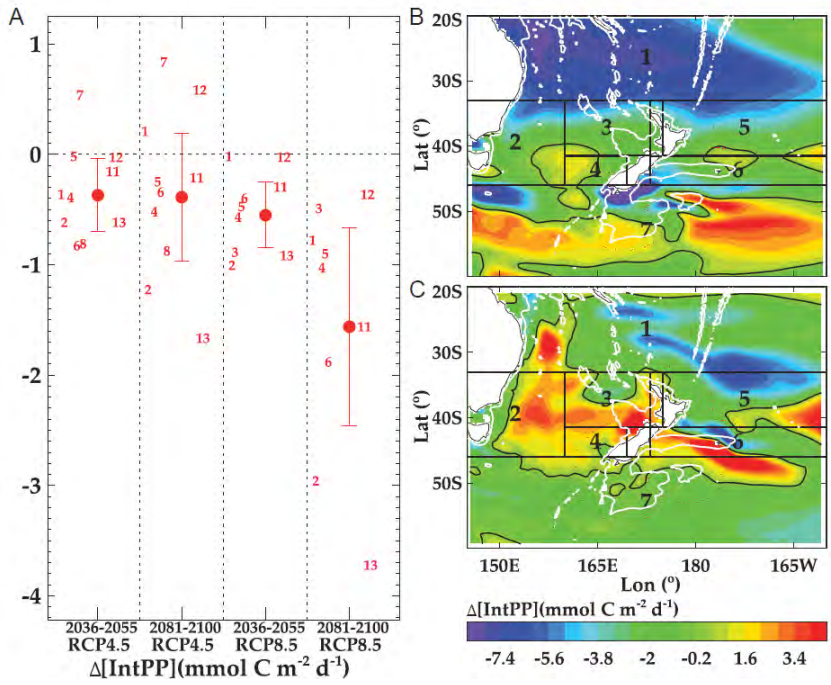


Figure 12.18: Projections of mean change in mean integrated primary production (ΔNPP, ±1 standard deviation) for Mid and End-Century under RCP4.5 and 8.5 (delineated by vertical dashed lines) from the best ESMs. Negative values indicate a decrease in rate and the horizontal dashed line indicates zero change. The numbers indicate the means for each ESM. Regional variation of the projected change in NPP for the End-Century under RCP8.5, using B, ESM2 and C, ESM5. The regional boxes are indicated by number, the white contour is the 1000 metre isobath and the black contours indicate zero change (from Law et al. 2018b).

- The most vulnerable regions were identified where regional extremes occurred in three or more climate variables (see Figure 12.19). Both ESMs identified Chatham Rise as the most significant vulnerable area, with a number of parameters increasing or decreasing in this region. Polar waters south of NZ and subtropical waters north of 30°S were also identified as vulnerable

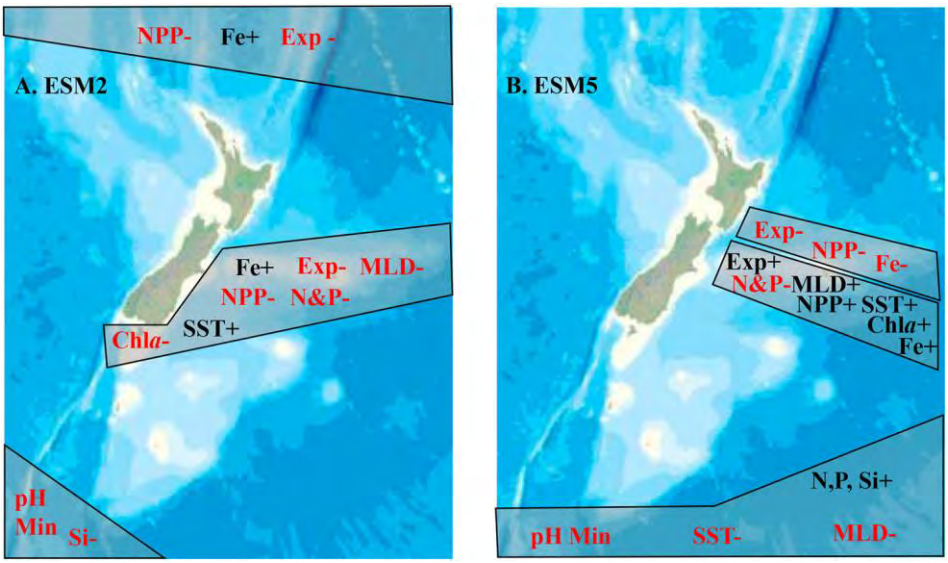


Figure 12.19: Regional extremes for climate-sensitive variables in the surface ocean around New Zealand projected for 2080–2100 using A) ESM2 and B) ESM5. The shaded areas represent potentially vulnerable regions, where two or more variables show significant change relative to the NZ mean. The change response is indicated by colour and sign, with a significant decrease indicated in red with a – symbol, and a significant increase in black with a +symbol. Key: SST: Sea Surface Temperature; MLD: Mixed Layer Depth; N&P: Nitrate and Phosphate; Si: Silicate; Fe: Dissolved Iron; Chla: Chlorophyll-a; NPP: Integrated Primary Production; Exp: Particle flux; pH Min: lowest regional pH (from Law et al. 2018b)

12.3 OCEAN CLIMATE TRENDS AND NEW ZEALAND FISHERIES

This section is quoted almost directly from the summary in Hurst et al. (2012) with updates from more recent literature where relevant. General observations on recent trends in some of the key ocean climate indices that have been found to be correlated with a variety of biological processes among fish (including recruitment fluctuations, growth, distribution, productivity and catch rates) are:

- The Interdecadal Pacific Oscillation (IPO): available from 1871; time scale 15–30 years. The IPO has been found to have been correlated with decadal changes ('regime shifts') in north-east Pacific ecosystems (e.g., Alaska salmon catches). In the New Zealand region, there is evidence of a regime shift into the negative phase of the IPO in about 2000. During the positive phase, from the late 1970s to 2000, New Zealand experienced periods of enhanced westerlies, with associated cooler air and sea temperatures and enhanced upwelling on western coasts. Opposite patterns are expected under a negative phase. For most New Zealand fisheries, monitoring of changes in populations began in the late 1970s, so there is little information on how New Zealand fishstocks might respond to these longer-term climatic fluctuations. Some of the recent changes in fish populations since the mid-1990s, for example, low western stock hoki recruitment indices (Francis 2009) and increases in some elasmobranch abundance indices (Dunn et al. 2009) may be shorter-term fluctuations that could be related to regional warming and longer-term monitoring is necessary to establish whether they might be related to longer-term ecosystem changes.
- The Southern Oscillation Index: available from 1876; best represented as smoothed values over at least 3–5 months. Causal relationships of correlations of SOI with fisheries processes are poorly understood but probably related in some way to one or more of the underlying ocean climate processes such as winds or temperatures. When the index is strongly negative, an El Niño event is taking place and New Zealand tends to experience increased westerly and south-westerly winds, cooler sea surface temperatures and enhanced upwelling in some areas (see, for example, the correlation of monthly SST at Leigh and Portobello with SOI indices, Figure 12.8). Upwelling has been found to be related to increased nutrient flux and phytoplankton growth in areas such as the west coast South Island, Pelorus Sound and north-east coast of the North Island (Willis et al. 2007, Zeldis et al. 2008). El Niño events are likely to occur on 3–7-year time scales and are likely to be less frequent during the negative phase of the IPO, which began in about 2000. This is likely to impact positively on species that show stronger recruitment under increased temperature regimes (e.g., snapper, Francis 1993, 1994a, 1994b).
- Surface wind and pressure patterns: available from the 1940s. Variation in pressure patterns can be high over monthly and annual time scales and many of the indices are correlated with each other, and with SOI and IPO indices (e.g. more zonal westerly winds, more frequent or regular cycles in southerlies in the positive IPO, 1977–2000). Correlations with biological process in fish stocks may occur over short time scales (e.g. impact on fish catchability) as well as seasonal and annual scales (e.g. impact on recruitment success). Wind and pressure patterns have been found to be correlated with fish abundance indices for southern gemfish (Renwick et al. 1998), hake, red cod and red gurnard (Dunn et al. 2009), rock lobster (Booth et al. 2000), and southern blue whiting (Willis et al. 2007, Hanchet & Renwick 1999). The mechanisms implied by the correlations are at best poorly understood, however they motivate hypothesis testing into the relationship between wind and pressure patterns and fisheries.
- Temperature and sea surface height: available at least monthly over either long time scales (air temperatures from 1906) or relatively short time scales (ocean temperatures to 800 m since 1986, satellite SST since 1981 and SSH since 1993). Ocean temperatures, SST and SSH are all correlated with each other and smoothed air temperatures correlate well with SST in terms of inter-annual and seasonal variability; there are also some correlations of SST and SSH with surface wind and pressure patterns (Dunn et al. 2009). SST has been found to be correlated with relative fish abundance indices (derived from fisheries and/or trawl

surveys) for elephantfish, southern gemfish, hoki, red cod, red gurnard, school shark, snapper, stargazer and tarakihi (Francis 1994a, 1994b, Renwick et al. 1998, Beentjes & Renwick 2001, Gilbert & Taylor 2001, Dunn et al. 2009). Air temperatures in New Zealand have increased since 1900 with most of the increase occurring since the mid-1940s. Increases from the late 1970s to 2000 may have been moderated by the positive phase of the IPO. Coastal SST records from 1954 at Portobello show a slight increase through the series. Other time series (SSH, ocean temperature to 800 m) are comparatively short but show cycles of warmer and cooler periods on 1–6-year time scales. All air and ocean temperature series show the significant warming event during the late 1990s which has been followed by a relatively stable period.

- Ocean colour and upwelling: these are important time series because they potentially have a more direct link to biological processes in the ocean and are more easily incorporated into hypothesis testing. The ocean colour series starts in late 1997, so is not able to study changes that occurred before the late 1990s warming event. These indices also need to be analysed with respect to SST, SSH and wind patterns at similar locations or on similar spatial scales. Preliminary series developed exhibit some important spatial differences and trends that may warrant further investigation in relation to fish abundance indices. Of note are increased chlorophyll indices off the west and south-west coast of the South Island in spring/summer during the last 5–6 years and relatively low upwelling indices off the west coast South Island during winter in the late-1990s (Hurst et al. 2012).
- Currents: there are no general indices of trends or variability at present. Improvements in monitoring technology (e.g., satellite observations of SSH; CTD; ADCP; Argo floats) have resulted in more information becoming available to study currents directly and enable the development of numerical models. Recent analyses of the currents along the eastern New Zealand margin (Fernandez et al. 2018) indicate that the currents are highly variable and that there is little coherence between water transports along the boundary. Furthermore, there are no discernible trends in the transports since the SSH measurements began in 1993. On the open ocean scale, there is considerable complexity in the New Zealand zone (e.g., frontal systems, eddy systems off the east coast). The coastal zone is further complicated by the effects of tides, winds and freshwater (river) forcing. Nevertheless, the importance of current systems is starting to become more recognised and has been incorporated into analysis and modelling of fisheries processes and trends. Recent examples include the retention of rock lobster phyllosoma (mid-stage larvae) in eddy systems (Chiswell & Booth 2005, 2007), the apparent bounding of orange roughy nursery grounds by the presence of a cold-water front (Dunn et al. 2009) and the drift of toothfish eggs and larvae (Hanchet et al. 2008).
- Acidification: Recent model projections indicate a decrease in mean pH by about 0.33 to 7.77 by 2100. A 20-year long New Zealand time-series shows acidification of Sub-Antarctic Water consistent with the increase in atmospheric CO_2. This timeseries is now complemented by a network of coastal stations that also highlight regional variation in pH. Some coastal waters, such as the Firth of Thames, already experience a seasonal decline in pH to 7.9, confirming that NZ coastal waters already experience greater acidification than open-ocean waters. A review of the impacts of ocean acidification on NZ species and ecosystems identifies that organisms with carbonate shells and skeletons are potentially most sensitive to a decrease in pH and that early life-history stages (eggs/larvae) may be particularly susceptible.
- Climate change was not specifically addressed as part of the report by Hurst et al. (2012), although the indices described are integral parts of monitoring the progress and impacts of climate change. As noted in the air temperature section, the slightly increasing trend in temperatures since the mid-1940s is likely to have been moderated by the positive phase of the IPO from the late 1970s to the late 1990s. With the shift to a negative phase of the IPO in 2000, it is likely that temperatures will increase more steeply. Continued monitoring of the ocean environment and response is critical. This includes not only the impacts on productivity, at all levels, but also increasing ocean acidification.

In conclusion, key ocean climate drivers in the New Zealand region for the last few decades have been:

- the significant warming event in the late 1990s;
- the shift to the negative phase of the IPO in about 2000, which is likely to result in fewer El Niño events for a 20–30-year period, i.e., fewer zonal westerly winds (already apparent compared to the 1980–2000 period) and increased temperatures; this is the first IPO change to occur since most of our fisheries monitoring time series have started (the previous shift was in the late 1970s); and
- global trends of increasing air and sea temperatures and ocean acidification.

12.4 REFERENCES

Bates, N R; Astor, Y M; Church, M J; Currie, K; Dore, J E; Gonzalez-Davila, M; Lorenzoni, L; Muller-Karger, F; Olaffson, J; Santana-Casiano, J M (2014) Changing ocean chemistry: A time-series view of ocean uptake of anthropogenic CO2 and ocean acidification. *Oceanography 27 (1)*: 126–141.

Beentjes, M P; Renwick, J A (2001) The relationship between red cod, *Pseudophycis bachus*, recruitment and environmental variables in New Zealand. *Environmental Biology of Fishes 61*: 315–328.

Behrens, E; Fernandez, D; Sutton, P (2018) Meridional oceanic heat transport influences marine heatwaves in the Tasman Sea on interannual to decadal timescales. Submitted to Frontiers in Marine Science.

Booth, J D; Bradford, E; Renwick, J (2000) *Jasus edwardsii* puerulus settlement levels examined in relation to the ocean environment and to subsequent juvenile and recruit abundance. *New Zealand Fisheries Assessment Report 2000/34*. 48 p.

Bopp, L; Resplandy, L; Orr, J; Doney, S; Dunne, J; Gehlen, M; Halloran, P; Heinze, C; Ilyina, T; Séférian, R et al. (2013) Multiple stressors of ocean ecosystems in the 21st century: projections with CMIP5 models. *Biogeosciences.* 10:6225–6245. doi:10.5194/bg-10-6225-2013

Bowen, M M; Sutton, P J H; Roemmich, D (2006) Wind-driven and steric fluctuations of sea surface height in the southwest Pacific, *Geophysical Research Letters 33*: L14617. doi:10.1029/2006GL026160.

Boyd, P; LaRoche, J; Gall, M; Frew, R; McKay R M L (1999) Role of iron, light, and silicate in controlling algal biomass in subantarctic waters SE of New Zealand. *Journal of Geophysical Research 104 (C6)*: 13395–13408. doi:10.1029/1999JC900009.

Bradford-Grieve, J M; Chang, F H; Gall, M; Pickmere, S; Richards, F (1997) Size-fractionated phytoplankton standing stocks and primary production during austral winter and spring 1993 in the Subtropical Convergence region near New Zealand. *New Zealand Journal of Marine and Freshwater Research 31*: 201–224.

Bradford-Grieve, J M; Probert, P K; Nodder, S D; Thompson, D; Hall, J; Hanchet, S, et al. (2003) Pilot trophic model for subantarctic water over the Southern Plateau, New Zealand: a low biomass, high transfer efficiency system. *Journal of Experimental Marine Biology and Ecology 289 (2)*: 223–262.

Brailsford G W; Stephens B B; Gomez A J; Riedel K; Mikaloff Fletcher S E; Nichol S E; Manning M R (2012) Long-term continuous atmospheric CO2 measurements at Baring Head, New Zealand. *Atmospheric Measurement Techniques 5(12)*: 3109–3117.

Brix, H; Currie, K I; Mikaloff-Fletcher, S (2013) Seasonal variability of the carbon cycle in subantarctic surface water in the South West Pacific. *Global Biogeochemical Cycles 27*: 1–12.

Bylenga, C; Cummings, V J; Ryan, K (2015) Fertilisation and larval development in the Antarctic bivalve, *Laternula elliptica* under reduced pH and elevated temperatures. *Marine Ecology Progress Series 536*: 187–201. Retrieved from http://www.int-res.com/prepress/m11436.html.

Byrne, M; Lamare, M; Winter, D; Dworjanyn, S A; Uthicke, S (2013) The stunting effect of a high CO2 ocean on calcification and development in sea urchin larvae, a synthesis from the tropics to the poles. *Philosophical Transactions of the Royal Society B 368*: 20120439. doi:10.1098/rstb.2012.0439.

Chang, F H; Mullan, B (2003) Occurrence of major harmful algal blooms in New Zealand: is there a link with climate variation. *The Climate Update 53*: 4.

Chiswell, S M (1996) Variability in the Southland Current, New Zealand. *New Zealand Journal of Marine and Freshwater Research 30*: 1–17.

Chiswell, S M; Booth, J D (2005) Distribution of mid- and late-stage *Jasus edwardsii* phyllosomas: implications for larval transport. *New Zealand Journal of Marine and Freshwater Research 39*. 1157–1170.

Chiswell, S M; Booth, J D (2007) Sources and sinks of larval settlement in *Jasus edwardsii* around New Zealand: Where do larvae come from and where do they go? *Marine Ecology Progress Series 354*: 201–217.

Chiswell, S M; Bostock, H C; Sutton, P J H; Williams, M J M (2015) Physical oceanography of the deep seas around New Zealand: a review. *New Zealand Journal of Marine and Freshwater Research 49 (2)*: 286–317. doi:10.1080/00288330.2014.992918.

Clark, M R (1999) Fisheries for orange roughy (*Hoplostethus atlanticus*) on seamounts in New Zealand. *Oceanologica Acta 22 (6)*: 593–602.

DeVries, T (2014) The oceanic anthropogenic CO2 sink: Storage, air-sea fluxes, and transports over the industrial era. *Global Biogeochemical Cycles*, *28*(7), pp.631–647.

Dunn, M R; Hurst, R; Renwick, J; Francis, R I C C; Devine, J; McKenzie, A (2009) Fish abundance and climate trends in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 31*. 75 p.

Feely, R A; Sabine, C L; Lee, K; Berelson, W; Kleypas, J; Fabry, V J; Millero, F J (2004) Impact of Anthropogenic CO2 on the CaCO3 System in the Oceans (abstract). *Science 305 (5682)*: 362–366. doi:10.1126/science.1097329.

Fernandez, D; Bowen, M; Sutton P (2018) Variability, coherence and forcing mechanisms in the New Zealand ocean boundary currents. *Progress in Oceanography (165)*: 168–188. doi.org/10.1016/j.pocean.2018.06.002

Field, C B; Behrenfeld, M J; Randerson J T; Falkowski, P (1998) Primary Production of the Biosphere: Integrating Terrestrial and Oceanic Components. *Science (281)*: 237–240. doi:10.1126/science.281.5374.237.

Forcén-Vázquez, A (2015) Oceanography of the New Zealand subantarctic region. PhD Thesis, Victoria University of Wellington, Wellington, New Zealand. 190 p.

Francis, M P (1993) Does water temperature determine year class strength in New Zealand snapper (*Pagrus auratus*, Sparidae)? *Fisheries Oceanography 2 (2)*: 65–72.

Francis, M P (1994a) Growth of juvenile snapper, *Pagrus auratus* (Sparidae). *New Zealand Journal of Marine and Freshwater Research 28*: 201–218.

Francis, M P (1994b) Duration of larval and spawning periods in *Pagrus auratus* (Sparidae) determined from otolith daily increments. *Environmental Biology of Fishes 39*: 137–152.

Francis, R I C C (2009) Assessment of hoki (*Macruronus novaezelandiae*) in 2008. *New Zealand Fisheries Assessment Report 2009/7*. 80 p.

Franke, A; Clemmesen, C (2011) Effect of ocean acidification on early life stages of Atlantic herring (*Clupea harengus* L.). *Biogeosciences Discussions 8*: 7097–7126.

Gammon, M J; Tracey, D M; Marriott, P M; Cummings, V J; Davy, S K (2018). The physiological response of the deep-sea coral Solenosmilia variabilis to ocean acidification. *PeerJ*, *6*, e5236.

Gilbert, D J; Taylor, P R (2001) The relationship between snapper (*Pagrus auratus*) year class strength and temperature for SNA 2 and SNA 7. *New Zealand Fisheries Assessment Report 2001/64*. 33 p.

Gordon, D P; Beaumont, J; MacDiarmid, A; Robertson, D A; Ahyong, S T (2010) Marine Biodiversity of Aotearoa New Zealand. *PLoS ONE 5 (8)*: e10905. doi:10.1371/journal.pone.0010905.

Hamilton, L J (2006) Structure of the Subtropical Front in the Tasman Sea. *Deep Sea Research I 53 (12)*: 1989–2009. doi:10.1016/j.dsr.2006.08.013.

Hanchet, S M; Fenaughty, J M; Dunn, A; Williams, M J H (2008) A hypothetical life cycle for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea region. *CCAMLR Science 15*: 35–53.

Hanchet, S M; Renwick, J A (1999) Prediction of year class strength in southern blue whiting (*Micromesistius australis*) in New Zealand waters. *New Zealand Fisheries Assessment Research Document 99/51*. 24 p. (Unpublished document held by NIWA library.)

Huang, B; Thorne, P W; Banzon, V F; Boyer, T; Chepurin, G; Lawrimore, J H; Menne, M J; Smith, T M; Vose, R S; Zhang H-M (2017) Extended Reconstructed Sea Surface Temperature version 5 (ERSSTv5), Upgrades, validations, and intercomparisons. *Journal of Climate*, 30: 8179–8205, https://doi.org/10.1175/JCLI-D-16-0836.1

Hurst, R J; Renwick, J A; Sutton, P J H; Uddstrom, M J; Kennan, S C; Law, C S; Rickard, G J; Korpela, A; Stewart, C; Evans, J (2012) Climate and oceanographic trends of potential relevance to fisheries in the New Zealand fisheries region. *New Zealand Aquatic Environment and Biodiversity Report No. 90*. 202 p.

Langer, G; Geisen, M; Baumann, K-H; Kläs, J; Riebesell, U; Thoms, S; Young, J R (2006) Species-specific responses of calcifying algae to changing seawater carbonate chemistry. *Geochemistry Geophysics Geosystems (7)*: Q09006. doi:10.1029/2005GC001227.

Law, C S; Bell, J J; Bostock, H C; Cornwall, C E; Cummings, V; Currie, K; Davy, S K; Gammon, M; Hepburn, C D; Hurd, C L; Lamare, M; Mikaloff-Fletcher, S E; Nelson, W A; Parsons, D M; Ragg, N L C; Sewell, M A; Smith, A M; Tracey, D M (2018a) Ocean Acidification in New Zealand waters. *New Zealand Journal of Marine & Freshwater Research* 52:2, 155–195, DOI: 10.1080/00288330.2017.1374983

Law, C S; Rickard, G J; Mikaloff-Fletcher, S E; Pinkerton, M H; Behrens, E; Chiswell, S M; Currie, K (2018b) Climate Change projections for the surface ocean around New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 52:3, 309–335, DOI: 10.1080/00288330.2017.1390772

Le Quéré, C; Andrew, R M; Friedlingstein, P; Sitch, S; Pongratz, J; Manning, A C; Korsbakken, J-I; Peters, G P; Canadell, J G; Jackson, R B; Boden, T A; Tans, P P; Andrews, O D; Arora, V K; Bakker, D C E; Barbero, L; Becker, M; Betts, R A; Bopp, L; Chevallier, F; Chini, L P; Ciais, Cosca, C E; Cross, J; Currie, K; Gasser, T; Harris, I; Hauck, J; Haverd, V; Houghton, R A; Hunt, C W; Hurtt, G; Ilyina, T; Jain, A K; Kato, E; Kautz, M; Keeling, R F; Goldewijk, K K; Körtzinger, A; Landschützer, P; Lefèvre, N; Lenton, A; Lienert, S; Lima, I; Lombardozzi, D; Metzl, N; Millero, F; Monteiro, P M S; Munro, D R; Nabel, J E M S; Nakaoka, S-I; Nojiri, Y; Padin, X A; Peregon, A; Pfeil, B; Pierrot, D; Poulter, B; Rehder, G; Reimer, J; Rödenbeck, C; Schwinger, J; Séférian, R; Skjelvan, I; Stocker, B D; Tian, H; Tilbrook, B; Tubiello, F N; van der Laan-Luijkx, I T; van der Werf, G R; van Heuven, S; Viovy, N; Vuichard, N; Walker, A P; Watson, A J; Wiltshire, A J; Zaehle,

S; Zhu, D (2018): Global carbon budget 2017. *Earth System Science Data*, **10** (1), 405–448. doi.org/10.5194/essd-10-405-2018

Lutz, M J; Caldeira, K; Dunbar, R B; Behrenfeld, M J (2007) Seasonal rhythms of net primary production and particulate organic carbon flux to depth describe the efficiency of biological pump in the global ocean. *Journal of Geophysical Research Oceans 112*: C10011. doi: 10.1029/2006JC003706.

Mu, J; Jin, F; Wang, J, Zheng, N; Cong, Y (2015) Effects of CO2-driven ocean acidification on early life stages of marine medaka (*Oryzias melastigma*). *Biogeosciences 12*: 3861–3868. doi:10.5194/bg-12-3861-2015.

Mullan, A B (1995) On the linearity and stability of Southern Oscillation-climate relationships for New Zealand. *International Journal of Climatology 15*: 1365–1386.

Mullan, A B; Stuart, S J; Hadfield, M G; Smith, M J (2010) Report on the Review of NIWA's 'Seven-Station' Temperature Series. *NIWA Information Series No. 78*. 175 p.

Munday, P L; Cheal, A J; Dixson, D L; Rummer, J L; Fabricius, K E (2014) Behavioural impairment in reef fishes caused by ocean acidification at CO2 seeps. *Nature Climate Change 4*: 487–492. doi:10.1038/nclimate2195.

Munday, P L; Watson, S A; Parsons, D M; King, A; Barr, N G; Mcleod, I M; Allan, B; Pether, S M (2015). Effects of elevated CO2 on early life history development of the yellowtail kingfish, *Seriola lalandi*, a large pelagic fish. *ICES Journal of Marine Science*, *73*(3), 641–649. DOI: 10.1093/icesjms/fsv210

Orr, J C; Fabry, V J; Aumont, O; Bopp, L; Doney, S C; Feely, R A; Gnanadesikan, A; Gruber, N; Ishida, A; Joos, F; Key, R M; Lindsay, K; Maier-Reimer, E; Matear, R; Monfray, P; Mouchet, A; Najjar, R G ;Plattner, G-K; Rodgers, K B; Sabine, C L; Sarmiento, J L; Schlitzer, R; Slater, R D; Totterdell, I J; Weirig, M-F; Yamanaka, Y; Yool, A (2005) Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. *Nature 437*: 681–686.

Pinkerton, M H; Richardson, K M; Boyd, P W; Gall, M P; Zeldis, J; Oliver, M D; Murphy, R J (2005) Intercomparison of ocean colour band-ratio algorithms for chlorophyll concentration in the Subtropical Front east of New Zealand. *Remote Sensing of Environment 97*: 382–402.

Reid, J L (1986) On the total geostrophic circulation of the South Pacific Ocean: flow patterns, tracers and transports. *Progress in Oceanography 16*: 1–61.

Renwick, J A; Hurst, R J; Kidson, J W (1998) Climatic influences on the recruitment of Southern Gemfish (*Rexea solandri*, Gempylidae) in New Zealand waters. *International Journal of Climatology 18*: 1655–1667.

Reynolds, R W; Smith, T M; Liu, C; Chelton, D B; Casey, K S; Schlax, M G (2007) Daily High-Resolution-Blended Analyses for Sea Surface Temperature. *Journal of Climate*, 20, 5473–5496, https://doi.org/10.1175/2007JCLI1824.1

Rhein, M; Rintoul, S R; Aoki, S; Campos, E; Chambers, D; Feely, R A; Gulev, S; Johnson, G C; Josey, S A; Kostianoy, A; Mauritzen, C; Roemmich, D; Talley, L D; Wang, F (2013) Observations: Ocean. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Rickard, G J; Behrens, E; Chiswell, S M (2016) CMIP5 earth system models with biogeochemistry: an assessment for the southwest Pacific Ocean. *Journal of Geophysical Research Oceans.* 121:7857–7850. doi:10.1002/2016JC011736.

Ridgway, K R; Dunn, J R (2003) Mesoscale structure of the mean East Australian Current system and its relationship with topography. *Progress in Oceanography 56*: 189–222.

Roemmich, D; Gilson, J K; Davis, R; Sutton, P; Wijffels, S; Riser, S (2007) Decadal Spin-up of the South Pacific Subtropical Gyre. *Journal of Physical Oceanography 37*: 162–173.

Roemmich, D; Gilson, J; Sutton, P; Zilberman, N (2016) Multidecadal Change of the South Pacific Gyre Circulation. *Journal of Physical Oceanography, 46*: 1871- 1883. DOI: 10.1175/JPO-D-15_0237.1

Roemmich, D; Sutton, P (1998) The mean and variability of ocean circulation past northern New Zealand: determining the representativeness of hydrographic climatologies. *Journal of Geophysical Research. 103:13,041-13,054.* doi:10.1029/98JC00583.

Sabine, C L; Feely, R A; Gruber, N; Key, R M; Lee, K; Bullister, J L; Wanninkhof, R; Wong, C S; Wallace, D W; Tilbrook, B; Millero, F J; Peng, T H; Kozyr, A; Ono, T; Rios, A F (2004) The oceanic sink for anthropogenic CO2. *Science 305*: 367–371.

Salinger, J; Behrens, E; Mullan, A B; Diamond, H J; Renwick, J; Sirguey, P; Smith, R O; Trought, M C T; Alexander, L V; Cullen, N J; Fitzharris, B B; Hepburn, C D, Parker A K; Sutton, P J (2018) The unprecedented coupled ocean-atmosphere summer heatwave in the New Zealand Region 2017/18: Drivers, Mechanisms and Impacts. *Environmental Research Letters* (in press).

Shirtcliffe, G G L; Moore, M I; Cole, A G; Viner, A B; Baldwin, R; Chapman, B (1990) Dynamics of the Cape Farewell upwelling plume, New Zealand. *New Zealand Journal of Marine and Freshwater Research 24*: 555–568.

Smith, R O; Vennell, R; Bostock, H C; Williams, M J M (2013) Interaction of the subtropical front with topography around southern New Zealand. *Deep Sea Research I 76*: 13–26.

Stanton, B R (1981) An oceanographic survey of the Tasman Front. *New Zealand Journal of Marine and Freshwater Research 15*: 289–297.

Stanton, B R; Ridgway, N M (1988) An oceanographic survey of the subtropical convergence zone in the Tasman Sea. *New Zealand Journal of Marine and Freshwater Research 22*: 583–593.

Stramma, L; Peterson, R G; Tomczak, M (1995) The South Pacific Current. *Journal of Physical Oceanography 25:* 77-91.

Sutton, P J H (2003) The Southland Current: a subantarctic current. *New Zealand Journal of Marine and Freshwater Research 37*: 645–652.

Sutton, P J H; Bowen, M (2014) Flows in the Tasman Front south of Norfolk Island. *Journal of Geophysical Research: Oceans 119*: 3041–3053.

Sutton, P J H; Bowen, M (2019) Ocean Temperature Change around New Zealand over the last 36 years. *New Zealand Journal of Marine and Freshwater Research* Doi: 10.1080/00288330.2018.1562945

Sutton, P; Bowen, M; Roemmich, D (2005) Decadal Temperature changes in the Tasman Sea. *New Zealand Journal of Marine and Freshwater Research 39 (6)*: 1321–1329.

Sutton, P J H; Roemmich, D (2001) Ocean temperature climate off north-east New Zealand. *New Zealand Journal of Marine and Freshwater Research 35*: 553–565.

Thompson, D W J; Solomon, S (2002) Interpretation of recent Southern Hemisphere climate change. *Science 296*: 895–899.

Thompson, P A; Baird, M E; Ingleton, T; Doblin, M A (2009) Long-term changes in temperature Australian waters: implications for phytoplankton. *Marine Ecology Progress Series 394*: 1–19.

Thresher, R E; Guinotte, J; Matear, R; Fallon, S (2012) Adapting to the effects of climate change on Australia's deep marine reserves. Fisheries Research and Development Corporation (FRDC)/National Climate Change Adaptation Research Fund (NCCARF) Report 2010/510. 68 p.

Tracey, D M; Rowden, A A; Mackay, K A; Compton, T (2011) Habitat-forming cold-water corals show affinity for seamounts in the New Zealand region. *Marine Ecology Progress Series 430*: 1–22.

Watson, S A; Allan, B J; McQueen, D E; Nicol, S; Parsons, D M; Pether, S M; Pope, S; Setiawan, A N; Smith, N; Wilson C; Munday, P L (2018). Ocean warming has a greater effect than acidification on the early life history development and swimming performance of a large circumglobal pelagic fish. *Global Change Biology* 24(9):4368–4385. DOI: 10.1111/gcb.14290

Willis, T J; Handley, S J; Chang, F H; Morrisey, D J; Mullan, B; Pinkerton, M; Rodgers, K L; Sutton, P J H; Tait, A (2007) Climate change and the New Zealand marine environment. *NIWA Client Report NEL2007-025*, prepared for the Department of Conservation. NIWA Project: DOC08305.

Zeldis, J R (2004) New and remineralised nutrient supply and ecosystem metabolism on the northeastern New Zealand continental shelf. *Continental Shelf Research 24*: 563–581.

Zeldis, J R; Howard-Williams, C; Carter, C M; Schiel, D R (2008) ENSO and riverine control of nutrient loading, phytoplankton biomass and mussel aquaculture yield in Pelorus Sound, New Zealand. *Marine Ecology Progress Series 371*: 131–142. doi:10.3354/meps07668.

Zeldis, J R; Swaney, D P (2018) Balance of Catchment and Offshore Nutrient Loading and Biogeochemical Response in Four New Zealand Coastal Systems: Implications for Resource Management. *Estuaries and Coasts*, 1–20.

13 TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

Status of chapter	This chapter has not been updated for AEBAR 2018.
Scope of chapter	This chapter outlines the global and New Zealand understanding of trophic and ecosystem-level effects of fishing, with respect to types of effects, their causes, the types of ecosystems most likely to be affected, the spatial scales of effects, and indicators of trophic and ecosystem-level effects.
Area	All areas and fisheries.
Focal localities	Whole EEZ.
Key issues	Organisms in an ecosystem are linked by trophic (feeding) connections. Changes to one organism (by whatever means) can affect other organisms and sometimes large parts of the food web. Changes occurring across many trophic levels (ecosystem-level changes) can have implications for ecosystem resilience.
Emerging issues	Ecosystem approach to fisheries and how fishing interacts with other stressors of marine ecosystems
MPI research (current)	ZBD200505 *Long term change in New Zealand coastal ecosystems*; HMS2014-05 *Stable isotope analysis of highly migratory species to assess trophic linkages and spatial and temporal movement trends of HMS sharks*.
NZ government research (current)	• NIWA core funding - Coasts & Oceans centre: 'Ecosystem structure and function' and 'Marine Biological Resources'; Fisheries centre: 'Ecosystem effects of fishing'. • Climate Change Impacts and Implications (MBIE Contestable, http://ccii.org.nz). • Marine Futures (MBIE Contestable, http://www.niwa.co.nz/coasts-and-oceans/research-projects/marine-futures).
Related chapters/issues	Effects of fishing on ecologically dependent species. Benthic impacts of fishing (including habitats of particular significance for fisheries management). Climate and oceanographic context of New Zealand fisheries (including effects of climate variability and change). Land-based effects on fisheries. Marine biodiversity. Marine biosecurity Other work on fishstocks, marine mammals, seabirds, bycatch, etc.

13.1 CONTEXT

13.1.1 SCOPE OF CHAPTER

This chapter addresses trophic and ecosystem-level effects which may arise from fishing or from other drivers of change on marine ecosystems in the New Zealand region. 'Trophic effects' are changes to the structure and function of ecosystems occurring entirely or largely because of changes in the feeding of organisms within a food web. 'Ecosystem-level effects' are defined as changes occurring across several trophic levels.[1] An ecosystem is defined as a biological community of interacting organisms and their physical environment. The region of interest for the purposes of this chapter is the New Zealand marine exclusive economic zone (EEZ) and territorial waters,

[1] 'Trophic level' is a measure of the position of an organism within a food web. Primary producers have trophic level 1, herbivores have trophic level 2, and carnivores have trophic levels between about 3 and 5 in aquatic systems (Lindeman 1942).

including coastal and offshore regions. The focus is on wild-caught fisheries rather than aquaculture.

This chapter focuses on trophic and ecosystem-level effects that are relevant to the sustainability and environmental effects of New Zealand fisheries as set out in the relevant New Zealand legislation, current New Zealand government strategic/operational policies, and international best practice. Relevant legislation, policies and best practices are summarised in Chapter 1 (Sections 1.2 and 1.3). The relevance of these specifically to trophic and ecosystem-level effects include:

- The Fisheries Act 1996 requires that (a) associated or dependent species should be maintained above a level that ensures their long-term viability; (b) biological diversity of the aquatic environment should be maintained.
- MPI's Strategy 'Our Strategy': to grow the sustainable use of our natural resources.[2]
- FAO best practice requires the application of scientific methods and tools that go beyond the single-species approaches: '*Managers and decision-makers must now explicitly consider interactions in the ecosystem*' and scientific advice should include ecosystem considerations (FAO 2008).
- Marine Stewardship Council (MSC) Principle 2: '*Fishing operations should allow for the maintenance of the structure, productivity, function and diversity of the ecosystem (including habitat and associated dependent and ecologically related species) on which the fishery depends*.' (Marine Stewardship Council 2010). This only applies to those fisheries that are MSC certified.

Effects of fishing on target species are considered in the annual New Zealand Fisheries Assessment Plenary (Ministry for Primary Industries 2014) The Fisheries Assessment Plenary also includes consideration of the effects of fishing on the aquatic environment (under the 'environmental and ecosystem considerations' section for each stock). Effects of fishing all stocks on protected species, non-protected bycatch species, and on the benthos are given in other chapters of this AEBAR document. In particular, effects of fishing on seabirds and marine mammals which occur through trophic connections (e.g., fishing affecting the availability of prey for seabirds) are considered in Theme 1 of this report.

13.1.2 WHAT ARE TROPHIC AND ECOSYSTEM-LEVEL EFFECTS?

Trophic and ecosystem-level effects are changes to multiple parts of the food web. Such effects can occur in coastal or deepwater ecosystems and can involve a wide range of biological, chemical and physical processes. Because trophic and ecosystem-level effects occur over a range of different organisms and time/space scales, it is often difficult to be sure of the magnitude of the change or its underlying cause. This has led to much speculation and disagreement as to the mechanism or processes involved, and a corresponding high level of disagreement as to what management should have done to prevent it, or should do to respond to the change once it has occurred (Schiermeier 2004, Hilborn 2007, Murawski et al. 2007, Schiel 2013). Sometimes controlled experiments are conducted to see if trophic effects can be simulated, but low statistical power is a common problem of this kind of test (Schroeter et al. 1993). In general, international research on trophic and ecosystem-level effects is active and one where there are generally more hypotheses than well-accepted empirical demonstrations of the effects. It is probably useful to start with a few examples of some trophic and ecosystem-level effects.

As part of the widespread pattern of collapses of cod (*Gadus morhua*) populations in the North Atlantic in the late 1980s and the 1990s, cod biomass off the US East Coast dropped by a factor of five, from more than 150 000 metric tonnes (MT) to about 30 000 t (Mayo et al. 1998). With some slight lag, local stocks of the cod's favoured prey, Atlantic herring (*Clupea harengus*), increased over the same period 20-fold, to nearly 2 million t (NEFSC 1998). Elsewhere, on the opposite side of the Atlantic, a collapse of the cod resource in the Baltic Sea was followed by an

[2] Ministry for Primary Industries. Our Strategy. http://www.mpi.govt.nz/about-mpi/our-strategy.

eight fold increase in abundance of European sprat (*Sprattus sprattus*) – a major prey item for cod in that ecosystem (Köster et al. 2003b, Casini et al. 2008, 2009). In these cases, a reduction in the abundance of a piscine predator by fishing led to an increase in the prey species – a large-scale 'predation release' effect (see Section 13.1.3.1).

In New Zealand, observations in a number of northern marine reserves showed an increase in the abundance and size of red rock lobsters and piscine predators of algal grazing invertebrates which coincided with a gradual decrease in urchin density and an increase in algal cover (Babcock et al. 1999, Shears & Babcock 2002, 2003, Salomon et al. 2008, Babcock et al. 2010). These changes, suggestive of a trophic cascade (see Section 13.1.3.2) are consistent with the results of ecosystem models of the role of rock lobsters in New Zealand rocky reef ecosystems, using both qualitative (Beaumont et al. 2009) and quantitative frameworks (Pinkerton et al. 2008, Eddy et al. 2014, Pinkerton 2012). Shears et al. (2008) found that the occurrence of this trophic cascade in northern New Zealand was likely to vary at local and regional scales in relation to abiotic factors. From a New Zealand-wide perspective, Schiel (2013) concludes that urchin predators play a role in the dynamics of kelp beds only in some northern localities, and that environmental and climatic influences, species' demographics, and catchment-derived sedimentation are generally more important.

13.1.3 TYPES OF TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

13.1.3.1 FIRST ORDER TROPHIC EFFECTS: PREY AVAILABILITY AND PREDATION RELEASE

Changes to the abundance, size structure and functional type[3] of a species can affect both its predators and prey by trophic interactions (Pace et al. 1999). Increasing the abundance of a prey species may positively affect its predators (because they have to work less hard to find food) whereas reducing the abundance of a prey item may have a detrimental effect on the predators (by requiring them to hunt more intensively or by forcing a change in their diet); these are 'prey availability' or bottom-up effects (Trillmich et al. 1991, Jahncke et al. 2004). Alternatively, changing the abundance of a predator may affect the abundance of some or all of its prey by changing their natural mortality rates (a top-down effect; Northcote 1988). Decreasing the abundance of a predator (for example by fishing a predatory fish) may cause the abundance of some or all of its prey to increase (a 'predation release' effect; Casini et al. 2012). These effects act over one trophic link and are hence called 'first order' trophic effects.

13.1.3.2 TROPHIC CASCADES

Changes in the abundance of one species may go on to affect other species that are neither its predators nor its prey. This is a second-order trophic effect (occurring via an intermediate organism), often called a 'trophic cascade'. The awareness of trophic cascades arose originally from work in the marine intertidal zone, and lakes (Hrbácek et al. 1961, Shapiro et al. 1975, Paine 1980), but has since become the focus of considerable theoretical and empirical research in marine ecosystems (Carpenter et al. 1985, McQueen & Post 1988a, 1988b, Christoffersen et al. 1993, Pace et al. 1999, Frank et al. 2005, Borer et al. 2005, Daskalov et al. 2007, Möllmann et al. 2008, Casini et al. 2009, Schiel 2013). While the term trophic cascade was originally termed for top-down effects of predators, it is now usually defined as the propagation of indirect effects between nonadjacent trophic levels in a food chain or food web, whatever the direction of forcing (Gruner 2013). Thus, trophic cascades may also occur when changes in the populations of primary producers force changes at higher tropic levels (Beaugrand & Reid 2003, Bakun 2010). The potential for cascading effects of fishing in marine ecosystems is now thought to be as strong as or stronger than in freshwater ecosystems (Pace et al. 1999, ICES 2005, Borer et al. 2005).

[3] 'Functional type' refers to the collection of life history and ecological characteristics of an organism, including whether it is an herbivore, carnivore or omnivore, its feeding behaviour (including size of prey), location in the water column/benthos, and mobility.

A well-recognised example of a top-down cascade is the sea otter (*Enhydra lutris*), urchin (*Strongylocentrotus* spp.), kelp (*Macrocystis pyrifera* and other kelps) cascade in the north-east Pacific where hunting of sea otters in the eighteenth and nineteenth centuries allowed urchin populations to increase leading to over grazing of kelp beds (Szpak et al. 2013). Protection of sea otters and subsequent expansion or reintroduction of populations into its former range reversed this cascade (Estes & Palmisano 1974, Estes 1996, Estes & Duggins 1995). The generality of the sea otter-urchin-kelp cascade has been questioned; for example, based on experimental treatments, Carter et al. (2007) concluded that 'the sea otter-trophic cascade paradigm is not universally applicable across locations or habitat types.'

Where ecosystems are subject to stressors acting on different parts of the system together, changes due to cascading trophic effects can be extensive. For example, using field data collected over a 33-year period, Casini et al. (2008, 2009) showed a four-level community-wide trophic cascade in the open waters of the Baltic Sea. The dramatic reduction of the cod (*Gadus morhua*) population directly affected its main prey, the zooplanktivorous sprat (*Sprattus sprattus*) and indirectly the summer biomass of zooplankton and phytoplankton. Changes to the stock size of cod also affected the type of ecosystem control at the level of zooplankton. The cod-dominated configuration was characterized by low sprat abundance and independence between zooplankton and sprat variations (zooplankton abundance was controlled by oceanographic forcing). An alternate sprat-dominated configuration also existed in which cod biomass was low and zooplankton were strongly controlled by sprat predation (Casini et al. 2009).

13.1.4 REGIME SHIFT AND INVASIVE SPECIES

An ecosystem can change to an alternative state if perturbations are greater than its resilience can accommodate – this transition is called a regime-shift (Aebischer et al. 1990, Estes & Duggins 1995, Beaugrand et al. 2002, Daskalov et al. 2007). Regime shifts can occur over large scales, affect many parts of the ecosystem and may be hard or slow to reverse ('hysteresis'). It has been suggested that ecosystem-level restructuring may maintain the system in its new state by means of negative feedbacks (Bakun 2006, Casini et al. 2009, Möllmann et al. 2009, Lindegren et al. 2010). Well-documented oceanographic-induced regime shifts in marine ecosystems have historically had substantial, long-lasting and typically (but not always) negative effects on fisheries. For example, during the 1980s, the North Sea experienced a change in hydro-climatic forcing that caused a rapid, temperature-driven ecosystem shift (Beaugrand & Ibanez 2004). In the North Sea the new dynamic regime after the late 1980s favoured jellyfish in the plankton and decapods and detritivores (echinoderms) in the benthos (Kirby et al. 2008, 2009). The cod stocks in the North Sea and central Baltic Sea collapsed simultaneously with the ecosystem changes caused by the large-scale oceanographic changes (Reid et al. 2003, Beaugrand 2004, Weijerman et al. 2005, Casini et al. 2008, Möllmann et al. 2008, Lindegren et al. 2010).

In another type of regime shift, there has been much recent debate as to whether in some regions, more intense, more frequent or more extensive blooms of jellyfish [4] are occurring in response to trophic and ecosystem-level changes in ocean ecosystems (Brodeur et al. 1999, 2002, Mills 2001, Lynam et al. 2006). In an example reported by Bakun & Weeks (2006), a massive ctenophore ('comb jelly') breakout in the early 1990s led to a nearly total collapse of fisheries in the Black Sea. The Black Sea ecosystems' historically dominant zooplanktivore, European anchovy (*Engraulis encrasicolus*), is a small, filter-feeding pelagic fish. In the late 1980s anchovy landings in the Black Sea increased to levels approaching 900 000 t per year. At their maximum, in 1988, the catch of anchovy represented more than 60% of the total fishery catches taken from the Black Sea. As a result of heavy fisheries exploitation, anchovy spawning biomass in the following year declined by more than 85%. Shiganova (1998) reports that in the year after this drastic reduction in anchovy biomass, zooplankton abundance increased markedly. It was at this point, probably due to the enhanced food source, that the biomass of the ctenophore *Mnemiopsi leidyi* (a gelatinous zooplanktivorous species) in the Black Sea increased to a billion tonnes.

[4] 'Jellyfish' is often taken to include Medusozoa, Ctenophora and Thaliacea (Condon et al. 2013) but should strictly be limited to Medusozoa and Ctenophora (Gibbons & Richardson 2013).

Condon et al. (2013) assembled all available published and unpublished long-term time series on jellyfish abundance across the oceans (no data from the New Zealand region) and found evidence of an approximately 20-year oscillation in global jellyfish abundance. Although an overall global increase in jellyfish abundance over the whole observational period 1874–2011 could not be detected, there was a weak but significant overall increase in jellyfish abundance since 1970. Gibbons & Richardson (2013) note that it is clear that we currently do not know whether there are really global increases in jellyfish, but that a more relevant question is whether jellyfish abundances are increasing in areas that are particularly important for humans – i.e., the coastal zone and important fishing areas – because costs of jellyfish blooms in these areas can be considerable. Recent increases in jellyfish abundance may be linked to one or more of: (a) warmer seas that enhance production, feeding and growth rates of jellyfish (Purcell 2005); (b) overfishing of competitors of jellyfish (Daskalov et al. 2007); (c) increased supply of planktonic food for jellyfish associated with eutrophication of coastal waters (Parsons & Lalli 2002); (d) the spread of hypoxia, to which jellyfish exhibit greater tolerance than most other metazoans (Vaquer-Sunyer & Duarte 2008, Purcell 2012); and (e) increase of artificial structures in coastal zones that may be habitats for jellyfish polyps (Duarte et al. 2012).

13.1.4.1 EFFECTS OF CLIMATE CHANGE

Internationally and domestically, there is increasing recognition of the potential impacts of climate change on fisheries (IPCC 2007a, 2007b, Valdes et al. 2009, Rice & Garcia 2011). A changing climate may:

- affect individual physiological and behavioural responses of organisms (or some life stages of organisms; Petitgas et al. 2013), which could lead to effects at the population level (Rijnsdorp et al. 2009, O'Connor et al. 2007, Perry et al. 2005);
- change species proportions in fish assemblages (Engelhard et al. 2011, Fulton 2011);
- lead to ocean acidification, which may affect lower food web structure and adversely impact calcifying organisms such as shellfish and corals (Fabry et al. 2008, Cooley & Doney 2009);
- increase climate variability (Collins 2000), which may increase the risk of regime shift (Mullan et al. 2001, Beaugrand 2004);
- change species ranges, which might destabilise species relationships that help maintain ecosystem processes (Rice & Garcia 2011);
- lead to phonological (timing patterns) mismatches of grazers and predators (Sydeman & Bograd 2009);
- lead to invasive species becoming a greater threat (ICES 2005).

The global scientific understanding of how a changing climate may affect marine ecosystems is largely hypothetical to date, but it seems likely that impacts of climate change are likely to be largely trophic or ecosystem-level effects in nature (reviews by Lehodey et al. 2006, Drinkwater et al. 2010, Bakun 2010, Portner & Peck 2010, Ottersen et al. 2010, Overland et al. 2010, Hollowed et al. 2013).

13.1.4.2 POTENTIAL FOR RECOVERY FOLLOWING OVER-DEPLETION

It is possible that trophic and system-level effects of fishing can affect the ability of fisheries to recover (rebuild) following over-exploitation, but this is disputed. Some scientists suggest that after a fisheries collapse the collapsed population often takes much longer to recover than expected based on known biological parameters, the previously observed carrying capacity of the habitat, and the fact that each adult female fish may spawn tens of thousands to millions of eggs (Hutchings 2000, Steele & Schumacher 2000). It is argued that something durable and significant can be done to the ecosystem during over-exploitation and that this inhibits recovery even if fishing mortality is reduced. For example, in the mid-1960s the sardine fishery in the northern Benguela collapsed from a high point of annual catches of about 1.5 million t (Boyer 1996). Meanwhile, the other major fishery resources of the region, hake (*Merlucius paradoxus* and *M. capensis*) and horse mackerel (*Trachurus trachurus capensis*) also fell to low abundance levels and have not recovered (Bakun & Weeks 2006). The suggestion is that sardines previously occupied the key central position in the ecosystem structure and that these exploitable species have now been largely replaced by a combination of 'jelly predators' and pelagic gobies in a stable, alternative ecosystem state (Boyer & Hampton 2001, Lynam et al. 2006, Bakun & Weeks 2006).

One hypothesis for how trophic effects can prevent stock recovery is the 'cultivation/depensation' mechanism (Köster & Möllmann 2000, Walters & Kitchell 2001). In this hypothesis, consider a species X whose adults predate a species Y, but whose recruits are predated by species Y. If adults of X are abundant they can create favourable conditions for their own offspring by reducing the abundance of Y and hence reducing mortality of their pre-recruits. If the abundance of adults of X is reduced by fishing, expansion of Y may prevent re-establishment of the former species by increasing predation on the recruits of X (Folke et al. 2004). A less theoretical example is that of Casini et al. (2008), based on a 33-year time series in the Baltic Sea, that showed the reduction of the cod population by fishing led to increases in abundances of sprat. Sprat, besides being preyed upon by cod, prey heavily on cod eggs and early larvae (Casini et al. 2004). Some authors have concluded that this predation, together with the likelihood that zooplanktivorous cod larvae may suffer food competition with the high sprat population, was probably a significant factor preventing the resurgence of that cod population (Jarre-Teichmann et al. 2002, Köster et al. 2003a, 2003b, Casini et al. 2009).

However, the prevalence of trophic or ecosystem-level effects slowing or stopping recovery after fisheries collapses is disputed. Cardinale & Svedäng (2011) studied the recent recovery of the eastern Baltic cod stock after more than 20 years of low biomass and productivity and concluded that the recovery was driven by a sudden reduction in fishing mortality and occurred in the absence of any exceptionally large year classes. The recovery of the cod stock during a 'cod-hostile' ecological regime is taken by Cardinale & Svedäng (2011) as indicative of fisheries (rather than climate or food web effects) being the main regulator of cod population dynamics in the Baltic Sea. Cardinale & Svedäng (2011) concluded that single species regulation still seems to be a well-functioning approach in handling natural resources, provided that it includes both temporal and spatial aspects of stock dynamics and fleet behaviour.

13.1.4.3 EFFECTS ON SCAVENGING SPECIES

Offal and discards from fishing vessels can be important sources of food for some marine species, and this constitutes a trophic perturbation to the ecosystem. In addition to scavenging of discards, fish are known to prey on biota damaged or revealed by recent trawling (Kaiser & Spencer 1994). This may include benthic prey items not normally available to the fish (Dunn et al. 2009a). Seabird diets (and ecological success) are also potentially affected by availability of offal and discards near the sea surface. Globally, populations of many scavenging seabirds have grown in recent years (e.g., Lloyd et al. 1991) and it is likely that some species have significantly benefitted from fishery discards (e.g., Furness & Barrett 1985, ICES 2005). However, population growth in scavenging seabirds can lead to displacement of other species because of limited suitable breeding habitat (Howes & Montevecchi 1993). For example, in Europe, many tern species have been displaced by larger gull species (Theissen 1986, Becker & Erdelen 1986). This has led in many instances to the culling of the large gulls in order to allow terns to return to their original nesting sites (Wanless 1988, Wanless et al. 1996).

13.2 WHAT CAUSES TROPHIC AND ECOSYSTEM-LEVEL EFFECTS?

As can be seen in the examples given so far, trophic and ecosystem-level effects in marine systems can be caused by a variety of factors, often acting simultaneously. These factors are often called stressors. Stress in this context refers to physical, chemical and biological constraints on the productivity of species, their interdependencies, and on the structure and function of the ecosystem. Stressors can act over various spatial scales (from local to basin-scale) and various time scales (from days to decadal). Stressors can be natural environmental factors or they may result from the activities of humans. Trophic and ecosystem-level effects can occur because of fishing, because of environmental factors entirely disconnected to fishing (especially related to climate variability/change) or by a combination of fishing and environmental variability/change acting together (Mackinson et al. 2009, Frank et al. 2007, Schiermeier 2004, Schiel 2013). Trophic and system-level effects can also result from outbreaks of disease (Cobb & Castro 2006, Freeman & MacDiarmid 2009, Shields 2011), from the arrival of non-indigenous invasive species (Mead et al. 2013) and from eutrophication in estuarine ecosystems (Daskalov et al. 2007, Oguz & Gilbert 2007, Osterblom et al. 2007, Möllmann et al. 2008). Some of these causes of trophic and ecosystem-level effects are discussed further below.

13.2.1 ENVIRONMENTAL-DRIVEN CHANGE

Marine ecosystem are intimately linked to environmental (climate) forcing (Fasham et al. 2001, Schiermeier 2004, Frank et al. 2007, Mackinson et al. 2009). Variability of climate forcing of the ocean occurs on a wide range of time scales from seasonal periods, to 1–3 year oscillating but erratic periods, to decadal aperiodic variability at 5–50 years, to centennial and longer periods, and can include sudden, large-scale shifts in environmental forcing (Overland et al. 2010). Climate trends (such as due to global warming) are defined as changes that are not cyclical or seasonal and exist over a relatively long period (more than decadal).

There are many examples internationally of trophic and ecosystem-level effects occurring as a result of environmental change affecting the bottom of the food web (Mackinson et al. 2009, Frank et al. 2007, Schiermeier 2004). For example, during the 1980s, the North Sea experienced a change in hydro-climatic forcing that caused a rapid, temperature-driven ecosystem shift (Beaugrand & Ibanez 2004). This change in sea surface temperature (SST) altered the plankton and negatively affected the recruitment of cod (Beaugrand & Reid 2003, Heath 2005). Changes in the North Sea plankton, following the ecosystem shift, included an increase in microalgae (Kirby et al. 2008), a change in the composition and abundance of zooplankton (Beaugrand et al. 2002), increases in the frequency of jellyfish (Kirby et al. 2009), increases in the abundance of decapod and echinoderm larvae, and a decrease in bivalve larvae (Kirby et al. 2008). Another example of bottom-up effects on upper-trophic-level marine predators is the abrupt decline in local primary and secondary production caused by El Niño/Southern Oscillation (ENSO) events in eastern Pacific boundary currents (Barber & Chavez 1983, Pearcy et al. 1985, Arcos et al. 2001, Hollowed et al. 2001). During these ENSO events, the production of small pelagic fishes can be drastically reduced (Barber & Chavez 1983, Rothschild 1994), and predatory fish, seabirds and pinnipeds, which are dependent on these small pelagic fish have been shown to shift their distributions, suffer reduced productivity, and have increased rates of mortality (Trillmich et al. 1991, Jahncke et al. 2004).

13.2.2 FISHERIES-DRIVEN CHANGE

To some degree, trophic effects will always arise as a consequence of fisheries. As well as reducing the overall abundance of fish, fishing usually reduces the average size of fish in harvested communities and can change the mix of species in a fish community (Pope & Knights 1982, Pope et al. 1987, Dayton et al. 1995). Fishing also has effects beyond changes to the abundance and population structure of target and bycatch species, including (a) the introduction of discarded bycatch/offal/bait into the ecosystem, (b) the alteration of fish behaviour (and potentially genetic make-up) as a result of fishing, and (c) the modification of the benthos by fishing gear. Fishing will certainly lead to changes (of greater or lesser magnitude) in predation pressure on prey species. Marine ecosystems seem to be remarkably resilient to even quite large trophic changes of this kind, but there are clearly limits to this resilience. Virtually all well-documented regime shifts seem to have been initiated from large-scale climate or oceanographic changes rather than excessive fishing pressure. In some cases however, ecosystem-level changes (regime shifts) have been demonstrated empirically to occur in very highly impacted (highly overfished/collapsed) systems as a result principally of trophic effects (Estes & Duggins 1995, Daskalov et al. 2007). For example, the round sardinella (*Sardinella aurita*) stock off West Africa collapsed in the 1970s following exceptionally high catches made possible by oceanographic changes (Bakun & Weeks 2006). This collapse resulted in a substantial and widespread outbreak of grey triggerfish (*Balistes capriscus*), which lasted through the 1970s and 1980s until the sardinella population rebuilt. At that point, grey triggerfish essentially disappeared from the ecosystem again. It seems possible that the juvenile triggerfish, being pelagic plankton feeders, took advantage of the collapse of the sardinella population to temporarily replace it as the dominant nektonic zooplanktivore of the ecosystem through one or more trophic effects. For example: (1) the sardinella collapse may have led to increased zooplanktonic food resources and hence accelerated the production rate of triggerfish; (2) the sardinella collapse may have promoted increased recruitment of triggerfish by reduced predation on their eggs and larvae (Bakun & Weeks 2006).

13.2.3 COMBINED EFFECTS OF FISHING AND ENVIRONMENTAL VARIABILITY/CHANGE

Although there have been few unequivocal empirical demonstrations of large-scale trophic and system-level effects arising solely from fishing, very many studies have pointed to the potential of fishing to lead to trophic and ecosystem-level effects in concert with other factors, such as environmental variability and change (e.g., Winder & Schindler 2004, Brierley & Kingsford 2009, Kirby et al. 2009, Perry et al. 2010). The effects of fishing that may lead to reduced ecosystem resilience (see

Table 13.1 for definition of 'ecosystem resilience') include:

- Alteration of demographic structure. Size-selective removal truncates the population's age structure and lowers the buffering capacity of the population (its ability to withstand long periods of environmental conditions that are adverse for recruitment). This leads to the prediction that the relative importance of recruitment variability will be greater in exploited populations as has been observed in a comparison between exploited and unexploited fishes in the California Current Ecosystem (Hsieh et al. 2006).
- Alteration of spatial structure. The spatial structures of marine fish populations can encompass a wide range of configurations, including patchy populations, networks, and meta-populations (Kritzer & Sale 2004). Removal or curtailment of population spatial structure by fishing is likely to increase the sensitivity of the overall population to climate fluctuations at inter-annual to multi-decadal scales (e.g., Ottersen et al. 2006).
- Alteration of life-history traits. Perry et al. (2010) suggest that fishing would be likely to accelerate the response of populations to climate forcing by providing selective pressure to decrease growth rates and decrease age-at-maturity (Law 2000, de Roos et al. 2006).
- Alteration of habitat structure. Changes to benthic habitat by the direct effects of fishing may lead to a reduction in ecosystem resilience (Thrush & Dayton 2002).
- Alteration of ecosystem trophic structure. Theoretically, ecosystems under intense exploitation are likely to evolve towards stronger bottom-up control (Figure 13). Exploitation leads to a decrease in stock sizes of piscine predators, which may (a) reverse the control structure in top-down ecosystems to bottom-up control, and (b) amplify the control in already bottom-up controlled ecosystems. Multiple weak interactions and generalist predators may stabilise ecosystems by dampening oscillations caused by strongly interacting species (Shin & Cury 2001, Polunin & Pinnegar 2002, Rooney et al. 2006, McCann & Rooney 2009, Johnson et al. 2014) and by preferentially consuming competitively dominant prey species (Brose et al. 2005). Changes to trophic structure by fishing are hence predicted to increase ecosystem variability and reduce resilience (Jackson et al. 2001, Perry et al. 2010).

Theoretically therefore, fishing is predicted to strengthen the relation between oceanographic forcing and ecosystem variability and hence reduce ecosystem resilience. There are limited real-world, empirical examples of this. For example, the regime shifts of the North Sea and central Baltic Sea are considered to have been driven by the combined and synergistic effects of intense fishing and climate variability (Weijerman et al. 2005, Möllmann et al. 2009). Using a 47-year time series, Kirby & Beaugrand (2009) showed that the effects of temperature can be magnified by propagation through indirect pathways in the food web. This 'trophic amplification' can intensify the effect of environmental variability, potentially leading to a new stable or unstable ecosystem state (Scheffer & Carpenter 2003, Muradian 2001, Taylor 2002, Hsieh et al. 2005). Elsewhere, Ottersen et al. (2006) analysed the Arcto-Norwegian cod stock in the Barents Sea over the last 60 years and found evidence of a strengthening of the climate-cod recruitment link during the last decades.

Table 13.1: Ecosystem resilience.

Fishing can affect ecosystem resilience, the capacity of an ecosystem to absorb disturbance and reorganise while undergoing change so as to retain essentially the same function, structure, identity, and feedbacks (Pimm 1982, Holling 1973, Cohen et al. 1990, Walker et al. 2004). Three measures of ecosystem resilience have been identified:

- Does the ecosystem retain essentially the same function, structure, identity, and feedbacks after perturbation as before (Walker et al. 2004)?
- Do perturbations to one part of the ecosystem spread out and affect biota across many trophic levels or remain localised (i.e., are ecosystem-level changes likely)?
- How long does it take a food web to return to its original configuration when perturbed? Stable (resilient) food webs can absorb more perturbation without undergoing wholesale reorganisation, tend to have low tendency for ecosystem-level trophic cascades (food web perturbations remain local) and have short return times (Walker et al. 2004).

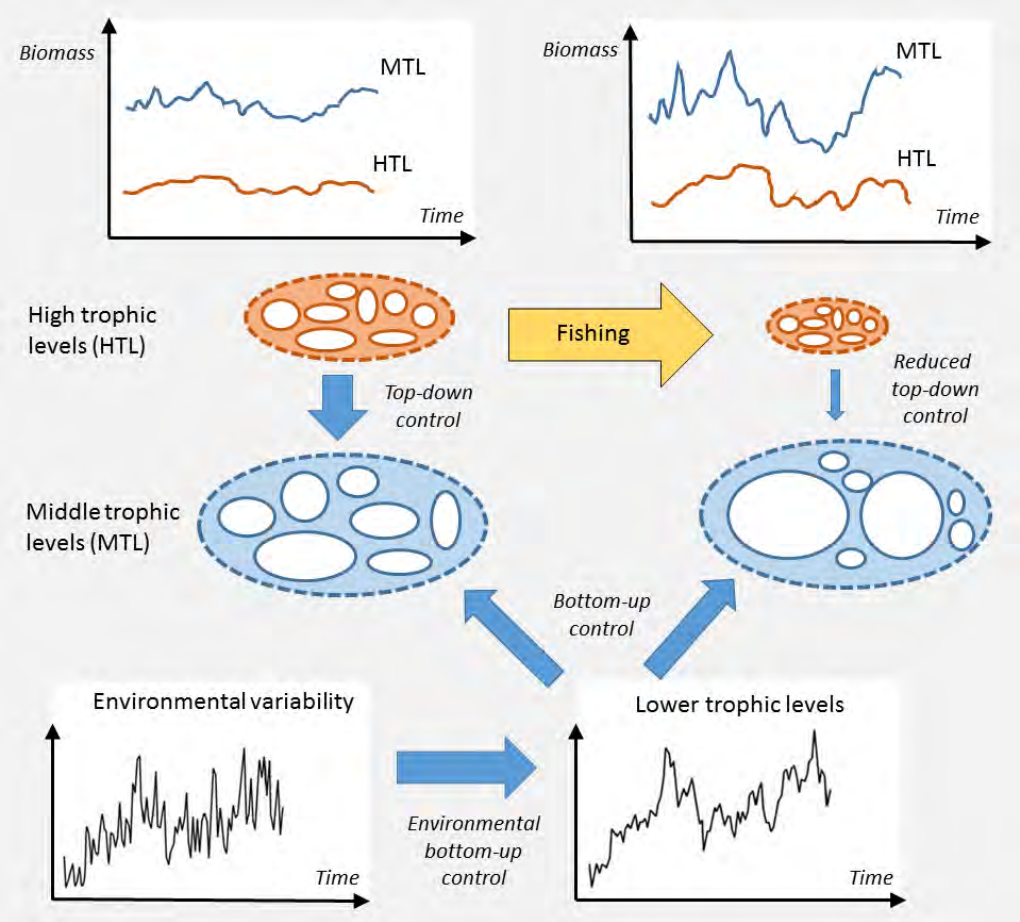


Figure 13.1: Schematic illustrating expected responses of unexploited and exploited marine ecosystems to climate forcing. Left side shows an unexploited ecosystem with multiple high trophic level (HTL) species that have relatively large abundances supported by several mid-trophic level (MTL) species, and how their aggregate biomasses vary through time (top left) in response to environmental variability acting on the lower food web. The right side illustrates how that same climate forcing is experienced by an ecosystem which has been exploited (top right graph). The abundances of the high trophic level species have decreased due to fishing, weakening the top-down control on the MTL. This is hypothesised to make the mid-trophic level groups less even causing their aggregate biomass to track the environmental forcing more closely (after Perry et al. 2010).

13.3 WHAT TYPES OF ECOSYSTEM ARE LIKELY TO BE MOST AFFECTED?

13.3.1 GLOBAL UNDERSTANDING

The scale and significance of trophic and ecosystem-level effects depend on the particular characteristics of the ecosystem as well as on the drivers of change (Pace et al. 1999, Brose et al. 2005, Pascual & Dunne 2006, Brander 2010, Jennings & Brander 2010). Ecosystems appear to be prime examples of complex adaptive systems (Levin 1998, 1999); ecosystems typically have non-linear dynamics, with thresholds (also called tipping-points) and positive and negative feedback loops (Hsieh et al. 2005). The complex behaviour of ecosystems over a wide range of time and space scales coupled with the myriad nature of stressors means that it is hard to forecast the response of ecosystems or establish quantitative estimates of tipping-points to guide management.

A number of multispecies or ecosystem models have been developed that can be used to investigate the potential for trophic and ecosystem-level effects in ecosystems (Plagányi 2007, Plagányi et al. 2014). These include Ecopath with EcoSim (EwE; Christensen & Walters 2004), Atlantis (Fulton et al. 2004, 2005), OSMOSE (Shin et al. 2004, Travers et al. 2009) and a range of models of intermediate complexity (MICE; Plagányi et al. 2014). Multispecies and ecosystem models can provide useful strategic insights for fishery and resource managers (Plagányi 2007, Fulton et al. 2005, Smith et al. 2011). However, there are often differences in model predictions about ecosystem consequences (or lack thereof) of fishing, especially in ecosystem-scale models, so model outputs need to be used cautiously for tactical decisions (Smith et al. 2011). MICE-models (where only part of the ecosystem is modelled) are likely to provide more robust guidance for tactical decision-making (Plagányi et al. 2014).

There have also been attempts to use knowledge of the structure of the food web to suggest types of behaviour and response to fishing and other changes as an alternative to dynamic ecosystem models (Ulanowicz & Puccia 1990, Libralato et al. 2006, Pinkerton & Bradford-Grieve 2014). Rice (2001) concluded that trophic and ecosystem-level effects of fishing depend on the overall type of ecosystem forcing structure. Three patterns of ecosystem forcing structure have been described: (a) top-down forced, (b) bottom-up forced, or (c) forced from the middle outwards or wasp-waisted (13.2). These patterns of ecosystem forcing have been the focus of hundreds of research articles. These three patterns should be considered as modes of forcing (rather like principal components); most real ecosystems will be a mixture of these types of forcing that may change over time (Rice 2001). Indeed, Pace et al. (1999) cautions that 'although there is some descriptive value in the use of top-down or bottom-up control, this motif also creates a false dichotomy.' Nevertheless, identifying dominant patterns of ecosystem behaviour may help to predict or explain the types of trophic and ecosystem-level behaviour resulting from the combined effects of fisheries harvesting, climate variability/change and other human activities (Rice 2001). For example, Pinsky et al. (2011) uncovered a high incidence of fisheries collapse among small, short-lived, middle trophic-level species of a type that are often the wasp-waist of the ecosystem. Even though short-lived species may recover quickly from excessive fishing mortality (Hutchings 2000), changes to them can have substantial impacts on the food web (Duffy 1983, Frederiksen et al. 2004, Crawford 2007).

Table 13.2: Overall types of ecosystem forcing. [Continued on next page]

Bottom-up ecosystem forcing	If the ecosystem-level properties (i.e., across organisms at many trophic levels) respond strongly to changes in the environment (e.g., oceanography, water column structure), the ecosystem is said to show strong bottom-up forcing. There are many examples internationally of trophic and ecosystem-level effects occurring as a result of environmental changes at the bottom of the food web (Mackinson et al. 2009, Frank et al. 2007, Schiermeier 2004).

Table 13.3 [Continued]:

Top-down ecosystem forcing	An ecosystem is said to show strong top-down forcing if it responds strongly to changes in the abundance of top predators (seabirds, marine mammals, high trophic level fishes). Understanding of how predators shape marine ecosystems has arisen largely from experimental studies where the effect of predation is controlled either by removing predators or introducing them to the ecosystem under study, usually in the intertidal or nearshore subtidal zone (Hunt & McKinnell 2006 and references therein). In the open ocean, increases in prey populations upon the removal of their predators (e.g., by fisheries) have been taken as evidence of top-down limitation (e.g., Furness 2002, Worm & Myers 2003, Frank et al. 2005). Other evidence of top-down regulation in a marine ecosystem appears where predators are abundant at one site, but largely absent from a similar, nearby site. For example, Birt et al. (1987) found that small flatfish populations were depressed in a bay in Newfoundland that was frequented by cormorants compared to a bay that was located farther from the colony. In general, top-down ecosystem forcing is predicted to be stronger in aquatic than terrestrial ecosystems, and strongest in marine ecosystems where the predators are large and mobile with high metabolic rate, where prey species are long-lived, functional predator diversity is low, and predator intra-guild predation is weak or absent (Shurin et al. 2002, Borer et al. 2005, Heithaus et al. 2008).
Middle-out forced (wasp-waisted) ecosystem	Wasp-waist control of energy flow in marine ecosystems occurs when one or a very few species have a substantial influence on the flow of energy through the mid-trophic levels. The term has most frequently been applied to the role of small pelagic fishes that transfer energy from the plankton to larger predatory fish, seabirds and marine mammals (Rice 1995, Cury et al. 2000, 2004, Bakun 2004, 2006). Ecosystems with wasp-waist control are typically coastal, highly productive systems with relatively short food chains. However, waist-controlled ecosystems also include capelin in North Atlantic ecosystems (Lilly 1993, Bogstad & Mehl 1997, Leggett et al. 1984, Taggert & Leggett 1987, MacKenzie & Leggett 1991, Fossum 1992), krill in the Antarctic (Murphy et al. 1998) and, *Calanus* sp., when functioning as a 'gatekeeper' (sensu Steele 1998). When the species at the waist declines abruptly, predators often cannot compensate, at least fully, and suffer reduced growth, survivorship, and reproduction (Mehl & Sunnana 1991, Kjesbu et al. 1998, Dutil & Lambert 2000). Predators may control the wasp-waist when they are at intermediate population sizes (Bakun 2006). At other times, year-class strengths of species at the waist demonstrate strong, direct effects of environmental forcing. Wasp-waisted ecosystems typically follow from: (1) a food web containing a highly influential intermediate node that has a strong environmental signal in recruitment (Rice 2001) and/or (2) middle-trophic level fishery.

13.3.2 NEW ZEALAND

13.3.2.1 BOTTOM-UP FORCING

A New Zealand example of bottom-up forcing is the driver of mussel (*Perna canaliculus*) yield in Pelorus Sound in northern South Island. Though this example is from aquaculture, it is likely to also apply to wild mussels. Zeldis et al. (2008) correlated physical, chemical and biological data collected within a nine-year time series. Starting in early 1999, farm production in the sound declined by about 25% in terms of per-capita meat yield, followed by yield recovery through to 2002. These changes resulted in substantial economic impacts within the industry. Over-grazing by mussels (i.e., top-down effects on mussel food availability) did not explain the yield minimum. Instead, bottom-up (environmental) effects of nitrogen supply from

oceanic and river sources drove the variation by affecting the abundance of seston[5] for the filter-feeding mussels. A subsequent study (Zeldis et al. 2013) provided quantitative models for Pelorus Sound mussel per-capita meat yield and elucidated the underlying oceanographic mechanisms. Yield was best predicted using biological variables, including the concentration of seston, based on measurements made next to the mussel farms, but it was also predictable using only physical variables that index large-scale environmental processes (Southern Oscillation Index, along-shelf winds, sea surface temperature and river flow).

13.3.2.2 TOP-DOWN FORCING

In moderately exposed coastal marine reserves in north-eastern New Zealand, predation by recovering populations of snapper (*Pagrus auratus*) and spiny lobsters (*Jasus edwardsii*) have gradually decreased the abundance of the grazing sea urchin (*Evechinus chloroticus*) and allowed turfing algae and kelp (*Ecklonia radiata*) to replace urchin grazed rock flats (Babcock et al. 1999, Shears & Babcock 2002, 2003). This is indicative of top-down forcing in the ecosystem. In adjacent areas which are heavily fished there are more urchins, and areas free of turfing algae and kelp are common (Shears et al. 2008). It seems that the occurrence of this trophic cascade varies at local and regional scales in relation to abiotic factors, implying some interplay with larger-scale bottom-up forcing (Shears et al. 2008).

A long-term study of changes to the ecosystem of the Hauraki Gulf region developed five balanced, quantitative models of the food web of the region (MPI project ZBD200505: Pinkerton 2012): (1) present day; (2) AD 1950, just prior to onset of industrial-scale fishing; (3) AD 1790, before European whaling and sealing; (4) AD 1500, early Maori settlement phase; (5) AD 1000, before human settlement in New Zealand. These models were used to estimate the strengths of trophic connections between different groups of organisms based on single-step and multiple step measures of trophic importance (Ulanowicz & Puccia 1990, Libralato et al. 2006). Before humans arrived in New Zealand, the models suggest that cetaceans and fur seals/sea lions were the most trophically important groups in the Hauraki Gulf ecosystem, implying the potential for strong top-down ecosystem control. With the extirpation[6] of seals/sea lions from the Hauraki Gulf ecosystem before the arrival of Europeans and the reduction in the abundance of cetaceans following European arrival, the trophic importance of these air-breathing predators drastically reduced. The trophic importance of other predators in the models of the Hauraki Gulf ecosystem also reduced over time as a result of human harvesting (rock lobsters and sharks especially) suggesting a transition to a more bottom-up controlled system.

13.3.2.3 MIDDLE-OUT (WASP-WAIST) FORCING

Research into deepwater ecosystems in the New Zealand EEZ is most advanced in the Chatham Rise region. Elevated primary production here is due to the convergence of subantarctic and subtropical water (Bradford-Grieve et al. 1997, Boyd et al. 1999, Murphy et al. 2001, Sutton 2001) and supports valuable deepwater fisheries, an unusually rich benthic ecosystem (Probert et al. 1996, McKnight & Probert 1997, Bowden 2011), and large seabird populations (Taylor 2000a, 2000b). Ecosystem modelling of the Chatham Rise food web has been underway since 2006, the most recent version being Pinkerton (2013) (Figure 13.2). Trophic impact matrices (Ulanowicz & Puccia 1990, Libralato et al. 2006) were calculated from the balanced model to investigate patterns of trophic interactions. Middle trophic level groups, especially small demersal fishes and mesozooplankton, had some of the highest trophic importances amongst consumers. Mesopelagic fishes, hoki, and arthropods (benthic prawns and shrimps) also had high trophic importances (Pinkerton 2013). These patterns of trophic importance were robust to uncertainties in the model parameterisation and balancing (Pinkerton 2014b). These results suggest some degree of middle-out control in the system, though the number and function diversity of these groups is higher than in other systems characterised in this way.

[5] Organisms and non-living matter swimming or floating in a water body.

[6] Made locally extinct.

13.4 OVER WHAT SPATIAL SCALES DO TROPHIC AND ECOSYSTEM-LEVEL CHANGE OCCUR?

13.4.1 GLOBAL UNDERSTANDING

Delineating ecosystems is an important first step towards evaluating trophic and ecosystem-level effects of fishing. There are not usually clear spatial boundaries between different ecosystems. Instead, different parts of ecosystems vary on different spatial scales; higher trophic-level organisms usually move over a greater spatial extent than lower trophic-level organisms. For example, some seabirds and marine mammals may move large distances seasonally and move between different ecosystems. In contrast, most phytoplankton, smaller zooplankton and most benthic invertebrates will live and die within a few kilometres. Some fish move long distances, but others remain in a small area all their lives (e.g., on a reef). Marine ecosystems should hence be viewed as an interlocking matrix of the life ranges of different organisms. As such, it is difficult to unambiguously separate different ecosystems but a number of approaches have been developed to do so. These include: (a) defining ecosystems on the basis of their physical properties, either using a priori thresholds (e.g., fixed depth ranges) or by multivariate clustering of physical properties (Snelder et al. 2005, Grant et al. 2006); (b) using maps of species occurrence to map biological assemblages (e.g., Leathwick et al. 2006); (c) relating community composition to environmental variables (e.g., generalised dissimilarity analysis; Ridgeway 2006, Leathwick et al. 2009) and using these relationships to extrapolate spatially.

13.4.2 NEW ZEALAND

The importance of spatial scale in the study of the ecosystem effects of fisheries has been recognised in New Zealand (e.g., Leathwick et al. 2006, 2009). In their assessment of the New Zealand hoki fishery for the Marine Stewardship Council (MSC), Akroyd & Pierre (2013) noted that there is currently no specific definition of 'regional effects' but MSC is working on adding clarity to the definition of regions and bioregions as part of the work on their current benthic impacts project in recognition that some areas are more vulnerable to impact than others.

A number of approaches have been developed in New Zealand to identify or describe ecosystem types:

- MacDiarmid et al. (2012) identified 62 distinct marine habitat types occurring within New Zealand's Territorial Sea and EEZ as part of an assessment of anthropogenic threats to New Zealand marine habitats. The approach taken by MacDiarmid et al. (2012) was to build on Halpern et al.'s (2007) list of marine habitats used in a global assessment of anthropogenic impacts on the global marine environment.
- New Zealand's Department of Conservation, jointly with MPI, have used a marine habitat classification system based on four depth intervals (intertidal, 0–30 m, 30–200 m, more than 200 m), seven substrate classes (mud, sand, gravel, undefined substrate, mixed sediment and rock, rock, and biogenic), and three exposure categories (exposed, moderate, sheltered). This habitat classification was used to define 58 habitats in the Territorial Sea alone in order to meet the needs of biodiversity conservation (DOC-MPI 2011).
- New Zealand Marine Environment Classification (MEC; Snelder et al. 2005). The MEC is a physically based classification, determined using multivariate clustering of several spatially explicit data layers that describe the physical environment (including depth, slope, orbital velocity at the sea floor, mean solar radiation, SST amplitude, SST gradient, winter SST, mean tidal current velocity). Large biological datasets were used to tune the classification so that the physically based classes maximised discrimination of variation in biological composition at various levels of classification detail. The classification was not optimised for a specific ecosystem component (e.g., fish communities or individual species) but sought to provide a general classification that had relevance to a broad range of biological groups. Depending on user requirements the MEC can provide two to 270 classes of classification.
- Leathwick et al. (2006) demonstrated how spatial analysis using boosted regression trees could provide distribution maps of over 100 species of demersal fish. Fish were chosen as there were good quality distributional data available from a series of scientific trawl surveys in deep waters. The overall approach used by Leathwick et al. (2006) was to fit statistical models relating the distributions of 122 fish species to a set of environmental variables,

with the latter chosen for their functional relevance.

- A Benthic-optimised Marine Environment Classification (BOMEC; Leathwick et al. 2009) was developed specifically to identify New Zealand benthic bioregions that can be considered to be ecologically distinct to some degree. BOMEC was developed by combining data on the benthic community (made up of over 100 demersal fish species, and seven groups of invertebrates: asteroids, bryozoans, foraminifera, octocorals, polychaetes, scleractinian corals, sponges), and environmental data including sediment type. A multivariate technique for fitting community compositions to environmental data, Generalised Dissimilarity Analysis, was used (Leathwick et al. 2009). BOMEC is restricted to depths less than 3000 m where reasonable amounts of scientific sampling have been conducted (Leathwick et al. 2009).
- The Ocean Survey 20/20 Chatham-Challenger biotic habitat classification (Hewitt et al. 2011) used benthic invertebrate and environmental data from the Chatham Rise and Challenger to delineate ecosystems in terms of their community and biogenic habitat associations.
- Sharp et al. (2007) summarised lessons learned from New Zealand's bioregionalisation experience for CCAMLR. The main conclusion was that bioregionalisations based on simple clustering of physical variables are likely to perform poorly in terms of separating assemblages of species (communities or ecosystems); measurements of the actual distributions and abundances of key organisms are needed to use physical environmental data to delineate bioregions effectively.

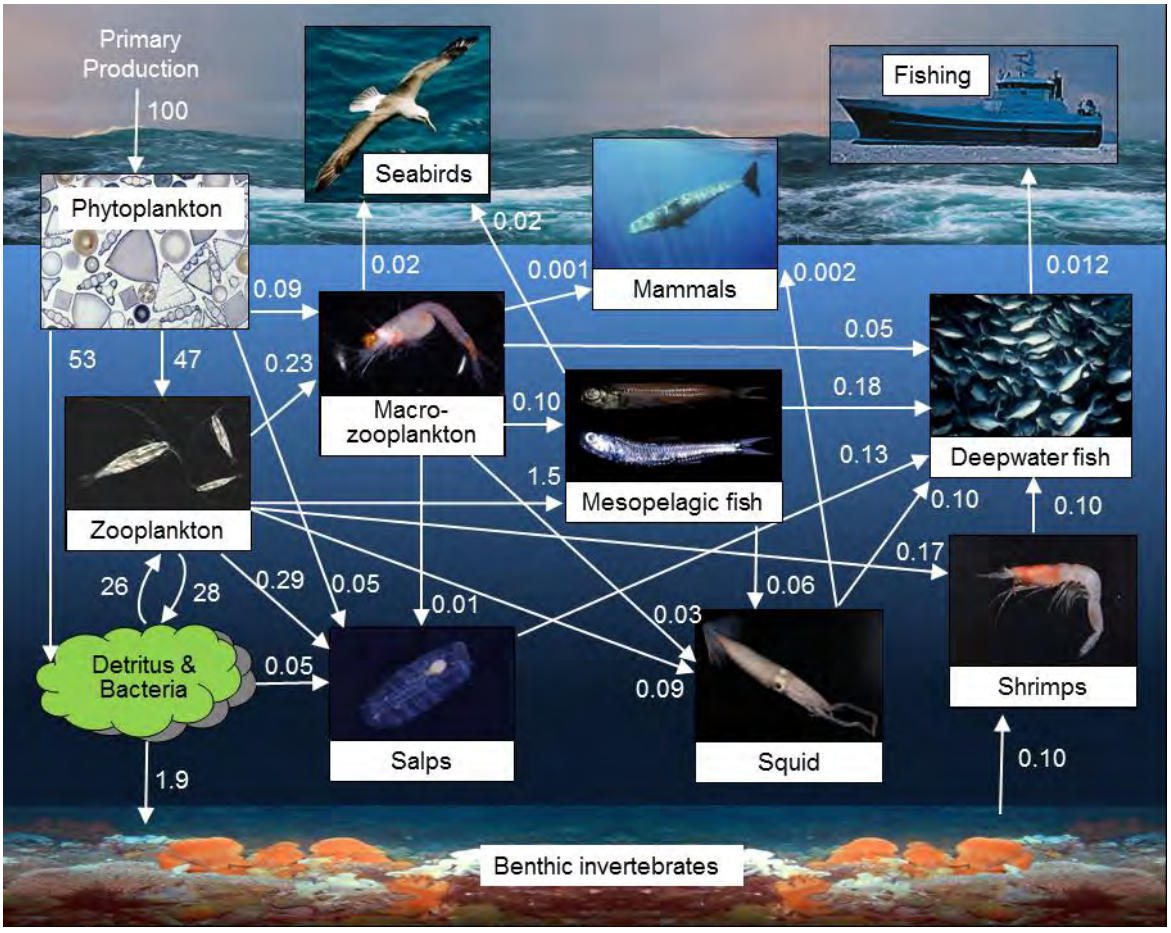


Figure 13.2: Simplified trophic model of the Chatham Rise, New Zealand (based on Pinkerton 2013). The growth of phytoplankton generates organic matter that is the fuel for the marine ecosystem. Figures show the annual flow of energy through unit area of the food web normalised to a net primary productivity (NPP) of100, based on an equilibrium mass-balance model (similar to Ecopath).

13.5 HOW CAN TROPHIC AND ECOSYSTEM-LEVEL EFFECTS BE DETECTED?

13.5.1 GLOBAL UNDERSTANDING

There has been increasing recognition over the last two decades that time series are essential to detect and potentially understand a trophic or ecosystem-level change in marine ecosystems. This has led to a high level of interest in the development and interpretation of indicators of the marine environment and its ecosystems. A huge number (more than 300) of marine ecosystem indicators are in use or proposed around the world (Cury et al. 2005, Rochet & Rice 2005, Rice 2003), with consensus that a suite of indicators is needed to monitor and understand the impact of human activities on marine ecosystems (Cury & Christensen 2005, Rice & Rochet 2005). Give the multi-trophic nature of ecosystem-level effects, indicators are needed that span the ecosystem, including primary producers, the microbial system, middle trophic levels, fish communities, the benthic community and top predators. A summary of some recommended indicators is given below.

13.5.1.1 MARINE PRIMARY PRODUCTION

The growth of phytoplankton in the upper layers of the ocean provides the vast majority of the energy that fuels marine ecosystems, and most fisheries, worldwide. Only in some (predominantly coastal) areas are other primary producers important: macroalgae (seaweed), seagrass, mangroves, epiphytes, autotrophic periphytes, microphytobenthos and chemosynthesisers. Light, temperature, and nutrient concentrations are major factors controlling net [7] primary production (NPP) by phytoplankton growth in the ocean (Parsons et al. 1977, Arrigo 2005). NPP can be measured accurately from ships (typically using radioactive carbon incubations), but because of the high spatial and temporal variability of NPP, ship-based sampling is not adequate for monitoring. Instead, remotely sensed data from sensors on Earth-observing satellites are typically used to estimate NPP. There are significant differences between different methods of estimating NPP from satellite data (Campbell et al. 2002). Often, the concentration of chlorophyll-a, the ubiquitous pigment in phytoplankton, is used as a proxy for phytoplankton biomass and NPP, because this can be measured remotely with better accuracy than NPP using ocean colour satellite sensors.

13.5.1.2 LOWER FOOD WEB (MICROBIAL SYSTEM)

Rice (2001) notes that processes that make large alterations to the allocation of production between the microbial loop, benthic detrital pathways and mesopelagic consumers may have much more impact on the dynamics of higher trophic levels than processes that alter NPP. More recently, Friedland et al. (2012) examined the relationships between NPP, fisheries yields, and parameters describing the transfer of organic matter through 52 large marine ecosystems and found that chlorophyll-a concentration, the particle-export ratio (p-ratio: the proportion of NPP exported from the surface layer of the ocean) and the ratio of mesozooplankton productivity to NPP (z-ratio) were all significantly related to fisheries yields. Stock & Dunne (2010) suggest that a warmer ocean will lead to lower z-ratio (less mesozooplankton for a given NPP) and Friedland et al. (2012) show that lower z-ratios correspond to lower fisheries yields at basin scales.

13.5.1.3 MIDDLE TROPHIC LEVELS

Small mesopelagic [8] and hyperbenthic [9] organisms are an important part of marine ecosystems. They act as the link between the microbial/planktonic system and larger predators such as seabirds, marine mammals, and larger fish. These 'middle trophic level' organisms are diverse, and include hard-bodied crustaceans (such as copepods, euphausiids, amphipods, prawns and shrimps), 'jellies' (such as jellyfish and salps), cephalopods (squids and octopods), and a range of small fishes (including juveniles of larger species) living in the water column (especially myctophids or lanternfishes) or near the seabed. These

[7] 'Net' means after allowing for phytoplankton respiration.
[8] 'Mesopelagic': inhabiting the intermediate depths of the sea, between about 200 and 1000 m down.
[9] 'Hyperbenthic': ecologically associated with the seabed, but living for some time in the lower water column.

species are likely to be affected both by fishing, which may reduce top-down predation control, and by climate-driven changes in lower trophic food web components (Frank et al. 2007, Richardson 2008). Middle trophic level species have a key role in ocean ecology (e.g., Banse 1995, Marine Zooplankton Colloquium 2 2001, Smetacek et al. 2004, Pinkerton 2013). Studying these middle trophic level organisms is challenging: they are typically diverse, with varied and complex life histories, can be hard to capture, and have abundances that vary over a wide range of space and time scales. Consequently, the factors that affect their dynamics are generally poorly understood. Two methods have been used for monitoring middle trophic levels. First, in other parts of the world, long time series of measurements of the zooplankton community by the Continuous Plankton Recorder (CPR) has demonstrated change in marine ecosystem (Beaugrand et al. 2002, Aebischer et al. 1990, Reid et al. 1998, Beare & McKenzie 1999), and been recommended as an effective way of monitoring the state of pelagic ecosystems (Beaugrand 2005). Second, multifrequency acoustics have been used to monitor abundances of mesopelagics over extended time and space scales (McClatchie & Dunford 2003, O'Driscoll et al. 2009, Trenkel & Berger 2013).

13.5.1.4 DEMERSAL FISH COMMUNITIES

Most of the international effort on developing ecosystem indicators have focused on those for the demersal fish community, usually based on commercial landings data or, less commonly, on catch data from fisheries surveys. Consequently, very many indicators have been proposed – a selection is discussed below.

- Marine Trophic Index: MTI is the mean trophic level of fisheries landings (Pauly & Watson 2005) and was recently recommended for use with commercial catch data by the United Nations Biodiversity Convention as a widely applicable and cost-effective indicator for monitoring reductions in biodiversity loss in marine ecosystems (CBD 2004). A gradual decline in trophic level of about 0.2 since industrialised fishing began has been observed in many finfish fisheries around the world (Pauly et al. 1998a, Christensen et al. 2003), ascribed to fisheries targeting high trophic level species and moving on to lower trophic level species as these large species are depleted, a change called 'fishing down the food web'. Essington et al. (2006) noted that 'fishing through the food web', where higher trophic level fish landings are maintained but catch of lower trophic level species increases over time, may occur more often. MTI calculated from total commercial catch will vary with changes in the mix of species targeted by different fisheries over time, the relative importance of different fisheries sectors (e.g., finfish versus invertebrate fisheries), how much of the catch is reported, the quality of identification of species, and for other reasons not necessarily associated with effects of fishing (Caddy et al. 1998, Pauly et al. 1998b, Tuck et al. 2009, Branch et al. 2010). As such, MTI based on scientific surveys is likely to be a better indicator of change in fish communities (Branch et al. 2010).
- Species-based indicators: Many indices of diversity have been applied to fish communities (e.g., Peet 1974, Warwick & Clarke 1995, Bianchi et al. 2000, Greenstreet & Rogers 2006). These diversity indices are joint constructs of how many species are present (richness), and how similar their abundances are (evenness). Some indices give additional emphasis to the most important species in a community (dominance). Measures vary in the relative weight given to each of these factors, and on the metric used for similarity between species (e.g., by including a measure of taxonomic distinctiveness or not; Warwick & Clarke 1995). Fishing rarely causes large-scale extirpation so that measures of total species richness are likely to be less sensitive to change in trophic or ecosystem-level properties than measures of evenness. Different measures of evenness respond variously to fishing; they can increase, reduce or be unaffected by fishing depending on the initial characteristics of the ecosystem. A community

initially dominated by k-selected[10] species would be expected to become more even and show increasing diversity metrics due to fishing; fishing would be expected to allow the faster growing (initially minor species) to increase at the expense of the slower growing (initially dominant) species. In contrast, diversity and evenness metrics may be expected to decrease after fishing if the ecosystem were originally dominated by r-selected[11] species.

- Functional group based indicators: Changes to the relative abundance of different functional groups in an ecosystem can indicate trophic or ecosystem-level changes (Fulton et al. 2005, Methratta & Link 2007, Shannon et al. 2009). Functional groups can be based on various descriptors of ecological niche, such as position in the water column (e.g., pelagic, demersal, benthic), trophic guild/feeding type (e.g., piscivore, pelagic invertebrate feeder, benthic feeder, scavenger), taxonomy (e.g., elasmobranch, gadoid, macrourid), or a combination of multiple ecological and life-history traits (Methratta & Link 2007), which can be combined to suggest high or low resilience (Tuck et al. 2009). A simple and commonly used index is the proportion of piscivorous fish to all fish caught. As piscivorous fish tend to be disproportionately impacted by fishing (Caddy & Garibaldi 2000), their relative abundance in fish assemblages is a measure of ecosystem state and may reveal a trophic or system-level impact of fishing.
- Size based indicators: Marine trophic processes tend to be strongly structured by size (Badalamenti et al. 2002, Jennings et al. 2002). Fishing may lead to substantial modifications in the size structure of exploited populations because (a) high-value, generally larger species are targeted by fisheries, (b) fishing gears are size selective, often designed to catch larger fish and let smaller ones escape, (c) the cumulative effect of fishing (over the life of a cohort) leads to fewer older (larger) fish, and (d) long-lived species tend to be affected more as they have lower potential rates of increase. Several size-based metrics have been used to detect trophic and ecosystem-level changes (e.g., Murawski & Idoine 1992, Pope et al. 1987, Pope & Knights 1982, Rice & Gislason 1996). Size-based indicators can be applied at a species or community level. Applied to a given species, possible size-based indicators include: (a) mean length at age; (b) condition (weight at length; e.g., Winters & Wheeler 1994); (c) proportion of large fish; and (d) mean length at maturity in the population. Size-based methods at the community level include: (a) mean length in the community; (b) proportion of large individuals in the community; (c) the biomass size-spectrum; and (d) the diversity size spectrum (Rice & Gislason 1996).
- Spatial distributions: Fishing and climate/oceanographic variability/change can alter the geographic distribution of fish species (Perry et al. 2010) and this can indicate an ecosystem-level change. The percentage area of a research survey in which most (typically 90%) of the population occurs has been used as an ecosystem indicator (e.g., Fisher & Frank 2004, Tuck et al. 2009).
- Diet-based indicators: The change of diet (or trophic position) of a species of fish may reveal that trophic or ecosystem-level changes have occurred (e.g., Smith & Lucey 2014), but trophic position may change less than the underlying ecosystem structure (Badalamenti et al. 2002). 'Niche width' measured in terms of the range of carbon and nitrogen isotope ratios occupied by a species has also been suggested as indicative of trophic changes in a marine ecosystem especially in relation to upper trophic level predators (Layman et al. 2007), but the utility of this has been questioned (Hoeinghaus & Zeug 2008).

13.5.1.5 TOP PREDATORS

Top predators (upper trophic level consumers) can be used in two ways as indicators of the state of marine ecosystems. First, an OECD core indicator is the overall ecological threat status of species in the ecosystem, often with an emphasis placed on top predators (OECD 2003). Second, particular

[10] Those that produce relatively low numbers of offspring, typically growing more slowly and maturing later.

[11] Those that produce high numbers of offspring, typically growing faster and maturing sooner.

ecological aspects of selected predator species can be used to indicate changes in ecosystems. For example, top predators are widely used in monitoring the ecosystem effects of fishing krill in the Southern Ocean (Reid et al. 2005, Constable 2006), with information on the breeding of penguins, albatross, petrels, and seals collected, summarised and considered in management annually (CEMP 2004, Agnew 1997). Monitoring top predators as 'bellweathers' of ecosystem health is also increasingly used elsewhere (Boyd et al. 2006, Ainley 2002) as they are recognised as potentially useful downstream integrators of change in the marine ecosystem, exploit marine resources at similar spatial and temporal scales to humans, and receive high public interest. However, given that predators respond in complex ways to many factors simultaneously, ascertaining the appropriate management response to change of a predator-based indicator is difficult (Boyd et al. 2006).

13.5.2 NEW ZEALAND

There has been much work in New Zealand on developing indicators of the marine environment. MPI have carried out a number of projects looking at indicators and time series, including of oceanographic/climate variables (Hurst et al. 2008, Dunn et al. 2007, Pinkerton et al. 2014a), demersal fish communities based on data from scientific trawls (Tuck et al. 2009), and a suite of indicators relevant to deepwater fisheries (Tuck et al. 2014). Other work in New Zealand on marine ecosystem indicators include reports under NIWA Core funding (Pinkerton 2010) and in relation to national environmental reporting (Gilbert et al. 2000, Pinkerton 2007, Pinkerton 2014a).

13.5.2.1 MARINE PRIMARY PRODUCTION

Ocean colour satellite data have been used for more than a decade in New Zealand to investigate spatial and seasonal patterns in phytoplankton abundance and NPP (Murphy et al. 2001, Pinkerton 2007). There is a limited number of data available in New Zealand waters to develop locally tuned estimates of NPP from satellite data, and the concentration of chlorophyll-a is preferred for the purposes of monitoring change in primary production over time (Pinkerton et al. 2014a). Since 2002, mean concentrations of chlorophyll-a in the EEZ have decreased by an average of about 1% per year (Pinkerton, unpublished data). This is likely to be related, at least in part, to oceanographic cycles such as the Interdecadal Pacific Oscillation index[12] and the Southern Oscillation Index,[13] as well as potentially to long-term climate change.

13.5.2.2 LOWER FOOD WEB (MICROBIAL SYSTEM)

Changes to primary production also do not necessarily translate to less food available for higher trophic levels. Virtually all wild-caught seafood in New Zealand are carnivorous, with a mean trophic index of about 4.1 (MacDiarmid et al. 2013) The trophic efficiency by which energy passes between trophic levels is often considered to be about 10% (Pauly & Christensen 1995), meaning that only about one-tenth of the energy consumed by marine organisms is used to build new body mass. This means that each tonne of wild-caught seafood in New Zealand has been supported by over a thousand tonnes of primary production that has been moved through at least two intermediate levels in the marine food web before being consumed by the target species. A change to the lower and middle parts of the New Zealand food web hence have the potential to affect food availability for, and potentially yield of, commercially important fish stocks. At present, there are no data available to monitor for changes in the functioning of the lower trophic levels of New Zealand's marine ecosystems.

[12] The Interdecadal Pacific Oscillation (also called the Pacific Decadal Oscillation) is a 15–30-year cycle that affects parts of the Pacific Basin, causing variability in climate and oceanography, and has substantial and long-lasting effects on regional ecosystems (Kennedy et al. 2002).

[13] The Southern Oscillation Index is related to the strength of the trade winds in the Southern Hemisphere tropical Pacific (Mullan 1995) and SOI values for May–September are often used as an indicator of El Niño-La Niña Southern Oscillation (ENSO).

13.5.2.3 MIDDLE TROPHIC LEVELS

Middle trophic level organisms in the New Zealand ocean are diverse (more than 21 species of myctophids occur on the Chatham Rise for example; Pinkerton, unpublished data). Although they form the basis of the diet of many commercially-important New Zealand fish species (Dunn et al. 2009a), the basic abundance, distribution and ecology of key middle-trophic level groups like myctophids and hyperbenthic arthropods (prawns and shrimps) are generally poorly known. Two time series of data for middle trophic level organisms in the New Zealand ocean may be useful to investigate trophic and ecosystem-level effects: (a) New Zealand acquired a Continuous Plankton Recorder (CPR) in 2008 and this has been deployed on a transit extending from Oamaru (approximately 45°S) to the Ross Sea annually since summer 2008–09; approximately 1200 km of this transect are in the subantarctic New Zealand EEZ (Robinson et al. 2013); (b) recent work has shown that multifrequency acoustic backscatter data taken from research vessels during the annual surveys of fish on the Chatham Rise can be used to derive indices of abundance of mesopelagic fish and invertebrates (McClatchie & Dunford 2003, O'Driscoll et al. 2009, Oeffner et al. 2014). Similar acoustic methods could provide time series of middle trophic level species in the Hauraki Gulf and subantarctic plateau in the near future.

13.5.2.4 DEMERSAL FISH COMMUNITIES

There are three series of scientific trawls in New Zealand waters that are particularly valuable for understanding ecosystem dynamics and for monitoring for trophic and ecosystem-level effects at the level of the demersal fish community (Tuck et al. 2009): (a) a scientific trawl survey has been carried out on the Chatham Rise region approximately annually since 1992; (b) a similar survey has been carried out over the subantarctic plateau over the same period but less frequently (Bagley & O'Driscoll 2012, Tuck et al. 2009); (c) a total of 15 trawl surveys have also been carried out in the Hauraki Gulf region between 1980 and 2000. Each of these trawl surveys used a consistent methodology based on scientific bottom trawl gear. Tuck et al. (2009) used these scientific surveys to investigate change in a series of indicators based on the demersal fish community.

Data from Chatham Rise trawl surveys between 1992 and 2007 showed evidence of increasing evenness (reducing diversity) but no evidence that species were being lost from the food web (Tuck et al. 2009). Some size characteristics of fish in research trawls on the Chatham Rise had changed, with fewer fish longer than 30 cm or heavier than 750 g being taken by trawl gear, although the median length of the catch did not change. Preliminary analysis of the mean trophic level index (MTI) in the demersal fish community of the Chatham Rise (Pinkerton 2010) indicated that this also decreased over the same period, and decreased more in the trawl survey data than in the commercial catch data. The proportion of piscivorous fish and of true demersal (rather than bentho-pelagic) species also declined over this period (Tuck et al. 2009). Somewhat counterintuitively, threatened[14] species and species defined by Tuck et al. (2009) as 'low-resilience', such as dogfish and rays, have increased relative to other species on the Chatham Rise. This was confirmed by independent analyses of Chatham Rise trawl survey data (O'Driscoll et al. 2011) and may be due to a combination of a lack of incentive to catch these species by the fishing fleet and an increase in offal and discards that benefit demersal scavengers. There were changes in the spatial distribution of fish species, with 16 out of 47 species showing changes in the proportion of the study area over which 90% of their abundance by weight was caught. Of these, half showed declining range and half showed increasing range. Tuck et al. (2009) showed that on the Chatham Rise, the species showing contractions of range were generally the more abundant species whereas the species expanding in spatial range were generally the less abundant species. MPI project ZBD2004/02 (Dunn et al. 2009a; Horn & Dunn 2010) examined whether there was evidence of change in the diet of hoki, hake or ling on the Chatham Rise between 1990 and 2009. It appears likely that the importance of fish (primarily myctophids) as a prey item for hoki has increased slightly but steadily between 1990 and 2009, while the importance of euphausiids has declined. In contrast, there were no obvious between-year

[14] Species deemed more vulnerable according to the IUCN Red List (IUCN 2009); see Tuck et al. (2009).

differences or trends in hake diet from 1990 to 2009 (Horn & Dunn 2010). There were some marked between-year differences in ling diet in this period but no trends detected.

Discards and offal from fisheries is sometimes an important part of the diets of deepwater fish. For example, scavenged fishes accounted for up to a quarter of the diet of smooth skate (*Raja innominata*) in the Chatham Rise region (Dunn et al. 2009a, Forman & Dunn 2012). Anderson & Smith (2005) estimated that 11 000–14 000 t per year of non-commercial species and 600–2100 t per year of hoki are discarded by the New Zealand hoki fishery, leading to the potential for a significant modification of the diet of scavenging species (Forman & Dunn 2012). Interpreting changes in diet from discards in a way that can inform fisheries management is not straightforward. For the Chatham Rise, the changes covered a period of declining hoki spawning biomass (McKenzie 2013) and occurred at the same times as evidence of climate variation, namely a shift the prevalence of Kidson weather types (Kidson 2000) between 1992 and 2007 (Hurst et al. 2012). Disentangling these environmental and fishery drivers of changes to indicators of the demersal fish communities has not yet been attempted in New Zealand although the hypothesis that trophic or environmental factors were responsible for recent changes in hoki recruitment was investigated and was found not to be supported empirically (Francis et al. 2006, Bradford-Grieve et al. 2011).

13.5.2.5 TOP PREDATORS

Information on indicators of change in upper trophic levels in New Zealand are considered in Theme 1 of this report.

13.6 DISCUSSION

Marine ecosystems are complex, show non-linear dynamics (including potential tipping-points) and are subject to a wide range of impacts, including fishing, climate variability and change, coastal eutrophication and habitat change. Any activities that change the composition of species in the ecosystem (both in terms of size, functional group, ecosystem role, and diversity) will affect other groups in the ecosystem through trophic and other connections. A large range of trophic and ecosystem-level effects in marine systems have been documented internationally and these have generally been associated with negative impacts on fisheries (Garcia & Grainger 2005, Valdes et al. 2009, Worm et al. 2009). Understanding the scale and causes of these changes remains scientifically challenging (Rice 2001, Brander 2010, Jennings & Brander 2010, ter Hofstede et al. 2010). There remains substantial debate about the true extent and magnitude of these changes (Hilborn 2007, Murawski et al. 2007) and debate about how to allocate responsibility for these changes among different pressures, including fishing (Benoît & Swain 2008, Holt & Punt 2009, Kotta et al. 2009, Noakes & Beamish 2009, Rijnsdorp et al. 2009, Rice & Garcia 2011, Schiel 2013). Although ecosystem-level changes have rarely been ascribed solely to fisheries drivers, it appears that fishing is likely to make ecosystems less resilient to variability and change in climate/oceanographic forcing (Winder & Schindler 2004, Kirby et al. 2008, 2009). Reduced ecosystem resilience is an ecosystem-level effect that may predominantly occur through trophic mechanisms. Reduced ecosystem resilience may affect the long-term sustainability of harvesting (Hughes et al. 2005), increase ecosystem variability (Salomon et al. 2010), make fisheries less predictable and harder to manage in a variable and changing climate (Badjeck et al. 2010, Brander 2010, McIlgorm et al. 2010), reduce the ability of ecosystems to recover from overfishing (Neubauer et al. 2013), and increase the likelihood or consequence of regime shifts or invasive species (Folke et al. 2004, Salomon et al. 2010).

To date, it has generally not proved possible to realistically (as opposed to theoretically) identify at what point fishing or other pressure may cause serious disruptions in resource productivity or ecosystem function through trophic or ecosystem-level effects. For multi-species fisheries that are managed at a stock level close to B_{MSY} in a way that does not progressively degrade benthic habitat, it is not known whether it is necessary to take trophic and ecosystem-level effects into account more explicitly to ensure long-term sustainability of fisheries (ICES 2005). Some studies (e.g., Jackson et al. 2001, Jennings et al. 2002, Branch 2009), model analyses (Walters et al. 2005, Legovic et al. 2010, Gecek & Legovic 2012, Legovic & Gecek 2012, Ghosh & Kar 2013), and expert groups (Scientific Committee on Oceanographic Research/Intergovernmental Oceanographic Commission working group on indicators; Cury & Christensen 2005) have concluded that harvesting

many species in an ecosystem at B_{MSY} [15] can lead to increased chance of fisheries collapse in the medium to long term – an effect called 'ecosystem erosion' or 'ecosystem overfishing' (Murawski 2000, Coll et al. 2008).

ICES (2005) concluded that, for fisheries managed at or close to B_{MSY}, the priority was to avoid fishing practices that drastically changed benthic structure, trophic interactions, food web structures or nutrient cycling (ICES 2005). This is consistent with the widespread consensus that fisheries should be managed within an ecosystem context and by adopting a precautionary approach that includes acknowledging the potentially synergistic effects of fishing and climate change (CBD 2009, Perry et al. 2010, Rice & Garcia 2011). However, there is little consensus on what this actually means in practice (FAO 2008, Ecosystem Principles Advisory Panel 1999, Browman & Stergiou 2004, 2005, Garcia & Cochrane 2005, Murawski 2011). Work by NOAA fisheries (Marasco et al. 2007) towards a pragmatic approach to ecosystem-based fishery management recommended:

- incorporating a broader array of societal goals and uses for ecosystem products and services within a multiple use multiple stressors framework;
- recognising the significance of ocean-climate conditions;
- emphasising food web interactions (recognise that harvest of target species has profound impacts on ecosystem structure and function through trophic interactions);
- employing spatial representation (manage stocks consistent with spatial/habitat variation in productivity);
- increasing and expanding focus on characterising and maintaining viable fish habitats;
- expanding scope of research and monitoring (increased focus on understanding biological interactions/processes, and measuring total fishery removals of target and non-target species);
- acknowledging and responding to higher levels of uncertainty (realistically incorporate uncertainty due to trophic and food web effects into management policy);
- reviewing and improving ecosystem modelling/research.

The role of no-take reserves or marine protected areas (MPAs) in guarding against trophic and ecosystem-level effects remains controversial. A full review of the value of MPAs in this regard is beyond the scope of the present chapter. Suffice to say that some scientists believe strongly that MPAs can be effective at providing an 'ecological safety net' for trophic and ecosystem-level effects (Ballantine 2014, Edgar et al. 2014) whereas other scientists believe MPAs are too few and too small to have any value in this regard (Kaiser 2005, Mora et al. 2006). No-take marine reserves may have the most to contribute to our understanding of trophic and ecosystem effects by providing a 'reference ecosystem' in which populations experience low fishing pressure but a full range of other stressors (such as environmental variability/change, sedimentation, and pollution). Ecosystem changes in the reserve can then be contrasted with adjacent ecosystems exposed to the full range of fishing and other impacts (Micheli et al. 2005).

New Zealand is currently doing better than most countries with regard to many of the recommendations of Marasco et al. (2007). Pitcher et al. (2009) evaluated the performance of 33 countries for ecosystem-based management (EBM) of fisheries in three fields (principles, criteria and implementation). No country rated overall as 'good', only four countries, including New Zealand were 'adequate'. Specific recommendations from Marasco et al. (2007) are relevant to recent research initiatives in New Zealand. The newly announced Sustainable Seas research programme[16] aims to engage more closely with society to ensure that its goals and concerns are heard and addressed. Similarly, the MBIE Marine Futures project led by Dr Simon Thrush has used a multiple use framework to consider how ecosystem resilience can be promoted in the two focus

[15] The biomass that allows the maximum sustainable yield to be taken.

[16] Beehive. Sustainable Seas National Science Challenge launched. 4 September 2014. Retrieved from http://www.beehive.govt.nz/release/sustainable-seas-national-science-challenge-launched.

areas of the Hauraki Gulf and Chatham Rise. Hurst et al. (2012) and Dunn et al. (2009b) considered the impact of ocean-climate interactions on New Zealand fisheries. The Ocean Survey 20/20 voyages had an explicit focus on mapping the distribution of seafloor habitats important to fish stocks and associated species (Hewitt et al. 2011). Ecosystem modelling of key New Zealand regions has been an ongoing focus of NIWA core-funded research since 2005, and includes co-funded ecosystem modelling work with MPI (e.g., ZBD2005/05). Data collection towards building up a comprehensive predator-prey database began with the ZBD2004/01 project (Dunn et al. 2009) and continues on the Chatham Rise under NIWA core-funding, with a particular focus on middle trophic level organisms that are abundant. MfE aim to include multi-trophic indicators of marine ecological state in the National Environmental Reporting (Pinkerton et al. 2014, Pinkerton 2014b), DOC are aiming to develop marine ecological integrity indicators (Freeman, pers. comm.), and MPI are actively developing indicators of change in fish communities (Tuck et al. 2009, 2014).

Notwithstanding this progress, most New Zealand stocks are managed on a single-stock basis at close to B_{MSY} (Ministry of Fisheries 2008) irrespective of their role in the ecosystem. The balance of evidence suggests that fishing close to B_{MSY} and in particular using bottom trawling (which impacts on benthic ecosystem function; Thrush & Dayton 2002) is likely to reduce ecosystem resilience and increase ecosystem variability by trophic and ecosystem-level effects (Brock & Carpenter 2006, Carpenter & Brock 2006, van Nes & Scheffer 2007, Guttal & Jayaprakash 2008) and could increase recruitment variability. Fishing is also likely to strengthen bottom-up control of marine ecosystems and make ecosystems more sensitive to the effects of climate change (Kirby et al. 2009, Perry et al. 2010). Greater sensitivity of marine ecosystems to climate variability implies a higher potential for regime shift which may or may not be reversible or desirable (Hsieh et al. 2006). Stronger environmental (bottom-up) forcing of ecosystems suggests a greater likelihood of unexpected changes to fisheries due to extreme environmental events and that these changes may be more severe (Perry et al. 2010, Kirby & Beaugrand 2009).

Time series measurements are crucial to understanding ecosystem function and monitoring for trophic and ecosystem-level effects of fishing. There would seem to be high value in maintaining regular and frequent (annual) surveys of the demersal fish communities of key New Zealand regions (such as the Chatham Rise, Hauraki Gulf and subantarctic plateau). Information on the catches of all species by the fishing fleet is required to monitor for changes in trophically or ecologically important non-QMS species. A key knowledge gap is information to map and monitor abundances, trophic connections and community structure of middle trophic level species, especially mesozooplankton, mesopelagics and hyperbenthics in key fishing areas, such as the Chatham Rise, Hauraki Gulf and subantarctic plateau. Knowledge of the abundance and trophic ecology of small demersal fishes in these regions is notably lacking.

13.7 CONCLUSIONS

1. A range of trophic and ecosystem-level effects in marine systems have been documented internationally, and these have generally been associated with negative impacts on fisheries.
2. Trophic and ecosystem-level effects are not usually brought about by fishing alone, but fishing (especially overfishing but also at or close to B_{MSY}) in multispecies fisheries can make ecosystems less resilient and more sensitive to the effects of environmental variability and change.
3. New Zealand's marine ecosystems are particularly diverse and this provides special challenges in monitoring, understanding and managing fisheries operating in them.
4. There is currently no evidence of a large-scale trophic or ecosystem-level effect impacting New Zealand's deepwater fisheries, but the cause of some changes in New Zealand's marine ecosystem EEZ are not known (e.g., changes to hoki recruitment (Francis et al. 2006, Bradford-Grieve & Livingston 2011); trends in some demersal-fish indicators on the Chatham Rise and other areas (Tuck et al. 2009).
5. It is likely that the reduction in the abundance of sea urchin predators on some rocky reef systems in north-eastern New Zealand due to fishing has contributed to an ecosystem-level effect in these areas, but this effect is unlikely to be widespread in New Zealand coastal areas (Schiel 2013).

6. Multi-species fishing at close to B_{MSY} using predominantly bottom-trawling is likely to make New Zealand's marine ecosystems less resilient (compared to fishing more conservatively compared to B_{MSY} and not using predominantly bottom-trawling) to other anthropogenic disturbance and to environmental variability, including climate change, through trophic and ecosystem-level effects.
7. There are potential, but unknown, trophic and ecosystem-level consequences for fisheries management in New Zealand if populations of marine mammals, such as fur seals, rebuild to levels that some people have suggested existed before humans arrived in New Zealand (see Theme 1 of this report).
8. Time series monitoring of fish communities and middle trophic level species (mesozooplankton, mesopelagics, hyperbenthics) are crucial for understanding and monitoring for trophic and ecosystem-level effects, and the best current sources of these data are trawl surveys to the Chatham Rise, and subantarctic plateau.

13.8 REFERENCES

Aebischer, N J; Coulson, J C; Colebrook, J M (1990) Parallel long-term trends across four marine trophic levels and weather. *Nature 347*: 753–755.

Agnew, D J (1997) The CCAMLR Ecosystem Monitoring Programme. *Antarctic Science 9 (3)*: 235–242.

Ainley, D G (2002) The Adélie penguin. Columbia University Press, New York. 310 p.

Akroyd, J M; Pierre, J P (2013) Surveillance Report New Zealand Hoki Fishery. 24 p.

Anderson, O F; Smith, M H (2005) Fish discards and non-target fish catch in the New Zealand hoki trawl fishery, 1999–2000 to 2002–03. *New Zealand Fisheries Assessment Report 2005/3*. 37 p.

Arcos, D F; Cubillos, L A; Nunez, S P (2001) The jack mackerel fishery and El Nino 1997–98 effects off Chile. *Progress in Oceanography 49*: 597–617.

Arrigo, K (2005) Marine microorganisms and global nutrient cycles. *Nature 437*: 349–355.

Babcock, R C; Kelly, S; Shears, N T; Walker, J W; Willis, T J (1999) Changes in community structure in temperate marine reserves. *Marine Ecology Progress Series 189*: 125–134.

Babcock, R C; Shears, N T; Alcala, A C; Barrett, N S; Edgar, G J; Lafferty, K D; McClanahan, T R; Russ, G R (2010) Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *Proceedings of the National Academy of Sciences 107*: 18256–18261.

Badalamenti, F; D'Anna, G; Pinnegar, J; Polunin, N (2002) Size-related trophodynamic changes in three target fish species recovering from intensive trawling. *Marine Biology 141 (3)*: 561–570.

Badjeck, M C; Allison, E H; Halls, A S; Dulvy, N K (2010) Impacts of climate variability and change on fishery-based livelihoods. *Marine Policy 34*: 375–383.

Bagley, N W; O'Driscoll, R L (2012) Trawl survey of hoki, hake and ling in the Southland and Sub-Antarctic areas, November-December 2009 (TAN0911). *New Zealand Fisheries Assessment Report 2012/05*. 70 p.

Bakun, A (2004) Chapter 24 – Regime shifts. *In*: Robinson, A R; McCarthy, J; Rothschild, B J (Eds.), The Sea. Volume 13. pp 971–1018. Harvard University Press, Cambridge, MA.

Bakun, A (2006) Wasp-waist populations and marine ecosystem dynamics: navigating the 'predator pit' topographies. *Progress in Oceanography 68*: 271–288.

Bakun, A (2010) Linking climate to population variability in marine ecosystems characterized by non-simple dynamics: conceptual templates and schematic constructs. *Journal of Marine Systematics 79*: 361–373.

Bakun, A; Weeks, S J (2006) Adverse feedback sequences in exploited marine systems: Are deliberate interruptive actions warranted? *Fish and Fisheries 7*: 316–333.

Ballantine, B (2014) Fifty years on: Lessons from marine reserves in New Zealand and principles for a worldwide network. *Biological Conservation 176*: 297–307. doi:10.1016/j.biocon.2014.01.014.

Banse, K (1995) Zooplankton: Pivotal role in the control of ocean production. *ICES Journal of Marine Science 52*: 265–277.

Barber, R T; Chavez, F P (1983) Biological consequences of El Nino. *Science 222*: 1203–1210.

Beare, D J; McKenzie, E (1999) Connecting ecological and physical time series: the potential role of changing seasonality. *Marine Ecology Progress Series 178*: 307–309.

Beaugrand, G (2004) The North Sea regime shift: evidence, causes, mechanisms and consequences. *Progress in Oceanography 60*: 245–262.

Beaugrand, G (2005) Monitoring pelagic ecosystems using plankton indicators. *ICES Journal of Marine Science 62*: 333–338.

Beaugrand, G; Ibanez, F (2004) Monitoring marine plankton ecosystems (2): long-term changes in North Sea calanoid copepods in relation to hydro-climatic variability. *Marine Ecology Progress Series 284*: 35–47.

Beaugrand, G; Reid, P C (2003) Long-term changes in phytoplankton, zooplankton and salmon linked to climate. *Global Change Biology 9*: 801–817.

Beaugrand, G; Reid, P C; Ibanez, F; Lindley, J A; Edwards, M (2002) Reorganisation of North Atlantic marine copepod biodiversity and climate. *Science 296*: 1692–1694.

Beaumont, J; MacDiarmid, A; Morrison, M, et al. (2009) Rocky reef ecosystems - how do they function? Integrating the roles of primary and secondary production, biodiversity, and connectivity across coastal habitats. Final Research Report for project ZBD2005-09. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Becker, P H; Erdelen, M (1986) Trends of coastal bird populations of the German Wadden Sea since 1950: aspects relevant to species conservation. *Berichte der Deutschen Sektion des Internationalen Rats für Vogelschutz 26*: 63–73.

Benoît, H P; Swain, D P (2008) Impacts of environmental change and direct and indirect harvesting effects on the dynamics of a marine fish community. *Canadian Journal of Fisheries and Aquatic Sciences 65*: 2088–2104.

Bianchi, G; Gislason, H; Graham, K, et al. (2000) Impact of fishing on size composition and diversity of demersal fish communities. *ICES Journal of Marine Science 57 (3)*: 558–571.

Birt, V L; Birt, T P; Goulet, D; Cairns, D K; Montevecchi, W A (1987) Ashmole's halo: direct evidence for prey depletion by a seabird. *Marine Ecology Progress Series 40*: 205–208.

Bogstad, B; Mehl, S (1997) Interactions between Atlantic cod (*Gadus morhua*) and its prey species in the Barents Sea. *In*: Forage fishes in marine ecosystems. Proceedings of the International Symposium on the Role of Forage Fishes in Marine Ecosystems. pp 591–615. Alaska Sea Grant College Program Report No 97-01, University of Alaska Fairbanks.

Borer, E T; Seabloom, E W; Shurin, J B; Anderson, K E; Blanchette, C A; Broitman, B; Cooper, S D; Halpern, B S (2005) What determines the strength of a trophic cascade? *Ecology 86*: 528–537.

Bowden, D A (2011) Benthic invertebrate samples and data from the Ocean Survey 20/20 voyages to the Chatham Rise and Challenger Plateau, 2007. *New Zealand Aquatic Environment and Biodiversity Report No. 65*. 46 p.

Boyd, I L; Wanless, S; Camphuysen, C J (2006) Top predators in marine ecosystems: their role in monitoring and management. Conservation Biology 12. Cambridge University Press. 378 p.

Boyd, P; LaRoche, J; Gall, M; Frew, R; McKay, R M L (1999) The role of iron, light and silicate in controlling algal biomass in sub-Antarctic water SE of New Zealand. *Journal of Geophysical Research 104 (C6)*: 13395–13408.

Boyer, D (1996) Stock dynamics and ecology of pilchard in the northern Benguela. *In*: O'Toole, M J (Ed.), The Benguela Current and Comparable Eastern Boundary Upwelling Ecosystems. pp 79–82. Deutsche Gesellschaft fur Techhnische Zusammenarbeit (GTZ) GmbH, Eschborn, Germany.

Boyer, D C; Hampton, I (2001) An overview of the living marine resources of Namibia. *South African Journal of Marine Science 23*: 5–35.

Bradford-Grieve, J M; Chang, F H; Gall, M; Pickmere, S; Richards, F (1997) Size-fractionated phytoplankton standing stocks and primary production during austral winter and spring 1993 in the Subtropical Convergence region near New Zealand. *New Zealand Journal of Marine and Freshwater Research 31*: 201–224.

Bradford-Grieve, J M; Livingston, M E (Eds.) (2011) Spawning fisheries and the productivity of the marine environment off the west coast of the South Island, New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 84*. 136 p.

Branch, T A (2009) How do individual transferable quotas affect marine ecosystems? *Fish and Fisheries 10 (1)*: 39–57.

Branch, T A; Watson, R; Fulton, E A; Jennings, S; McGilliard, C R; Pablico, G T; Ricard, D; Tracey, S R (2010) The trophic fingerprint of marine fisheries. *Nature 468*: 431–435. doi:10.1038/nature09528.

Brander, K (2010) Impacts of climate change on fisheries. *Journal of Marine Systems 79*: 389–402.

Brierley, A S; Kingsford, M J (2009) Impacts of climate change on marine organisms and ecosystems. *Current Biology 19*: R602–R614.

Brock, W A; Carpenter, S R (2006) Variance as a leading indicator of regime shift in ecosystem services. *Ecology and Society 11 (2)*: 9.

Brodeur, R D; Mills, C E; Overland, J E; Walters, C E; Schumacher, J D (1999) Evidence for a substantial increase in gelatinous zooplankton in the Bering Sea, with possible links to climate change. *Fisheries Oceanography 8*: 296–306.

Brodeur, R D; Sugisaki, H; Hunt, G L Jr (2002) Increases in jellyfish biomass in the Bering Sea: implications for the ecosystem. *Marine Ecology Progress Series 233*: 89–103.

Brose, U; Berlow, E L; Martinez, N D (2005) Scaling up keystone effects from simple to complex ecological networks. *Ecology Letters 8 (12)*: 1317–1325.

Browman, H I; Stergiou, K I (Eds.) (2004) Perspectives on ecosystem-based approaches to the management of marine resources. *Marine Ecology Progress Series 274*: 269–303.

Browman, H I; Stergiou, K I (Eds.) (2005) Politics and socio-economics of ecosystem-based management of marine resources. *Marine Ecology Progress Series 300*: 241–296.

Caddy, J; Csirke, J; Garcia, S M; Grainger, R J L (1998) How pervasive is 'Fishing down marine food webs'? *Science 282*: 1383.

Caddy, J F; Garibaldi, L (2000) Apparent changes in the trophic composition of world marine harvests: the perspective from the FAO

capture database. *Ocean and Coastal Management 43*: 615–655.

Campbell, J; Antoine, D; Armstrong, R, et al. (2002) Comparison of algorithms for estimating ocean primary production from surface chlorophyll, temperature, and irradiance. *Global Biogeochemical Cycles 16 (3)*: 1035. doi:10.1029/2002GL015068.

Cardinale, M; Svedäng, H (2008) Mismanagement of fisheries: policy or science? *Fisheries Research 93*: 244–247.

Carpenter, S R; Brock, W A (2006) Rising variance: A leading indicator of ecological transition. *Ecology Letters 9 (3)*: 311–318.

Carpenter, S R; Kitchell, J F; Hodgson, J R (1985) Cascading trophic interactions and lake productivity. *Bioscience 35*: 634–639.

Carter, S K; VanBlaricom, G R; Allen, B L (2007) Testing the generality of the trophic cascade paradigm for sea otters: a case study with kelp forests in northern Washington, USA. *Hydrobiologia 579*: 233–249.

Casini, M; Blenckner, T; Möllmann, C; Gårdmark, A; Lindegren, M; Llope, M; Kornilovs, G; Plikshs, M; Stenseth, N C (2012) Predator transitory spillover induces trophic cascades in ecological sinks. *Proceedings of the National Academy of Sciences USA 109*: 8185–8189.

Casini, M; Cardinale, M; Arrhenius, F (2004) Feeding preferences of herring (*Clupea harengus*) and sprat (*Sprattus sprattus*) in the southern Baltic Sea. *ICES Journal of Marine Sciences 61*: 1267–1277.

Casini, M; Hjelm, J; Molinero, J C; Lövgren, J; Kornilovs, G; Cardinale, M; Bartolino, V; Belgrano, A (2009) Trophic cascades promote threshold-like sudden shifts in the functioning of a pelagic marine ecosystem. *Proceedings of the National Academy of Sciences 106*: 197–202.

Casini, M; Lövgren, J; Hjelm, J; Cardinale, M; Molinero, J C; Kornilovs, G (2008) Multi-level trophic cascades in a heavily exploited open marine ecosystem. *Proceedings of the Royal Society B, Biological Sciences 275*: 1793–1801.

CBD (2004) Convention on Biological Diversity Annex I, decision VII/30. The 2020 biodiversity target: a framework for implementation. p 351. Decisions from the Seventh Meeting of the Conference of the Parties of the Convention on Biological Diversity, Kuala Lumpur, 9–10 and 27 February 2004. Secretariat of the CBD, Montreal.

CBD (2009) Findings of the Ad Hoc Technical Expert Group on Biodiversity and Climate Change CBC CBCDD-0202. 100 p.

CEMP (2004) CCAMLR Ecosystem Monitoring Programme. CCAMLR Document, Hobart, Australia. 268 p.

Christensen, V; Guénette, S; Heymans, J J; Walters, C J; Watson, R; Zeller, D; Pauly, D (2003) Hundred-year decline of North Atlantic predatory fishes. *Fish and Fisheries 4*: 1–24.

Christensen, V; Walters, C J (2004) Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling 172*: 109–139.

Christoffersen, K; Riemann, B; Klysner, A; Søndergaard, M (1993) Potential role of fish predation and natural populations of zooplankton in structuring a plankton community in eutrophic lake water. *Limnology and Oceanography 38*: 561–573.

Cobb, J S; Castro, K M (2006) Shell disease in lobsters: a synthesis. Report prepared for the New England Lobster Research Initiative. *In*: Cohen, J E; Briand, F; Newman, C M (1990). Community food webs: data and theory. Springer-Verlag, New York. 18 p.

Cohen, J E; Cohen, F; Briandan, C M (Eds.) (1990) Community food web: Data and theory. Berlin, Springer-Verlag.

Coll, M; Libralato, S; Tudela, S; Palomera, I; Pranovi, F (2008) Ecosystem Overfishing in the Ocean. *PLoS ONE 3 (12)*: e3881. doi:10.1371/journal.pone.0003881.

Collins, M (2000) Understanding the uncertainties in the response of ENSO to greenhouse warming. *Geophysical Research Letters 27*: 3509–3512.

Condon, R H; Duarte, C M; Pitt, K A, et al. (2013) Recurrent jellyfish blooms are a consequence of global oscillations. *Proceedings of the National Academy of Sciences USA 110 (3)*: 1000–1005. doi:10.1073/pnas.1210920110.

Constable, A J (2006) Setting management goals using information from predators. *In*: Boyd, I L; Wanless, S; Camphuysen, C J (Eds.), Top predators in marine ecosystems: their role in monitoring and management. pp 324–346. Conservation Biology 12, Cambridge University Press.

Cooley, S R; Doney, S C (2009) Anticipating ocean acidification's economic consequences for commercial fisheries. *Environmental Research Letters 4 (2)*: 1–18.

Crawford, R J M (2007) Food, fishing and seabirds in the Benguela upwelling system. *Journal of Ornithology 148*: 253–260.

Cury, P; Bakun, A; Crawford, R J M; Jarre-Teichmann, A; Quinones, R A; Shannon, L J; Verheye, H M (2000) Small pelagics in upwelling systems: patterns of interaction and structural changes in 'wasp-waist' ecosystems. *ICES Journal of Marine Science 210*: 603–618.

Cury, P M; Christensen, V (2005) Quantitative ecosystem indicators for fisheries management. *ICES Journal of Marine Science 62 (3)*: 307–310.

Cury, P; Freon, P; Moloney, C L; Shannon, L; Shin, Y-J (2004) Processes and patterns of interactions in marine fish populations: an ecosystem perspective. *In*: Robinson, A R; McCarthy, J; Rothschild, B J (Eds.), The Sea. Volume 13. pp 475–533. Harvard University Press, Cambridge, MA.

Cury, P M; Shannon, L J; Roux, J-P; Daskalov, G M; Jarre, A; Moloney, C L; Pauly, D (2005) Trophodynamic indicators for an ecosystem

approach to fisheries. *ICES Journal of Marine Science 62*: 430–442.

Daskalov, G M; Grishin, A N; Rodionov, S; Mihneva, V (2007) Trophic cascades triggered by overfishing reveal possible mechanisms of ecosystem regime shifts. *Proceedings of the National Academy of Science USA 104 (25)*: 10518–10523.

Dayton, P K; Thrush, S F; Agardy, T M; Hofman, R J (1995) Environmental-Effects of Marine Fishing. *Aquatic Conservation-Marine and Freshwater Ecosystems 5 (3)*: 205–232.

de Roos, A M; Boukal, D S; Persson, L (2006) Evolutionary regime shifts in age and size at maturation of exploited fish stocks. *Proceedings of the Royal Society of London B 273 (1596)*: 1873–1880.

DOC-MPI (2011) Coastal marine habitats and marine protected areas in the New Zealand Territorial Sea: a broad scale gap analysis. Department of Conservation and Ministry of Fisheries, Wellington, New Zealand.

Drinkwater, K F; Beaugrand, G; Kaeriyama, M; Kim, S; Ottersen, G; Perry, R I; Pörtner, H-O; Polovina, J; Takasuka, A (2010) On the mechanisms linking climate to ecosystem changes. *Journal of Marine Systematics 79*: 374–388.

Duarte, C M; Pitt, K A; Lucas, C H, et al. (2012) Is global ocean sprawl a cause of jellyfish blooms? *Frontiers in Ecology and the Environment 11*: 91–97.

Duffy, D C (1983) Environmental uncertainty and commercial fishing: Effects on Peruvian guano birds. *Biological Conservation 26*: 227–238.

Dunn, M; Horn, P; Connell, A; Stevens, D; Forman, J; Pinkerton, M; Griggs, L; Notman, P; Wood, B (2009a) Ecosystem-scale trophic relationships: diet composition and guild structure of middle-depth fish on the Chatham Rise. Final Research Report for Ministry of Fisheries research project ZBD2004-02, Objectives 1–5. 351 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Dunn, M; Hurst, R; Francis, C; Devine, J; Renwick, J; McKenzie, A (2007) Fish abundance and climate: trends and possible linkages. Final Research Report for Ministry of Fisheries project SAM2005-02, Objective 1. NIWA, Wellington. 164 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Dunn, M R; Hurst, R; Renwick, J A; Francis, R I C C; Devine, J; McKenzie, A (2009b) Fish abundance and climate trends in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 31*. 74 p.

Dutil, J D; Lambert, Y (2000) Natural mortality from poor condition in Atlantic cod (Gadus morhua). *Canadian Journal of Fisheries and Aquatic Science 57*: 826–836.

Ecosystem Principles Advisory Panel (1999) Ecosystem-based fishery management. National Marine Fisheries Service, Washington, DC.

Eddy, T D; Pitcher, T J; MacDiarmid, A B; Byfield, T T; Tam, J C; Jones, T T; Bell, J J; Gardner, J P A (2014) Lobsters as keystone: only in unfished ecosystems? *Ecological Modelling 275*: 48– 72.

Edgar, G J; Stuart-Smith, R D; Willis, T J, et al. (2014) Global conservation outcomes depend on marine protected areas with five key features. *Nature 506*: 216–220. doi:10.1038/nature13022.

Engelhard, G H; Ellis, J R; Payne, M R; ter Hofstede, R; Pinnegar, J K (2011) Ecotypes as a concept for exploring responses to climate change in fish assemblages. *ICES Journal of Marine Science 68*: 580–591.

Essington, T E; Beaudreau, A H; Wiedenmann, J (2006) Fishing through marine food webs. *Proceedings of the National Academy of Sciences of the US 103*: 3171–3175.

Estes, J A (1996) The influence of large, mobile predators in aquatic food webs: examples from sea otters and kelp forests. *In*: Greenstreet, S; Tasker, M (Eds.), Aquatic Predators and their Prey. pp 65–72. Blackwell, London.

Estes, J A; Duggins, D O (1995) Sea otters and kelp forests in Alaska: generality and variation in a community ecological paradigm. *Ecological Monographs 65*: 75–100.

Estes, J A; Palmisano, J F (1974) Sea otters: their role in structuring nearshore communities. *Science 185*: 1058–1060.

Fabry, V J; Seibel, B A; Feely; R A; Orr, J C (2008) Impacts of ocean acidification on marine fauna and ecosystem processes. *ICES Journal of Marine Science 65*: 414–432.

FAO (2008) FAO Fisheries Technical Guidelines for Responsible Fisheries. No. 4, Suppl. 2, Add. 1. Rome, FAO. 78 p.

Fasham, M J R; Baliño, B M; Bowles, M C (Eds.) (2001) A new vision of ocean biogeochemistry after a decade of the Joint Global Ocean Flux Study (JGOFS). *Ambio Special Report 10*: 4–31.

Fisher, J A D; Frank, K T (2004) Abundance-distribution relationships and conservation of exploited marine fishes. *Marine Ecology Progress Series 279*: 201–213.

Folke, C; Carpenter, S; Walker, B; Scheffer, M; Elmqvist, T; Gunderson, L; Holling, C S (2004) Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology, Evolution and Systematics 35*: 557–581.

Forman, J S; Dunn, M R (2012) Diet and scavenging habits of the smooth skate *Dipturus innominatus*. *Journal of Fish Biology 80 (5)*: 1546–62. doi:10.1111/j.1095-8649.2012.03255.x.

Fossum, P (1992) The recovery of the Barents Sea capelin (*Mallotus villosus*) from a larval point of view. *ICES Journal of Marine Science 49*: 237–243.

Francis, R I C C; Hadfield, M G; Bradford-Grieve, J M; Renwick, J A; Sutton, P J H (2006) Links between climate and recruitment of New Zealand hoki (*Macruronus novaezelandiae*) now unclear. *New Zealand Journal of Marine and Freshwater Research 40*: 547–560.

Frank, K T; Petrie, B; Choi, J S; Leggett, W C (2005) Trophic cascades in a formerly cod-dominated ecosystem. *Science 308*: 1621–1623.

Frank, K T; Petrie, B; Shackell, N L (2007) The ups and downs of trophic control in continental shelf ecosystems. *Trends in Ecology and Evolution 22*: 237–242.

Frederiksen, M; Wanless, S; Harris, M P; Rothery, P; Wilson, L J (2004) The role of industrial fisheries and oceanographic change in the decline of North Sea black-legged kittiwakes. *Journal of Applied Ecology 41*: 1129–1139.

Freeman, D J; MacDiarmid, A B (2009) Healthier lobsters in a marine reserve: effects of fishing on disease incidence in the spiny lobster, *Jasus edwardsii*. *Marine and Freshwater Research 60*: 140–145.

Friedland, K D; Stock, C; Drinkwater, K F; Link, J S; Leaf, R T, et al. (2012) Pathways between primary production and fisheries yields of large marine ecosystems. *PLoS ONE 7 (1)*: e28945. doi:10.1371/journal.pone.0028945.

Fulton, E A (2011) Interesting times: winners, losers, and system shifts under climate change around Australia. *ICES Journal of Marine Science 68*: 1329–1343.

Fulton, E A; Smith, A D M; Punt, A E (2005) Which ecological indicators can robustly detect effects of fishing? *ICES Journal of Marine Science 62*: 540–551.

Fulton, E A; Smith, A D M; Webb, H; Slater, J (2004) Ecological indicators for the impacts of fishing on non-target species, communities and ecosystems: Review of potential indicators. *AFMA Final Research Report R99/1546*. 119 p.

Furness, R W (2002) Management implications of interactions between fisheries and sandeel-dependent seabirds and seals in the North Sea. *ICES Journal of Marine Science 59*: 261–269.

Furness, R W; Barrett, R T (1985) The food requirements and ecological relationships of a seabird community in North Norway. *Ornis Scandinavica 16*: 305–313.

Garcia, S M; Cochrane, K L (2005) Ecosystem approach to fisheries: a review of implementation guidelines. *ICES Journal of Marine Science 62*: 311–318.

Garcia, S M; Grainger, R J R (2005) Gloom and doom? The future of marine capture fisheries. *Philosophical Transactions Royal Society B 360*: 21–46.

Gecek, S; Legovic, T (2012) Impact of maximum sustainable yield on competitive community. *Journal of Theoretical Biology 307*: 96–103.

Ghosh, B; Kar, T K (2013) Possible ecosystem impacts of applying maximum sustainable yield policy in food chain models. *Journal of Theoretical Biology 329*: 6–14.

Gibbons, M J; Richardson, A J (2013) Beyond the jellyfish joyride and global oscillations: advancing jellyfish research. *Journal of Plankton Research 35 (5)*: 929–938.

Gilbert, D J; Annala, J H; Johnston, K (2000) Technical background to fish stock indicators for state-of-environment reporting in New Zealand. *Marine and Freshwater Research 51*: 451–464.

Grant, S; Constable, A; Raymond, B; Doust, S (2006) Bioregionalisation of the Southern Ocean: report of experts workshop. ACE-CRC and WWF Australia.

Greenstreet, S P R; Rogers, S I (2006) Indicators of the health of the fish community of the North Sea: identifying reference levels for an ecosystem approach to management. *ICES Journal of Marine Science 63*: 573–593.

Gruner, D S (2013) Trophic Cascades. John Wiley & Sons Ltd, Chichester.

Guttal, V; Jayaprakash, C (2008) Changing skewness: an early warning signal of regime shifts in ecosystems. *Ecology Letters 11*: 450–460.

Halpern, B S; Selkoe, K A; Micheli, F; Kappel, C V (2007) Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. *Conservation Biology 21*: 1301–1315.

Heath, M R (2005) Changes in the structure and function of the North Sea fish food web, 1973–2000, and the impacts of fishing and climate. *ICES Journal of Marine Science 62*: 847–868.

Heithaus, M R; Frid, A; Wirsing, A J; Worm, B (2008) Predicting the consequences of declines in marine top predators. *Trends in Ecology and Evolution 23 (4)*: 202–210.

Hewitt, J E; Julian, K; Bone, E K (2011) Chatham–Challenger Ocean Survey 20/20 Post-Voyage Analyses: Objective 10 – Biotic habitats and their sensitivity to physical disturbance. *New Zealand Aquatic Environment and Biodiversity Report No. 81*. 36 p.

Hilborn, R W (2007) Comment on Worm et al. *Science 316*: 1281–1282.

Hoeinghaus, D J; Zeug, S C (2008) Can stable isotope ratios provide for community-wide measures of trophic structure? Comment. *Ecology 89 (8)*: 2353.

Holling, C S (1973) Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics 4*: 1–23.

Hollowed, A B; Barange, M; Beamish, R J, et al. (2013) Projected impacts of climate change on marine fish and fisheries. *ICES Journal of Marine Science 70 (5)*: 1023–1037. doi:10.1093/icesjms/fst081.

Hollowed, A B; Hare, S R; Wooster, W S (2001) Pacific basin climate variability and patterns of Northeast Pacific marine fish production. *Progress in Oceanography 49*: 257–282.

Holt, C A; Punt, A E (2009) Incorporating climate information into rebuilding plans for overfished groundfish species of the U.S. west coast. *Fisheries Research 100*: 57–67.

Horn, P; Dunn, M R (2010) Inter-annual variability in the diets of hoki, hake, and ling on the Chatham Rise from 1990 to 2009. *New Zealand Aquatic Environment and Biodiversity Report No. 54*. 57 p.

Howes, L A; Montevecchi, W A (1993) Population trends and interactions among terns and gulls in Gros Morne National Park, Newfoundland. *Canadian Journal of Zoology 71*: 1516–1520.

Hrbácek, J; Dvorakova, M; Korinek, V; Prochazkova, L (1961) Demonstration of the effect of the fish stock on the species composition of zooplankton and the intensity of metabolism of the whole plankton assemblage. *Verhandlungen des Internationalen Verein Limnologie 14*: 192–195.

Hsieh, C-h; Glaser, S M; Lucas, A J; Sugihara, G (2005) Distinguishing random environmental fluctuations from ecological catastrophes for the North Pacific Ocean. *Nature 435*: 336–340.

Hsieh, C-h; Reiss, C S; Hunter, J R; Beddington, J R; May, R M; Sugihara, G (2006) Fishing elevates variability in the abundance of exploited species. *Nature 443*: 859–862.

Hughes, T P; Bellwood, D R; Folke, C; Steneck, R S; Wilson, J (2005) New paradigms for supporting the resilience of marine ecosystems. *Trends in Ecology and Evolution 20 (7)*: 380–386.

Hunt, G L; McKinnell, S (2006) Interplay between top-down, bottom-up, and wasp-waist control in marine ecosystems. *Progress in Oceanography 68*: 115–124.

Hurst, R J; Law, C S; Kennan, S C; Renwick, J A; Rickard, G J; Stewart, C; Sutton, P J H; Uddstrom, M J; Korpela, A; Evans, J (2008) Climate and oceanographic trends relevant to New Zealand fisheries. Final Research Report for Ministry of Fisheries project ENV200704. 92 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Hurst, R; Renwick, J A; Sutton, P; Uddstrom, M J; Kennan, S; Law, C; Rickard, G; Korpela, A; Stewart, C; Evans, J (2012) Climate and ocean trends of potential relevance to fisheries in the New Zealand region. *New Zealand Aquatic Environment and Biodiversity Report No. 90*. 90 p.

Hutchings, J A (2000) Collapse and recovery of marine fishes. *Nature 406*: 882–885.

ICES (2005) Ecosystem effects of fishing: Impacts, metrics, and management strategies. Rice, J C (Ed.). *ICES Cooperative Research Report No. 272*. 177 p.

IPCC (2007a) Climate Change 2007. Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Parry, M L; Canziani, O F; Palutikof, J P; van der Linden, P J; Hanson, C E (Eds.). Cambridge University Press, Cambridge, UK. 976 p.

IPCC (2007b) Climate Change 2007: the Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Solomon, S; Qin, D; Manning, M; Chen, Z; Marquis, M; Averyt, K B; Tignor, M; Miller, H L (Eds.). Cambridge University Press, Cambridge, UK. 996 p.

IUCN (2009) IUCN Red list of threatened species. Prepared by IUCN Species Survival Commission. IUCN, Gland, Switzerland.

Jackson, J B; Kirby, M X; Berger, W, et al. (2001) Historical overfishing and the recent collapse of coastal ecosystems. *Science 293*: 629–638.

Jahncke, J; Checkley, D; Hunt Jr G L (2004) Trends in carbon flux to seabirds in the Peruvian upwelling system: effects of wind and fisheries on population regulation. *Fisheries Oceanography 13*: 208–223.

Jarre-Teichmann, A; Wieland, K; MacKenzie, B; Hinrichsen, H-H; Plikhs, M; Aro, E (2002) Stock-recruitment relationships for cod (*Gadus morhua callarias*) in the central Baltic Sea incorporating environmental variability. *Archive of Fishery and Marine Research 48*: 97–123.

Jennings, S; Brander, K M (2010) Predicting the effects of climate variation and change on marine communities and the consequences for fisheries. *Journal of Marine Systems 79*: 418–426.

Jennings S; Greenstreet, S P R; Hill, L; Piet, G J; Pinnegar, J K; Warr, K J (2002) Long-term trends in the trophic structure of the North Sea fish community: evidence from stable-isotope analysis, size-spectra and community metrics. *Marine Biology 141*: 1085–1097.

Johnson, K D; Grabowski, J H; Smee, D L (2014) Omnivory dampens trophic cascades in estuarine communities. *Marine Ecology Progress Series 507*: 197–206.

Kaiser, M J (2005) Are marine protected areas a red herring or fisheries panacea? *Canadian Journal of Fisheries and Aquatic Sciences 62*: 1194–1199.

Kaiser, M J; Spencer, B E (1994) Fish scavenging behaviour in recently trawled areas. *Marine Ecology Progress Series 112*: 41–49.

Kennedy, V S; Twilley, R R; Kleypas, J A; Cowan Jr, J H; Hare, S R (2002) Coastal and marine ecosystems and global climate change: Potential effects on U.S. resources. Pew Center on Global Climate Change. 64 p.

Kidson, J W (2000) An analysis of New Zealand synoptic types and their use in defining weather regimes. *International Journal of Climatology 20*: 299–316.

Kirby, R R; Beaugrand, G (2009) Trophic amplification of climate warming. *Proceedings of the Royal Society B Biological Sciences 276*: 4095–4103.

Kirby, R R; Beaugrand, G; Lindley, J A (2008) Climate-induced effects on the merozooplankton and the benthic-pelagic ecology of the North Sea. *Limnology and Oceanography 53*: 1805–1815.

Kirby, R R; Beaugrand, G; Lindley, J A (2009) Synergistic effects of climate and fishing in a marine ecosystem. *Ecosystems 12*: 548–561.

Kjesbu, O S; Witthames, P R; Solemdal, P; Walker, M G (1998) Temporal variations in the fecundity of Arcto-Norwegian cod (*Gadus*

morhua) in response to natural changes in food and temperature. *Journal of Sea Research 40*: 303–321.

Köster, F W; Hinrichsen, H-H; Schnack, D, et al. (2003a) Recruitment of Baltic cod and sprat stocks: identification of critical life stages and incorporation of environmental variability into stock-recruitment relationships. *Scientia Marina 67 (supp.1)*: 129–154.

Köster, F W; Möllmann, C (2000) Trophodynamic control by clupeid predators on recruitment success in Baltic cod? *ICES Journal of Marine Science 57*: 311–323.

Köster, F W; Möllmann, C; Neuenfeldt, S, et al. (2003b) Fish stock development in the central Baltic Sea (1974–1999) in relation to variability in the environment. *ICES Marine Science Symposia 219*: 294–306.

Kotta, J; Kotta, I S; Pollupuu, M (2009) Separate and interactive effects of eutrophication and climate variables on the ecosystem elements of the Gulf of Riga. *Estuarine, Coastal and Shelf Science 84*: 509–518.

Kritzer, J P; Sale, P F (2004) Metapopulation ecology in the sea: from Levins' model to marine ecology and fisheries science. *Fish and Fisheries 5*: 131–140.

Law, R (2000) Fishing, selection, and phenotypic evolution. *ICES Journal of Marine Science 57*: 659–668.

Layman, C A; Quattrochi, J P; Peyer, C M; Allgeier, J E (2007) Niche width collapse in a resilient top predator following ecosystem fragmentation. *Ecology Letters 10*: 937–944.

Leathwick, J R; Francis, M; Julian, K (2006) Development of a demersal fish community map for New Zealand's Exclusive Economic Zone. *NIWA Client Report HAM2006-062*. 21 p.

Leathwick, J R; Rowden, A; Nodder, S; Gorman, R; Bardsley, S; Pinkerton, M; Baird, S J; Hadfield, M; Currie, K; Goh, A (2009) Development of a benthic-optimised marine environment classification for waters within the New Zealand EEZ. Final Research Report for Ministry of Fisheries project BEN200601. 52 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Leggett, W C; Frank, K T; Carscadden, J E (1984) Meteorological and hydrographic regulation of year-class strength in capelin (*Mallotus villosus*). *Canadian Journal of Fisheries and Aquatic Sciences 41*: 1193–1201.

Legovic, T; Gecek, S (2012) Impact of maximum sustainable yield on mutualistic communities. *Ecological Modelling 230*: 63–72.

Legovic, T; Klanjscek, J; Gecek, S (2010) Maximum sustainable yield and species extinction in ecosystems. *Ecological Modelling 221*: 1569–1574.

Lehodey, P; Alheit, J; Barange, M; Baumgartner, T; Beaugrand, G; Drinkwater, K; Fromentin, J-M; Hare, S; Ottersen, G; Perry, R I; Roy, C; van der Lingen, C D; Werner, F (2006) Climate variability, fish and fisheries. *Journal of Climate 19*: 5009–5030.

Levin, S A (1998) Ecosystems and the biosphere as complex adaptive systems. *Ecosystems 1*: 431–436.

Levin, S A (1999) Fragile Dominion: Complexity and the Commons. Perseus, Reading, MA.

Libralato, S; Christensen, V; Pauly, D (2006) A method for identifying keystone species in food web models. *Ecological Modelling 195*: 153–171.

Lilly, G R (1993) Predation by Atlantic cod on capelin on the southern Labrador and Northeast Newfoundland shelves during a period of changing spatial distributions. *In*: Jakobson, J, et al. (Eds.), Cod and climate change. pp 600–611. ICES Marine Science Symposium 198.

Lindegren, M; Diekmann, R; Mollman, C (2010) Regime shifts, resilience and recovery of a cod stock. *Marine Ecology Progress Series 402*: 239–253.

Lindeman, R L (1942) The trophic-dynamic aspect of ecology. *Ecology 23 (4)*: 399–418.

Lloyd, C S; Tasker, M L; Partridge, K (1991) The status of seabirds in Britain and Ireland. Poyser, London.

Lynam, C P; Gibbons, M J; Axelsen, B J, et al. (2006) Jellyfish overtake fish in a heavily fished ecosystem. *Current Biology 16 (13)*: R492–R493.

McCann, K S; Rooney, N (2009) The more food webs change, the more they stay the same. *Philosophical Transactions of the Royal Society of London B Biological Sciences 27*: 1789–1801. doi:10.1098/rstb.2008.0273.

McClatchie, S; Dunford, A (2003) Estimated biomass of vertically migrating mesopelagic fish off New Zealand. *Deep Sea Research I 50*: 1263–1281.

MacDiarmid, A B; Law, C S; Pinkerton, M H; Zeldis, J (2013) New Zealand marine ecosystem services. *In*: Dymond, J R (Ed.), Ecosystem services in New Zealand – conditions and trends. Manaaki Whenua Press, Lincoln, New Zealand.

MacDiarmid, A; McKenzie, A; Sturman, J; Beaumont, J; Mikaloff-Fletcher, S; Dunne, J (2012) Assessment of anthropogenic threats to New Zealand marine habitats. *New Zealand Aquatic Environment and Biodiversity Report No. 93*. 255 p.

McIlgorm, A; Hanna, S; Knapp, G; Le Floc'H, P; Millerd, F; Pan, M (2010) How will climate change alter fishery governance: insights from seven international case studies. *Marine Policy 34*: 170–177.

McKenzie, A (2013) Assessment of hoki (*Macruronus novaezelandiae*) in 2012. *New Zealand Fisheries Assessment Report 2013/27*. 65 p.

MacKenzie, B R; Leggett, W C (1991) Quantifying the contribution of small-scale turbulence to the encounter rates between larval fish and their zooplankton prey: effects of wind and tide. *Marine Ecology Progressive Series 73*: 149–160.

Mackinson, S; Daskalov, G; Heymans, J J; Neira, S; Arancibia, H; Zetina-Rejón, M; Jiang, H; Cheng, H Q; Coll, M; Arreguin-Sanchez, F; Keeble, K; Shannon, L (2009) Which forcing factors fit? Using ecosystem models to investigate the relative influence of fishing and changes in primary productivity on the dynamics of marine ecosystems. *Ecological Modelling 220 (21)*: 2972–2987.

McKnight, D G; Probert, P K (1997) Epibenthic communities on the Chatham Rise, New Zealand. *New Zealand Journal of Marine and Freshwater Research 31*: 505–513.

McQueen, D; Post, J (1988a) Limnocorral studies of cascading trophic interaction. *Verhandlungen - Internationale Vereinigung für Theoretische und Angewandte Limnologie 23*: 739–747.

McQueen, D; Post, J (1988b) Cascading trophic interactions: uncoupling at the zooplankton- phytoplankton link. *Hydrobiologia 159*: 277–296.

Marasco, R; Goodman, D; Grimes, C B; Lawson, P; Punt, A; Quinn, T (2007) Ecosystem-based fishery management; a pragmatic approach. *Canadian Journal of Fisheries and Aquatic Science 64*: 928–929.

Marine Stewardship Council (2010) MSC Fishery Standard: Principles and Criteria for Sustainable Fishing v1.1. Retrieved from http://www.msc.org/documents/scheme-documents/msc-standards.

Marine Zooplankton Colloquium 2 (2001) Future marine zooplankton research: a perspective. *Marine Ecology Progress Series 222*: 297–308.

Mayo, R K; O'Brien, L; Wiggley, S E (1998) Assessment of the Gulf of Maine cod stock for 1998. *Northeast Fisheries Science Center Reference Document 98–13*.

Mead, A; Griffiths, C L; Branch, G M; McQuaid, C D; Blamey, L K; Bolton, J J; Anderson, R J; Dufois, F; Rouault, M; Froneman, P W; Whitfield, A K; Harris, L R; Nel, R; Pillay, D; Adams J B (2013) Human-mediated drivers of change — impacts on coastal ecosystems and marine biota of South Africa. *African Journal of Marine Science 35*: 403–425.

Mehl, S; Sunnana, K (1991) Changes in growth of Northeast Arctic cod in relation to food consumption in 1984–1988. *ICES Journal of Marine Science 193*: 109–112.

Methratta, E T; Link, J S (2007) Evaluation of quantitative indicators for marine fish communities. *Ecological Indicators 6*: 575–588.

Micheli, F; Benedetti-Cecchi, L; Gambaccini, S; Bertocci, I; Borsini, C; Osio, G C; Romano, F (2005) Cascading human impacts, marine protected areas, and the structure of Mediterranean reef assemblages. *Ecological Monographs 75*: 81–102.

Mills, C E (2001) Jellyfish blooms: are populations increasing globally in response to changing ocean conditions? *Hydrobiologia 451*: 55–68.

Ministry for Primary Industries (2014) Fisheries Assessment Plenary, May 2014: stock assessments and stock status. Compiled by the Fisheries Science Group, Ministry for Primary Industries, Wellington, New Zealand. 1381 p.

Ministry of Fisheries (2008) Harvest Strategy Standard for New Zealand Fisheries. 30 p. Retrieved from http://fs.fish.govt.nz/Page.aspx?pk=104.

Möllmann, C; Diekmann, R; Müller-Karulis, B; Kornilovs, G; Plikshs, M; Axe, P (2009) Reorganization of a large marine ecosystem due to atmospheric and anthropogenic pressure: a discontinuous regime shift in the Central Baltic Sea. *Global Change Biology 15*: 1377–1393. doi:10.1111/j.1365-2486.2008.01814.x.

Möllmann, C; Müller-Karulis, B; Kornilovs, G; St. John, M A (2008) Effects of climate and overfishing on zooplankton dynamics and ecosystem structure regime shifts, trophic cascade and feedback loops in a simple ecosystem. *ICES Journal of Marine Science 65 (3)*: 302–310. doi:10.1093/icesjms/fsm197.

Mora, C; Andréfouët, S; Costello, M J; Kranenburg, C; Rollo, A; Veron, J; Gaston, K J; Myers, R A (2006) Coral reefs and the global network of marine protected areas. *Science 312*: 1750–1751.

Mullan, A B (1995) On the linearity and stability of Southern Oscillation-climate relationships for New Zealand. *International Journal of Climatology 15 (12)*: 1365–1386.

Mullan, A B; Wratt, D S; Renwick, J A (2001) Transient model scenarios of climate changes for New Zealand. *Weather and Climate 21*: 3–34.

Muradian, R (2001) Ecological thresholds: a survey. *Ecological Economics 38*: 7–24.

Murawski, S A (2000) Definitions of overfishing from an ecosystem perspective. *ICES Journal of Marine Science 57*: 649–658.

Murawski, S A (2011) Summing up Sendai: progress integrating climate change science and fisheries. *ICES Journal of Marine Science 68*: 1368–1372.

Murawski, S A; Idoine, J S (1992) Multispecies size composition: a conservative property of exploited fishery systems? *Journal of Northwest Atlantic Fisheries Sciences 14*: 79–85.

Murawski, S; Methot, R; Tromble, G (2007) Biodiversity loss in the ocean: how bad is it? *Science 316*: 1281–1284.

Murphy, E J; Wakins, J L; Reid, K; Trathan, P N; Everson, I; Croxall, J P; Priddle, J; Brandon, M A; Brierley, A S; Hoffman, E (1998) Inter-annual variability in the South Georgia marine ecosystem: biological and physical sources of variation in the abundance of krill. *Fisheries Oceanography 7*: 381–390.

Murphy, R J; Pinkerton, M H; Richardson, K M; Bradford-Grieve, J M; Boyd, P W (2001) Phytoplankton distributions around New Zealand

derived from SeaWiFS remotely-sensed ocean colour data. *New Zealand Journal of Marine and Freshwater Research, 35 (2)*: 343–362.

NEFSC (1998) Report of the 27th Northeast regional Stock Assessment Workshop Stock Assessment review Committee (SARC) consensus summary of assessments. *Northeast Fisheries Science Center Reference Document 98-15*.

Neubauer, P; Jensen, O P; Hutchings, J A; Baum, J K (2013) Resilience and recovery of overexploited marine populations. *Science 340*: 347–349.

Noakes, D J; Beamish, R J (2009) Synchrony of marine fish catches and climate and ocean regime shifts in the North Pacific Ocean. *Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 1*: 155–168.

Northcote, T (1988) Fish in the structure and function of freshwater ecosystems: a 'top-down' view. *Canadian Journal of Fisheries and Aquatic Sciences 45*: 361–379.

O'Connor, M I; Bruno, J F; Gaines, S D; Halpern, B S; Lester, S E; Kinlan, B P; Weiss, J M (2007) Temperature control of larval dispersal and the implications for marine ecology, evolution and conservation. *Proceedings of the National Academy of Sciences USA 104*: 1266–1271.

O'Driscoll, R L; Gauthier, S; Devine, J A (2009) Acoustic estimates of mesopelagic fish: as clear as day and night? *ICES Journal of Marine Science 66 (6)*: 1310–1317.

O'Driscoll, R L; MacGibbon, D; Fu, D; Lyon, W; Stevens, D W (2011) A review of hoki and middle depth trawl surveys on the Chatham Rise, January 1992–2010. *New Zealand Fisheries Assessment Report 2011/47*. 72 p.

OECD (2003) OECD Environmental Indicators: Development, measurement and use. Organization for Economic Cooperation and Development (OECD), Paris, France. 37 p.

Oeffner, J; Gauthier, S; Pinkerton, M H; O'Driscoll, R L (2014) A decision-tree model for species identification of acoustic marks: moving beyond acoustic indices to estimate mesopelagic biomass. Submitted to the Marine Ecology Progress Series.

Oguz, T; Gilbert, D (2007) Abrupt transitions of the top-down controlled Black Sea pelagic ecosystem during 1960–2000: Evidence for regime-shifts under strong fishery exploitation and nutrient enrichment modulated by climate-induced variations. *Deep Sea Research I 54*: 220–242.

Osterblom, H O; Hansson, S; Larsson, U; Hjerne, O; Wulff, F; Elmgren, R; Folke, C (2007) Human-induced trophic cascades and ecological regime shifts in the Baltic Sea. *Ecosystems 10*: 877–889.

Ottersen, G; Hjermann, D Ø; Stenseth, N C (2006) Changes in spawning stock structure strengthen the link between climate and recruitment in a heavily fished cod (*Gadus morhua*) stock. *Fisheries Oceanography 15*: 230–243.

Ottersen, G; Kim, S; Huse, G; Polovina, J J; Stenseth, N Chr. (2010) Major routes by which climate may force marine fish populations. *Journal of Marine Systematics 79*: 343–360.

Overland, J; Alheit, J; Bakun, A; Hurrell, J; Mackas, D; Miller, A (2010) Climate controls on marine ecosystems and fisheries. *Journal Marine Systematics 79*: 305–315.

Pace, M L; Cole, J J; Carpenter, S R; Kitchell, J F (1999) Trophic cascades in diverse ecosystems. *Trends in Ecology and Evolution 14*: 483–488.

Paine, R T (1980) Food web linkage, interaction strength and community infrastructure. *Journal of Animal Ecology 49*: 667–685.

Parsons, T R; Lalli, C M (2002) Jellyfish population explosions: Revisiting a hypothesis of possible causes. *La Mer 40*: 111–121.

Parsons, T R; Takahashi, M; Hargrave, B (1977) Biological oceanographic processes. Pergamon Press, Oxford. 332 p.

Pascual, M; Dunne, J A (2006) Ecological Networks: Linking Structure to Dynamics in Food webs. Oxford University Press. 371 p.

Pauly, D; Christensen, V (1995) Primary production required to sustain global fisheries. *Nature 374*: 225–257.

Pauly, D; Christensen, V; Dalsgaard, J; Froese, R; Torres, F (1998a) Fishing down marine food webs. *Science 279*: 860–863.

Pauly, D; Froese, R; Christensen, V (1998b) How pervasive is 'Fishing down marine food webs': response to Caddy et al. *Science 282*: 1383.

Pauly, D; Watson, R (2005) Background and interpretation of the 'Marine Trophic Index' as a measure of biodiversity. *Philosophical Transactions of the Royal Society B 360*: 415–423.

Pearcy, W; Fisher, J; Brodeur, R D; Johnson, S (1985) Effects of the 1983 El Nino on coastal nekton off Oregon and Washington. *In*: Wooster, W S; Fluharty, D L (Eds.), El Nino North. pp 188–204. Washington Sea Grant Program, University of Washington, Seattle, USA.

Peet, R K (1974) The measurement of species diversity. *Annual Review of Ecology and Systematics 5*: 285–307.

Perry, A L; Low, P J; Ellis, J R; Reynolds, J D (2005) Climate change and distribution shifts in marine fishes. *Science 308*: 1912–1915.

Perry, R I; Cury, P; Brander, K; Jennings, S; Möllmann, C; Planque, B (2010) Sensitivity of marine systems to climate and fishing: concepts, issues and management responses. *Journal of Marine Systems 79*: 427–435.

Petitgas, P; Rijnsdorp, A D; Dickey-Collas, M; Engelhard, G H; Peck, M A; Pinnegar, J K; Drinkwater, K; Huret, M; Nash, R D M (2013) Impacts of climate change on the complex life cycles of fish. *Fisheries Oceanography 22*: 121–139.

Pimm, S L (1982) Food webs. Chapman and Hall, London.

Pinkerton, M H (2007) Oceanic Primary Productivity in the New Zealand EEZ. *In*: State of the New Zealand Environment: Oceans. Report ENZ-07, Ministry for the Environment, New Zealand.

Pinkerton, M H (2010) Headline indicators for the New Zealand ocean. NIWA report. 42 p. Retrieved from www.niwa.co.nz/sites/niwa.co.nz/files/headline_indicators_59.pdf.

Pinkerton, M H (2012) Trophic modelling of a New Zealand marine shelf ecosystem since AD 1000. Draft final research report to Ministry for Primary Industries. 342 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Pinkerton, M H (2013) Ecosystem Modelling of the Chatham Rise. *NIWA Client Report WLG2013-17*, prepared for Chatham Rock Phosphate. 183 p.

Pinkerton, M H (2014a) Three potential Marine State statistics for the 2015 National Environment Reporting Synthesis Report: (1) Marine Trophic Index; (2) Acoustic Backscatter Index; (3) Zooplankton Index. NIWA unpublished report for Ministry for the Environment. 13 p.

Pinkerton, M H (2014b) Food web modelling of the Chatham Rise: Additional work requested by the expert conference on ecosystem effects. Letter to CRP and DMC, 14 October 2014. 23 p.

Pinkerton, M H; Bell, R; Chiswell, S M; Currie, K; Mullan, A B; Rickard, G; Stevens, C; Sutton, P (2014) Reporting on the state of the marine environment: recommendations for tier 1 oceanic statistics. Draft report for project ZBD2012-02. 82 p.

Pinkerton, M H; Bradford-Grieve, J M (2014) Characterizing food web structure to identify potential ecosystem effects of fishing in the Ross Sea, Antarctica. *ICES Journal of Marine Science 71 (7)*: 1542–1553. doi:10.1093/icesjms/fst230.

Pinkerton, M H; Lundquist, C J; Duffy, C A J; Freeman, D J (2008) Trophic modelling of a New Zealand rocky reef ecosystem using simultaneous adjustment of diet, biomass and energetic parameters. *Journal of Experimental Marine Biology and Ecology 367*: 61–266.

Pinsky, M L; Jensen, O P; Ricard, D; Palumbi, S R (2011) Unexpected patterns of fisheries collapse in the world's oceans. *Proceedings of the National Academy of Sciences of the US, 108 (20)*: 8317–8322. doi:10.1073/pnas.1015313108.

Pitcher, T J; Kalikoski, D; Short, K; Varkey, D; Pramoda, G (2009) An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. *Marine Policy 33*: 223–232.

Plagányi, E E (2007) Models for an ecosystem approach to fisheries. *FAO Fisheries Technical Paper 477*. FAO, Rome.

Plagányi, E E; Punt, A E; Hillary, R, et al. (2014) Multispecies fisheries management and conservation: tactical applications using models of intermediate complexity. *Fish and Fisheries 15 (1)*: 1–22.

Polunin, N V C; Pinnegar, J K (2002) Trophic ecology and the structure of food webs. *In*: Hart, P J B; Reynolds, J D (Eds.), Handbook of Fish and Fisheries. Volume 1. Blackwell, Oxford, UK.

Pope, J G; Knights, B J (1982) Comparison of the length distributions of combined catches of all demersal fishes in surveys in the North Sea and at Faroe Bank. *In*: Mercer, M C (Ed.), Multispecies approaches to fisheries management advice. pp 116–118. *Canadian Special Publication of Fisheries and Aquatic Sciences 59*.

Pope, J G; Stokes, T K; Murawski, S A; Idoine, J S (1987) A comparison of fish size composition in the North Sea and on Georges Bank. *In*: Wolff, W; Soeder, C-J; Drepper, F R (Eds.), Ecodynamics – Contributions to theoretical ecology. pp 146–152. Springer-Verlag, Berlin.

Portner, H O; Peck, M A (2010) Climate change effects on fishes and fisheries: towards a cause-and-effect understanding. *Journal of Fish Biology 77*: 1745–1779.

Probert, P K; Grove, S L; McKnight, D G; Read, G B (1996) Polychaete distribution on the Chatham Rise, Southwest Pacific. *Internationale Revue der Gesamten Hydrobiologie 81*: 577–588.

Purcell, J E (2005) Climate effects on formation of jellyfish and ctenophore blooms: A review. *Journal of the Marine Biological Association of the UK 85 (3)*: 461–476.

Purcell, J E (2012) Jellyfish and ctenophore blooms coincide with human proliferations and environmental perturbations. *Annual Review of Marine Science 4*: 209–235.

Reid, K; Croxall, J P; Briggs, D R; Murphy, E J (2005) Antarctic ecosystem monitoring: quantifying the response of ecosystem indicators to variability in Antarctic krill. *ICES Journal of Marine Science 62 (3)*: 366–373.

Reid, P C; Edwards, M; Beaugrand, G; Skogen, M D; Stevens, D (2003) Periodic changes in the zooplankton of the North Sea during the 20th century linked to oceanic inflow. *Fisheries Oceanography 12*: 260–269.

Reid, P C; Edwards, M; Hunt, H G; Warner, A J (1998) Phytoplankton changes in the North Atlantic. *Nature 391*: 546.

Rice, J (1995) Food web theory, marine food webs, and what climate change may do to northern fish populations. *In*: Beamish, R J (Ed.), Climate Change and Northern Fish Populations. *Canadian Special Publication in Fisheries and Aquatic Science 121*: 561–568.

Rice, J (2001) Implications of variability on many time scales for scientific advice on sustainable management of living marine resources. *Progress in Oceanography 49*: 189–209.

Rice, J (2003) Environmental health indicators. *Ocean & Coastal Management 46 (3–4)*: 235–259.

Rice, J C; Garcia, S M (2011) Fisheries, food security, climate change, and biodiversity: characteristics of the sector and perspectives on

emerging issues. *ICES Journal of Marine Science 68 (6)*: 1343–1353.

Rice, J; Gislason, H (1996) Patterns of change in the size spectra of numbers and diversity of the North Sea fish assemblage, as reflected in surveys and models. *ICES Journal of Marine Science 53*: 1214–1225.

Rice, J C; Rochet, M-J (2005) A framework for selecting a suite of indicators for fisheries management. *ICES Journal of Marine Science 62*: 516–527.

Richardson, A J (2008) In hot water: zooplankton and climate change. *ICES Journal of Marine Science 65 (3)*: 279–295.

Ridgeway, G (2006) Generalized Boosted Models: A guide to the gbm package. Gbm library for R.

Rijnsdorp, A D; Peck, M A; Engelhard, G H; Möllmann, C; Pinnegar, J K (2009) Resolving the effect of climate change on fish populations. *ICES Journal Marine Science 66*: 1570–1583.

Robinson, K V; Pinkerton, M H; Hall, J A; Hosie, G W (2013) Continuous Plankton Recorder Time Series. *New Zealand Aquatic Environment and Biodiversity Report No. 128*. 76 p.

Rochet, M-J; Rice, J (2005) Do explicit criteria help in selecting indicators for ecosystem-based fisheries management? *ICES Journal of Marine Science 62*: 528–539.

Rooney, N; McCann, K S; Gellner, G; Moore, J C (2006) Structural asymmetry and the stability of diverse food webs. *Nature 442*: 265–269.

Rothschild, B J (1994) Decadal transients in biological productivity, with special reference to cod populations of the North Atlantic. *ICES Marine Science Symposia 198*: 333–345.

Salomon, A K; Gaichas, S K; Shears, N T; Smith, J E; Madin, E M P; Gaines, S D (2010) Key features and context-dependent effects of fishery-induced trophic cascades. *Conservation Biology 24*: 382–394.

Salomon, A K; Shears, N T; Langlois, T J; Babcock, R C (2008) Cascading effects of fishing can alter carbon flow through a temperate coastal ecosystem. *Ecological Applications 18*: 1874–1887.

Scheffer, M; Carpenter, S R (2003) Catastrophic regime shifts in ecosystems: linking theory to observation. *Trends Ecology Evolution 18*: 648–656.

Schiel, D R (2013) The other 93%: trophic cascades, stressors and managing coastlines in non-marine protected areas. *New Zealand Journal of Marine and Freshwater Research 47*: 374–391. doi:10.1080/00288330.2013.810161.

Schiermeier, Q (2004) Climate findings let fishermen off the hook. *Nature 428*. doi:10.1038/428004a.

Schroeter, S C; Dixon, J D; Kastendiek, J; Smith, R O; Bence, J R (1993) Detecting the ecological effects of environmental impacts: a case study of kelp forest invertebrates. *Ecological Applications 3 (2)*: 331–350.

Shannon, L J; Coll, M; Neira, S (2009) Exploring the dynamics of ecological indicators using food web models fitted to time series of abundance and catch data. *Ecological Indicators 9*: 1078–1095.

Shapiro, J; Lammara, V; Lynch, M (1975) Biomanipulation: an ecosystem approach to lake restoration. *In*: Brezonik, P L; Fox, J L (Eds.), Water quality management through biological control. pp 85–96. University of Florida.

Sharp, B; Pinkerton, M H; Smith, A; Leathwick, J (2007) Marine Classification: Lessons from the New Zealand Experience. CCAMLR document WS-BSO-07-06, Hobart, Australia. CCAMLR Bioregionalisation of the Southern Ocean Experts workshop, 13–17 August 2007, Brussels.

Shears, N T; Babcock, R C (2002) Marine reserves demonstrate top-down control of community structure on temperate reefs. *Oecologia 132*: 131–142.

Shears, N T; Babcock, R C (2003) Continuing trophic cascade effects after 25 years of no-take marine reserve protection. *Marine Ecology Progress Series 246*: 1–16.

Shears, N T; Babcock, R C; Salomon, A K (2008) Context-dependent effects of fishing: variation in trophic cascades across environmental gradients. *Ecological Applications 18*: 1860–1873.

Shields, J D (2011) Diseases of spiny lobsters: a review. *Journal of Invertebrate Pathology 106*: 79–91.

Shiganova, T A (1998) Invasion of the Black Sea by the ctenophore Mnemiopsis leidyi and recent changes of pelagic community structure. *Fishery Oceanography 7*: 305–310.

Shin Y-J; Cury, P (2001) Exploring fish community dynamics through size-dependent trophic interactions using a spatialized individual based model. *Aquatic Living Resources 14 (2)*: 65–80. doi:10.1051/alr:2005001.

Shin, Y-J; Shannon, L J; Cury, P M (2004) Simulations of fishing effects on the southern Benguela fish community using an individual based model: learning from a comparison with ECOSIM. *In*: Shannon, L J; Cochrane, K L; Pillar, S C (Eds.), Ecosystem Approaches to Fisheries in the Southern Benguela. *African Journal of Marine Science 26*: 95–114.

Shurin, J B; Borer, E T; Seabloom, E W; Anderson, K; Blanchette, C A; Broitman, B R; Cooper, S D; Halpern, B (2002) A cross-ecosystem comparison of the strength of trophic cascades. *Ecology Letters 5*: 785–791.

Smetacek, V; Assmy, P; Henjes, J (2004) The role of grazing in structuring Southern Ocean pelagic ecosystems and biogeochemical cycles. *Antarctic Science 16 (4)*: 541–558.

Smith, B E; Lucey, S M (2014) Using fish diets as ecosystem indicators: are fish feeding down the food web on Georges Bank?

Proceedings of the 144th American Fisheries Society annual meeting, 17–21 August 2014, Quebec, Canada.

Smith, S G; Aulta, J S; Bohnsack, J A; Harper, D E; Luoa, J; McClellan, D B (2011) Multispecies survey design for assessing reef-fish stocks, spatially explicit management performance, and ecosystem condition. *Fisheries Research 109*: 25–41.

Snelder, T; Leathwick, J; Dey, K; Weatherhead, M; Fenwick, G; Francis, M (2005) The New Zealand marine environment classification. Ministry for the Environment.

Steele, J (1998) From carbon flux to regime shift. *Fisheries Oceanography 7*: 176–181.

Steele, J H; Schumacher, M (2000) Ecosystem structure before fishing. *Fisheries Research 44*: 201–205.

Stock, C; Dunne, J (2010) Controls on the ratio of mesozooplankton production to primary production in marine ecosystems. *Deep Sea Research I 57 (1)*: 95–112.

Sutton, P (2001) Detailed structure of the Subtropical Front over Chatham Rise, east of New Zealand. *Journal of Geophysical Research 106 (C12)*: 31045–31056.

Sydeman, W J; Bograd, S J (2009) Marine ecosystems, climate and phenology: introduction. *Marine Ecological Progress Series 393*: 185–188.

Szpak, P; Orchard, T J; Salomon, A K; Gröcke, D R (2013) Regional ecological variability and impact of the maritime fur trade on nearshore ecosystems in southern Haida Gwaii (British Columbia, Canada): evidence from stable isotope analysis of rockfish (Sebastes spp.) bone collagen. *Archaeological and Anthropological Sciences 5*: 159–182.

Taggert, C T; Leggett, W C (1987) Short-term mortality in post-emergent larval capelin Mallotus villosus. 1. Analysis of multiple in situ estimates. *Marine Ecology Progressive Series 41*: 205–217.

Taylor, A H (2002) North Atlantic climatic signals and the plankton of the European continental shelf. *In*: Sherman, K; Skjoldal, H R (Eds.), Changing states of the large marine ecosystems of the North Atlantic. pp 3–26. Elsevier Science, Amsterdam.

Taylor, G A (2000a) Action Plan for Seabird Conservation in New Zealand. Part A: Threatened Seabirds. Threatened Species Occasional Publication 16. 234 p.

Taylor, G A (2000b) Action Plan for Seabird Conservation in New Zealand. Part B: Non-Threatened Seabirds. *Threatened Species Occasional Publication 17*. 201 p.

ter Hofstede, R; Hiddink, J G; Rijnsdorp, A D (2010) Regional warming changes the species richness of marine fish in the eastern North Atlantic Ocean. *Marine Ecology Progress Series 414*: 1–9.

Theissen, H (1986) Population dynamics of gulls (Laridae) and terns (Sternidae) in Schleswig-Holstein – and thoughts on the so-called gull-problem. *Seevogel 7*: 1–12.

Thrush, S F; Dayton, P K (2002) Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. *Annual Review of Ecology and Systematics 33*: 449–473.

Travers, M; Shin, Y-J; Jennings, S; Machu, E; Huggett, J A; Field, J G; Cury P M (2009) Two-way coupling versus one-way forcing of plankton and fish models to predict ecosystem changes in the Benguela. *Ecological Modelling 220 (21)*: 3089–3099.

Trenkel, V; Berger, L (2013) A fisheries acoustic multi-frequency indicator to inform on large scale spatial patterns of aquatic pelagic ecosystems. *Ecological Indicators 30*: 72–79.

Trillmich, F; Ono, K A; Costa, D P; DeLong, R L; Feldkamp, S D; Francis, J M; Gentry, R L; Heath, C B; LeBoeuf, B J; Majluf, P; York, A E (1991) The effects of El Nino on pinniped populations in the eastern Pacific. *In*: Trillmich, F F; Ono, K A (Eds.), Pinnipeds and El Nino: Responses to Environmental Stress. pp 247–270. Springer-Verlag, Berlin.

Tuck, I; Cole, R; Devine, J A (2009) Ecosystem indicators for New Zealand fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 42*. 180 p.

Tuck, I; Pinkerton, M H; Tracey, D; Anderson, O A; Chiswell, S M (2014) Ecosystem and Environmental Indicators for Deepwater Fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 127*. 143 p.

Ulanowicz, R E; Puccia, C J (1990) Mixed trophic impacts in ecosystems. *Coenoses 5*: 7–16.

Valdes, L; Peterson,W; Church, J; Brander, K; Marcos, M (2009) Our changing oceans: conclusions of the first international symposium on the effects of climate change on the world's oceans. *ICES Journal of Marine Science 66*: 1435–1438.

van Nes, E H; Scheffer, M (2007) Slow recovery from pertubations as a genetic indicator of a nearby catastrophic shift. *American Naturalist 169*: 738–747.

Vaquer-Sunyer, R; Duarte, C M (2008) Thresholds of hypoxia for marine biodiversity. *Proceedings of the National Academy of the Sciences USA 105 (40)*: 15452–15457.

Walker, B; Holling, C S; Carpenter, S R; Kinzig, A (2004) Resilience, adaptability and transformability in social–ecological systems. *Ecology and Society 9 (2)*: 5.

Walters, C J; Christensen, V; Martell, S J; Kitchell, J F (2005) Possible ecosystem impacts of applying MSY policies from single-species assessment. *ICES Journal of Marine Science 62*: 558–568.

Walters, C J; Kitchell, J F (2001) Cultivation/depensation effects on juvenile survival and recruitment: implications for the theory of fishing. *Canadian Journal Fisheries and Aquatic Science 58*: 1–12.

Wanless, S (1988) The recolonisation of the Isle of May by common and arctic terns. *Scottish Birds 15*: 1–8.

Wanless, S; Harris, M P; Calladine, J; Rothery, P (1996) Modelling responses of herring gull and lesser black-backed gull populations to reduction of reproductive output: implications for control measures. *Journal of Applied Ecology 33*: 1420–1432.

Warwick, R; Clarke, K R (1995) New 'biodiversity' measures reveal a decrease in taxonomic distinctness with increasing stress. *Marine Ecology Progress Series 129*: 301–305.

Weijerman, M; Lindeboom, H J; Zuur, A (2005) Regime shifts in marine ecosystems of the North Sea and Wadden Sea. *Marine Ecology Progress Series 289*: 21–39.

Winder, M; Schindler, D E (2004) Climate change uncouples trophic interactions in an aquatic ecosystem. *Ecology 85 (8)*: 2100–2106.

Winters, G H; Wheeler, J P (1994) Length-specific weight as a measure of growth success of adult Atlantic herring (*Clupea harengus*). *Canadian Journal of Fisheries and Aquatic Sciences 51*: 1169–1179.

Worm, B; Hilborn, R; Baum, J K, et al. (2009) Rebuilding global fisheries. *Science 325 (5940)*: 578–585.

Worm, B; Myers, R A (2003) Meta-analysis of cod–shrimp interactions reveals top-down control in oceanic food webs. *Ecology 84*: 162–173.

Zeldis, J; Hadfield, M; Booker, D (2013) Influence of climate on Pelorus Sound mussel aquaculture yields: correlations and underlying mechanisms. *Aquaculture Environment Interactions 4*: 1–15. doi:10.3354/aei00066.

Zeldis, J R; Howard-Williams, C; Carter, C M; Schiel, D R (2008) ENSO and riverine control of nutrient loading, phytoplankton biomass and mussel aquaculture yield in Pelorus Sound, New Zealand. *Marine Ecology Progress Series 371*: 131–142.

14 HABITATS OF PARTICULAR SIGNIFICANCE FOR FISHERIES MANAGEMENT (HPSFM)

Status of chapter	This chapter has not been updated for AEBAR 2018.
Scope of chapter	This chapter highlights subject areas that might contribute to the management of HPSFM and hence provides a guide for future research.
Area	All of the New Zealand EEZ and Territorial Sea (inclusive of the freshwater and estuarine areas).
Locality hotspots	None formally defined, but already identified likely candidates include areas of biogenic habitat, e.g., Separation Point and Wairoa Hard, and areas identified with large catches and/or vulnerable populations of juveniles, e.g., Hoki Management Areas, packhorse crayfish legislated closures and toheroa beaches.
Key issues	Identifying likely HPSFM and potential threats to them.
Emerging issues	Connectivity and intra-population behaviour variability, multiple use.
MPI research (current)	HAB2007/01 *Biogenic habitats as areas of particular significance for fisheries management*; TOH2007/03 *Toheroa abundance*; ZBD2008/01 *Research on Biogenic Habitat-Forming Biota and their functional role in maintaining Biodiversity in the Inshore Region (5-150M Depths)* – this is also part-funded by Oceans Survey 2020, NIWA and MBIE; ENV2009/07 *Habitats of particular significance for fisheries management: Kaipara Harbour*; ENV2010/03 *Habitats of particular significance for inshore finfish fisheries management*; GMU2009/01 *Spatial Mixing of GMU1 using Otolith Microchemistry*.
NZ government research (current)	Ministry of Business, Innovation and Employment (MBIE) funded programmes (Coastal Conservation Management: C01X0907 *Protecting the functions of marine coastal habitats that support fish assemblages at local, regional and national scales*; C01X1229 *Predicting the occurrence of vulnerable marine ecosystems for planning spatial management in the South Pacific region*; C01X0906 *Impacts of resource use on vulnerable deep-sea communities*. NIWA Core funding in the 'Managing marine stressors' area under the 'Coasts and Oceans' centre, specifically the programme 'Managing marine resources' and the project 'Measuring mapping and conserving (C01X0505)'.
Related chapters/issues	Land-based impacts on fisheries and supporting biodiversity, bycatch composition, marine environmental monitoring.

14.1 CONTEXT

The Fisheries Act 1996, in Section 9 (Environmental principles) states that:

'All persons exercising or performing functions, duties, or powers under this Act, in relation to the utilisation of fisheries resources or ensuring sustainability, shall take into account the following environmental principles:

a. Associated or dependent species should be maintained above a level that ensures their long-term viability:
b. Biological diversity of the aquatic environment should be maintained:
c. Habitat of particular significance for fisheries management should be protected.'

Work is currently ongoing on a guidance document for implementing habitats of particular significance for fisheries management (HPSFM).

This chapter will focus on examples of habitats shown to be important for fisheries and concepts likely to be important to HPSFM. Examples of potential HPSFM include: sources of larvae; larval settlement sites; habitat for juveniles; habitat that supports important prey species; migration corridors; and spawning, pupping or egg-laying grounds. Some of these habitats may be important for only part of the life cycle of an organism, or for part of a year.

The relative importance of habitats, compared with other limiting factors, is largely unknown for most stocks. For example, some stocks may be primarily habitat limited, whereas others may be limited by oceanographic variability, food supply, predation rates (especially during juvenile phases), or a mixture of these and other factors. In the case of stocks that are habitat limited, a management goal might be to preserve or improve some aspect of the habitat for the stock.

Hundreds of legislated spatial fisheries restrictions already apply within New Zealand's Territorial Sea and Exclusive Economic Zone (www.nabis.govt.nz), but until further policy work and research is conducted we cannot be sure of what contribution they make to protecting HPSFM. Examples of these are listed below:

- Separation Point in Tasman Bay, and the Wairoa Hard in Hawke Bay, were created to protect biogenic habitat that was believed to be important as juvenile habitat for a variety of fish species (Grange et al. 2003).
- An area near North Cape is currently closed to packhorse lobster fishing to mitigate sub-legal handling disturbance in this area. This closure was established because of the small size of lobsters caught there and a tagging study that showed movement away from this area into nearby fished areas (Booth 1979).
- The largest legislated closures are the Benthic Protection Areas (BPAs) that protect about 1.2 million km^2 (about 31% of the EEZ) outside the Territorial Sea from contact of trawl and dredge gear with the bottom (Helson et al. 2010).
- Commercial fishers must not use New Zealand fishing vessels or foreign-owned New Zealand fishing vessels over 46 m in overall length for trawling in the Territorial Sea.

In addition to legislated closures, a number of non-regulatory management measures exist. For example:

- Spatial closures:
  - Trawlers greater than 28 m in length are excluded from targeting hoki in four Hoki Management Areas – Cook Strait, Canterbury Banks, Mernoo Bank, and Puysegur Bank (Deep Water Group 2008). These areas were chosen because of the larger number of juveniles caught, relative to adults in these areas.
  - Trawling and pair trawling are both closed around Kapiti Island.
- Seasonal closures:
  - A closure to trawling exists from 1 November until 30 April each year in Tasman Bay.
  - A closure to commercial potting exists for all of CRA 3 for the whole of the month of December each year.

The highly migratory fish plan addresses HPSFM in environment outcome 8.1 'Identify and where appropriate protect habitats of particular significance to highly migratory species, especially within New Zealand waters'. In the deepwater fish plan the Ministry proposes in Management Objective 2.3 'to develop policy guidelines to determine what constitutes HPSFM then apply these policy guidelines to fisheries where necessary'. Inshore fisheries management plans (freshwater, shellfish and finfish) all contain references to identifying and managing HPSFM. These plans recognise that not all impacts stem from fisheries activities, therefore managing them may include trying to influence others to better manage their impacts on HPSFM. Work is underway on a guidance document for HPSFM that will assist in implementing these outcomes and objectives.

14.2 GLOBAL UNDERSTANDING

This section focuses upon those habitats protected overseas for their value to fisheries and discusses important concepts that may help gauge the importance of any particular habitat to fisheries management. This information may guide future research into HPSFM in New Zealand and any subsequent management action.

14.2.1 HABITATS PROTECTED ELSEWHERE FOR FISHERIES MANAGEMENT

Certain habitats have been identified as important for marine species including: shallow sea grass meadows, wetlands, seaweed beds, rivers, estuaries, rhodolith beds, rocky reefs, crevices, boulders, bryozoans, submarine canyons, seamounts, coral reefs, shell beds and shallow bays or inlets (Kamenos et al. 2004, Caddy 2008, Clark 1999, Morato et al. 2010a). Discrete habitats (or parts of these) may have extremely important ecological functions, and/or be especially vulnerable to degradation. For example,

seabeds with high roughness are important for many fisheries and can be easily damaged by interaction with fishing gear (Caddy 2008). Examples of these include:

1. The *Oculina* coral banks off Florida were protected in 1994 as an experimental reserve in response to their perceived importance for reef fish populations (Rosenberg et al. 2000). Later studies confirmed that this area is the only spawning aggregation site for gag (*Mycteroperca microlepis*) and scamp (*M. phenax*) (both groper species), and other economically important reef fish in that region (Koenig et al. 2000). The size of the area within which bottom-tending gears were restricted was subsequently increased based on these findings (Rosenberg et al. 2000).
2. Lophelia cold-water coral reefs are now protected in at least Norway (Fosså et al. 2002), Sweden (Lundälv & Jonsson 2003) and the United Kingdom (European Commission 2003) due to their importance as habitat for many species of fish (Costello et al. 2005).
3. The Western Pacific Regional Fishery Management Council identified all escarpments between 40 m and 280 m as Habitat Areas of Particular Concern (HAPC) for species in the bottom-fish assemblage. The water column to a depth of 1000 m above all shallow seamounts and banks was categorised as HAPC for pelagic species. Certain north-west Hawaiian Island banks shallower than 30 m were categorised as HAPC for crustaceans, and certain Hawaiian Island banks shallower than 30 m were classified as Essential Fish Habitat (EFH) for precious corals. Fishing is closely regulated in the precious-coral EFH, and harvest is only allowed with highly selective gear types that limit impacts, such as manned and unmanned submersibles (Western Pacific Fishery Management Council 1998)

Examples of habitats protected for their freshwater fishery values also exist. For example, the US Atlantic States Interstate fishery management plan (Atlantic States Marine Fisheries Commission 2000) notes the Sargasso Sea is important for spawning, and that seaweed harvesting provides a threat of unknown magnitude to eel spawning. Habitat alteration and destruction are also listed as probably impacting on continental shelves and estuaries/rivers, respectively, but the extent to which these are important is unknown.

It is also possible that HPSFM may be defined by the functional importance of an area to the fishery. For example, large spawning aggregations can happen in midwater for set periods of time (Schumacher & Kendall 1991, Livingston 1990) these could also potentially qualify as HPSFM.

14.2.2 CONCEPTS POTENTIALLY IMPORTANT FOR HPSFM

Many nations are now moving towards formalised habitat classifications for their coastal and ocean waters, which may include fish dynamics in the classifcation, and could potentially help to define HPSFM. Such systems help provide formal definitions for management purposes, and to 'rank' habitats in terms of their relative values and vulnerability to threats. Examples include the Essential Fish Habitat (EFH) framework being advanced in North America (Benaka 1999, Diaz et al. 2004, Valavanis et al. 2008), and in terms of habitat, the developing NOAA Coastal and Marine Ecological Classification Standard for North America (CMECS) (Madden et al. 2005, Keefer et al. 2008), and the European Marine Life Information Network (MarLIN) framework, which has developed habitat classification and sensitivity definitions and rankings (Hiscock & Tyler-Walters 2006).

Habitat connectivity (the movement of species between habitats) operates across a range of spatial scales, and is a rapidly developing area in the understanding of fisheries stocks. These movements link together different habitats into 'habitat chains', which may also include 'habitat bottlenecks', where one or more spatially restricted habitats may act to constrain overall fish production (Werner et al. 1984). Human-driven degradation or loss of such bottleneck habitats may strongly reduce the overall productivity of populations, and hence ultimately reduce long-term sustainable fisheries yields. The most widely studied of these links is between juvenile nursery habitats and often spatially distant adult population areas. Most studies published have been focused on species that use estuaries as juveniles (e.g., blue grouper *Achoerodus viridis* (a large wrasse) (Gillanders & Kingsford 1986) and snapper *Pagrus auratus* (Hamer et al. 2005) in Australia; and gag (*Mycteroperca microlepis*) in the United States (Ross &

Moser 1995)), which make unidirectional ontogenetic habitat shifts from estuaries and bays out to the open coast as they grow from juveniles to adults. The extent of wetland habitats in the Gulf of Mexico has also been linked to the yield of fishery species dependent on coastal bays and estuaries. Reduced fishery stock production (of shrimp and the fish menhaden) followed wetland losses and, conversely, stock gains followed increases in the area of wetlands (Turner & Boesch 1987). Juvenile production was limited by the amount of available habitat but, equally, reproduction, larval settlement, juvenile or adult survivorship, or other demographic factors could also be limited by habitat loss or degradation, and these could have knock-on effects to stock characteristics such as productivity and its variability. Other examples include movements that may be bidirectional and regular in nature e.g., seasonal migrations of adult fish to and from spawning and/or feeding grounds, e.g., grey mullet *Mugil cephalus* off Taiwan (Chang et al. 2004).

How habitats are spatially configured to each other is also important to fish usage and associated fisheries production. For example, Nagelkerken et al. (2001) showed that the presence of mangroves in tropical systems significantly increases species richness and abundance of fish assemblages in adjacent seagrass beds. Jelbart et al. (2007) sampled Australian temperate seagrass beds close to (within 200 m) and distant from (more than 500 m from) mangroves. They found seagrass beds closer to mangroves had greater fish densities and diversities than more distant beds, especially of juveniles. Conversely, the densities of fish species in seagrass at low tide that were also found in mangroves at high tide were negatively correlated with the distance of the seagrass bed from the mangroves. This shows the important daily habitat connectivity that exists through tidal movements between mangrove and seagrass habitats. Similar dynamics may occur in more subtidal coastal systems at larger spatial and temporal scales. For example, Dorenbosch et al. (2005) showed that adult densities of coral reef fish, whose juvenile phases were found in mangrove and seagrass nursery habitats, were much reduced or absent on coral reefs located far distant from such nursery habitats, relative to those in closer proximity.

A less studied, but increasingly recognised theme is the existence of intra-population variability in movement and other behavioural traits. Different behavioural phenotypes within a given population have been shown to be very common in land birds, insects, mammals, and other groups. An example of this is a phenomenon known as 'partial migration', where part of the overall population migrates each year, often over very large distances, while another component does not move and remains resident. By definition, this partial migration also results in differential use of habitats, often over large spatial scales. Recent work on white perch (*Morone americana*) in the United States shows that this population is made up of two behavioural components: a resident natal freshwater contingent, and a dispersive brackish-water contingent (Kerr et al. 2010). The divergence appears to be a response to early life history experiences that influence individuals' growth (Kerr 2008). The proportion of the overall population that becomes dispersive for a given year class ranges from 0% in drought years to 96% in high-flow years. Modelling of how differences in growth rates and recruitment strengths of each component contributed to the overall population found that the resident component contributed to long-term population persistence (stability), whereas the dispersive component contributed to population productivity and resilience (defined as rebuilding capacity) (Kerr et al. 2010). Another species, winter flounder *Pseudopleuronectes americanus*, has also shown intra-population variability in spawning migrations; one group stays coastally resident while a second smaller group migrates into estuaries to spawn (De Celles & Cadrin 2010). The authors went on to suggest that coastal waters in the Gulf of Maine should merit consideration in the assignment of Essential Fish Habitat for this species.

Kerr & Secor (2009) and Kerr et al. (2010) argue that such phenotypic dynamics are probably very common in marine fish populations but have not yet been effectively researched and quantified. The existence of such dynamics would have important implications for fisheries management, including the possibility of spatial depletions of more resident forms and variability in the use of potential HPSFM between years. For instance, recent work on snapper in the Hauraki Gulf has shown that fish on reef habitats are more resident (i.e., have less propensity to migrate) than those of soft sediment habitats, and can experience higher fishing removals (Parsons et al. 2011).

The most effective means of protecting a HPSFM in terms of the benefit to the fishery may differ depending on the life-history characteristics of the fish. A variety of modelling, theoretical, and observational approaches have led to the conclusion that spatial protection performs best at

enhancing species whose adults are relatively sedentary but whose larvae are broadcast widely (Chiappone & Sealey 2000, Murawski et al. 2000, Roberts 2000, Warner et al. 2000). The sedentary habit of adults allows the stock to accrue the maximum benefit from the protection, whereas the broadcasting of larvae helps 'seed' segments of the population outside the protection. However, the role of spatial protection in directly protecting juveniles after they have settled to seafloor habitats (via habitat protection/recovery, and/or reduced juvenile bycatch), or their interaction with non-fisheries impacts has not yet been explicitly considered.

14.3 STATE OF KNOWLEDGE IN NEW ZEALAND

14.3.1 POTENTIAL HPSFM IN NEW ZEALAND

Important areas for spawning, pupping, and egg-laying are potential HPSFM. These areas (insofar as these are known) have been identified and described using science literature and fisheries databases and summarised within two atlases, one coastal (less than 200 m) and one deepwater (more than 200 m). Coastally, these HPSFM areas were identified for 35 important fish species by Hurst et al. (2000b). This report concluded that virtually all coastal areas were important for these functions for one species or another. The report also noted that some coastal species use deeper areas for these functions, either as juveniles, or to spawn (e.g., red cod, giant stargazer) and some coastal areas are important for juveniles of deeper spawning species (e.g., hake and ling). Some species groupings were apparent from this analysis. Elephant fish, rig, and school shark all preferred to pup or lay eggs in shallow water, and very young juveniles of these species were found in shallow coastal areas. Juvenile barracouta, jack mackerel (*Trachurus novaezelandiae*), kahawai, rig, and snapper were all relatively abundant (at least occasionally) in the inner Hauraki Gulf. Important areas for spawning, pupping, and egg-laying were identified for 32 important deepwater fish species (200 to 1500 m depth), 4 pelagic fish species, 45 invertebrate groups, and 5 seaweeds (O'Driscoll et al. 2003). This study concluded that all areas to 1500 m deep were important for either spawning or for juveniles of one or more species studied. The relative significance of areas was hard to gauge because of the variability in the data, however the Chatham Rise was identified as a 'hotspot'.

Areas of high juvenile abundances of certain species may be useful indicators of HPSFM for some species. A third atlas (Hurst et al. 2000b) details species distributions (mainly commercial) of adult and immature stages from trawl, midwater trawl and tuna longline where adequate size information was collected. No conclusions are made in this document, and generalisations across species are inherently difficult, therefore like the previous two atlases, this document is probably best examined for potential HPSFM in a species specific way.

Certain locations within New Zealand already seem likely to qualify as HPSFM under any likely definition. The Kaipara Harbour has been identified as particularly important for the SNA 8 stock. Analysis of otolith chemistry showed that, for the 2003 year class, a very high proportion of new snapper recruits to the SNA 8 stock were sourced as juveniles from the Kaipara Harbour (Morrison et al. 2008). This result is likely to be broadly applicable into the future as the Kaipara provides most of the biogenic habitat available for juvenile snapper on this coast. The Kaipara and Raglan harbours also showed large catches of juvenile rig and the Waitemata, Tamaki and Porirua harbours moderate catches (Francis et al. 2012). Recent extensive fish habitat sampling within the Kaipara harbour in 2010 as part of the MBIE Coastal Conservation Management programme showed juvenile snapper to be strongly associated with subtidal seagrass, horse mussels, sponges, and an introduced bryozoan. Negative impacts on such habitats have the potential to have far-field effects in terms of subsequent fisheries yields from coastal locations well distant from the Kaipara Harbour. Beaches that still retain substantive toheroa populations, e.g., Dargaville and Oreti beaches, may also potentially qualify as HPSFM (Beentjes 2010).

Consistent with the international literature, biogenic (living, habitat forming) habitats have been found to be particularly important juvenile habitat for some coastal fish species in New Zealand. For example: bryozoan mounds in Tasman Bay are known nursery grounds for snapper, tarakihi and John dory (Vooren 1975); northern subtidal seagrass meadows fulfil the same role for a range of fish including snapper, trevally, parore, garfish and spotties (Francis et al. 2005, Morrison et al. 2008, Schwarz et al. 2006, Vooren 1975); northern horse mussel beds for snapper and trevally (Morrison et al. 2009); and mangrove forests for grey mullet, short-finned eels, and parore (Morrisey et al. 2010). Many other types of biogenic habitats exist, and some of their locations are known (e.g., see Davidson et al. 2010 for biogenic habitats in the Marlborough Sounds), but their

precise role as HPSFM remains to be quantified. Examples include open coast bryozoan fields, rhodoliths, polychaete (worm) species ranging in collective form from low swathes to large high mounds, sea pens and sea whips, sponges, hydroids, gorgonians, and many forms of algae, ranging from low benthic forms such as *Caulerpa* spp. (sea rimu) through to giant kelp (*Macrocystis pyrifera*) forests in cooler southern waters. Similarly, seamounts are well known to host reef-like formations of deep-sea stony corals (e.g., Tracey et al. 2011), as well as being major spawning or feeding areas for commercial deepwater species such as orange roughy and oreos (e.g., Clark 1999, O'Driscoll & Clark 2005). However, the role of these benthic communities on seamounts in supporting fishstocks is uncertain, as spawning aggregations continue to form even if the coral habitat is removed by trawling (Clark & Dunn 2012). Hence the oceanography or physical characteristics of the seamount and water column may be the key drivers of spawning or early life-history stage development, rather than the biogenic habitat.

Freshwater eels are reliant upon rivers as well as coastal and oceanic environments. GIS modelling estimates that for longfin eels, about 30% of longfin habitat in the North Island and 34% in the South Island is either in a reserve or in rarely/non-fished areas, with about 49% of the national longfin stock estimate of about 12 000 t being contained in these waterways (Graynoth et al. 2008). More regional examination of the situation for eels also exists, e.g., for the Waikato Catchment (Allen 2010). Shortfin eels prefer slower-flowing coastal habitats such as lagoons, estuaries, and lower reaches of rims (Beentjes et al. 2005). In-stream cover (such as logs and debris) has been identified as important habitat, particularly in terms of influencing the survival of large juvenile eels (Graynoth et al. 2008). Short-fin eel juveniles and adults have also been found to be relatively common in estuarine mangrove forests, and their abundance positively correlated with structural complexity (seedlings, saplings, and tree densities) (Morrisey et al. 2010). In addition oceanic spawning locations are clearly important for eels, the location of these are unknown, although it has been suggested that these may be north-east of Samoa and east of Tonga for shortfins and longfins respectively (Jellyman 1994).

Many of the potential HPSFM are threatened by either fisheries or land-based effects, the reader should look to the land-based effects chapter in this document and the eel section of the Stock Assessment Plenary report for further details.

14.3.2 HABITAT CLASSIFICATION AND PREDICTION OF BIOLOGICAL CHARACTERISTICS

Habitat classification schemes focused upon biodiversity protection have been developed in New Zealand at both national and regional scales, these may help identify larger habitats which HPSFM may be selected from, but are unlikely to be useful in isolation for determining HPSFM. The Marine Environment Classification (MEC), the demersal fish MEC and the benthic optimised MEC (BOMEC) are national-scale classification schemes that have been developed with the goal of aiding biodiversity protection (Leathwick et al. 2004, 2006, 2012). A classification scheme also exists for New Zealand's rivers and streams based on their biodiversity values to support the Department of Conservations Waters of National Importance (WONI) project (Leathwick & Julian 2008). Regional classification schemes also exist such as ones mapping the Marine habitats of Northland, or Canterbury in order to assist in Marine Protected Area planning (Benn 2009, Kerr 2010).

Another tool that may help in terms of identifying HPSFM is the predictions of richness, occurrence and abundance of small fish in New Zealand estuaries (Francis et al. 2011). This paper contains richness predictions for 380 estuaries and occurrence predictions for 16 species. This could help minimise the need to undertake expensive field surveys to inform resource management, although environmental sampling may still be needed to drive some models.

14.3.3 CURRENT RESEARCH

Prior to 2007 research within New Zealand was not explicitly focused on identifying HPSFM. However, in line with international trends, this situation has changed in recent times, with recognition of some of the wider aspects of fisheries management.

A number of Ministry and other research projects were commissioned concerning HPSFM in the 2010–11 year. Project ENV200907, 'Habitat of particular significance to fisheries management: Kaipara Harbour', is underway and has the overall objective of identifying and mapping areas and habitats of particular significance in the Kaipara Harbour which support coastal fisheries; and identifying

and assessing threats to these habitats. Included in this work is the reconstruction of environmental histories through interviews of long time local residents who have experience of the harbour, and associated collation and integration of historical data sources (e.g., catch records, photographs, diaries, maps, and fishing logs). Another output of this work will be recommendations on the best habitats and methods of monitoring to detect change to HPSFM within Kaipara Harbour.

Biogenic habitats on the continental shelf from about 5 to 150 m depths are currently being characterised and mapped through the biodiversity project ZBD2008/01, this will also provide new information on fisheries species utilisation of these habitats. Interviews with 50 retired fishers have provided valuable information on biogenic habitat around New Zealand. A national survey to examine the present occurrences and extents of these biogenic habitats was completed in 2011 in collaboration with Oceans Survey 2020, NIWA and Ministry of Business, Innovation and Employment (MBIE) funding.

A number of other national-scale projects are also underway. A desktop review is collating information on the importance of biogenic habitats to fisheries across the entire Territorial Sea and Exclusive Economic Zone (project HAB2007/01). A project has been approved to review the literature and recommend the relative urgency of research on habitats of particular significance for inshore finfish species (project ENV2010/03).

The Ministry of Business, Innovation and Employment (MBIE) funded project Coastal Conservation Management started in 2009 and runs for six years. This programme aims to integrate and add to existing fish-habitat association work to develop a national-scale marine fish-habitat classification and predictive model framework. This project will also attempt to develop threat assessments at local, regional and national scales. MPI is maximising the synergies between its planned research and this project. As part of this synergy, work on the connectivity and stock structure of grey mullet (*Mugil cephalus*) is underway in collaboration with MPI project GMU2009/01. Otolith chemistry is being assessed for its utility in partitioning the GMU 1 stock into more biologically meaningful management units, and in quantifying the suspected existence of source and sink dynamics between the various estuaries that hold juvenile grey mullet nursery habitats.

In 2012 MBIE also funded the three-year project delivered by NIWA entitled 'Predicting the occurrence of vulnerable marine ecosystems for planning spatial management in the South Pacific region'. The development of predictive models of species occurrence under this project may also aid in identifying HPSFM. Identification of biogenic habitat has been part of the MBIE project 'Vulnerable deep-sea communities' since 2009 (and its predecessor seamount programme), which includes surveys of a range of habitats that may be important for various life-history stages of commercial fish species: seamounts, canyons, continental slope, hydrothermal vents and seeps.

14.4 INDICATORS AND TRENDS

As no HPSFM are defined this section cannot be completed.

14.5 REFERENCES

Allen, D (2010) Eels in the Waikato Catchment. Client report prepared for Mighty River Power Ltd. 105 p.

Allen, M; Rosell, R; Evans, D (2006) Predicting catches for the Lough Neagh (Northern Ireland) eel fishery based on stock inputs, effort and environmental variables. *Fisheries Management and Ecology 13*: 251–260.

Atlantic States Marine Fisheries Commission (2000) Interstate Fishery Management Plan for American Eel. Fishery Management Report No. 36 of the Atlantic States Marine Fisheries Commission. 93 p.

Beentjes, M (2010) Toheroa survey of Oreti Beach, 2009, and review of historical surveys. *New Zealand Fisheries Assessment Report 2010/6*. 40 p.

Beentjes, M; Boubée, J; Jellyman, J D; Graynoth, E (2005) Non-fishing mortality of freshwater eels (*Anguilla* spp.). *New Zealand Fisheries Assessment Report 2005/34*. 38 p.

Benaka, L (Ed.) (1999) Fish habitat: essential fish habitat and rehabilitation, American Fisheries Society, Bethesda, MD. 45 p.

Benn, L (2009) Marine Protected Areas (MPA): Habitat Maps for Canterbury. Internal Report for the Canterbury Conservancy.

Booth, J (1979) North Cape - a 'nursery area' for the packhorse rock lobster, *Jasus verreauxi* (*Deeapoda: Palinuridae*). *New Zealand Journal of Marine & Freshwater Research 13*: 521–528.

Caddy, J F (2008) The importance of 'Cover' in the life histories of demersal and benthic marine resources: A neglected issue in fisheries assessment and management. *Bulletin of Marine Science 83*: 7–52.

Chang, C; Iizyuka, Y; Tzeng, W (2004) Migratory environmental history of the grey mullet *Mugil cephalus* as revealed by otolith Sr:Ca ratios. *Marine Ecology Progress Series 269*: 277–288.

Chiappone, M; Sealey, K M S (2000) Marine reserve design criteria and measures of success: Lessons learned from the Exuma Cays Land and Sea Park, Bahamas. *Bulletin of Marine Science 66*: 691–705.

Clark, M R (1999) Fisheries for orange roughy (*Hoplostethus atlanticus*) on seamounts in New Zealand. *Oceanologica Acta 22*: 593–602.

Clark, M R; Dunn, M R (2012) Spatial management of deep-sea seamount fisheries: balancing exploitation and habitat conservation. *Environmental Conservation 39 (2)*: 204–214. doi:10.1017/S0376892912000021.

Clark, M R; Rowden, A A (2009) Effect of deepwater trawling on the macro-invertebrate assemblages of seamounts on the Chatham Rise, New Zealand. *Deep Sea Research I 56*: 1540–1554.

Costello, M; McCrea, M; Freiwald, A; Lundälv, T; Jonsson, L; Bett, B; van Weering, T; de Haas, H; Roberts, J; Allen, D (2005) Role of cold-water *Lophelia pertusa* coral reefs as fish habitat in the NE Atlantic. *In*: Freiwald, A; JM, R (Eds.), Cold-water Corals and Ecosystems. pp 771–805. Springer-Verlag, Berlin.

Davidson, R; Richards, L; Duffy, C; Kerr, V; Freeman, D; D'Archino, R; Read, G; Abel, W (2010) Location and biological attributes of biogenic habitats located on soft substrata in the Marlborough Sounds. *Survey and Monitoring Report 575*, prepared by Davidson Environmental Ltd. for Department of Conservation and Marlborough District Council.

De Celles, G R; Cadrin, S X (2010) Movement patterns of winter flounder (*Pseudopleuronectes americanus*) in the southern Gulf of Maine: observations with the use of passive acoustic telemetry. *Fisheries Bulletin 108*: 408–419.

Diaz, R; Solan, M; Valente, R (2004) A review of approaches for classifying benthic habitats and evaluating habitat quality. *Journal of Environmental Management 73*: 165–181.

Deep Water Group (2008) Operational Procedures: Hoki Fishery. 27 p.

Dorenbosch, M; Grol, M; Christianen, M; Nagelkerken, I; van der Velde, G (2005) Indo-Pacific seagrass beds and mangroves contribute to fish density coral and diversity on adjacent reefs. *Marine Ecology Progress Series 302*: 63–76.

Ellingsen, K; Hewitt, J; Thrush, S (2007) Rare species, habitat diversity and functional redundancy in marine benthos. *Journal of Sea Research 58*: 291–301.

European Commission (2003) Commission Regulation (EC) No 1475/2003 of 20 August 2003 on the protection of deep-water coral reefs from the effects of trawling in an area north west of Scotland. *Official Journal L 211*: 14–15.

Fosså, J; Mortensen, P; Furevik, D (2002) The deep-water coral *Lophelia pertusa* in Norwegian waters: distribution and fisheries impacts. *Hydrobiologia 471*: 1–12.

Francis, M; Lyon, W; Jones, E; Notman, P; Parkinson, D; Getzlaff, C (2012) Rig nursery grounds in New Zealand: a review and survey. *New Zealand Aquatic Environment and Biodiversity Report No. 95*. 50 p.

Francis, M; Morrison, M; Leathwick, J; Walsh, C (2011) Predicting patterns of richness, occurrence and abundance of small fish in New Zealand estuaries. *Marine and Freshwater Research 62*: 1327–1341.

Francis, R I C C; Hadfield, M G; Bradford-Grieve, J M; Renwick, J A; Sutton, P J H (2005) Environmental predictors of hoki year-class strengths: an update. *New Zealand Fisheries Assessment Report 2005/58*. 22 p.

Gillanders, B; Kingsford, M (1986) Elements in otoliths may elucidate the contribution of estuarine recruitment to sustaining coastal reef populations of a temperate reef fish. *Marine Ecology Progress Series 141*: 13–20.

Grange, K; Tovey, A; Hill, A F (2003) The spatial extent and nature of the bryozoan communities at Separation Point, Tasman Bay. *Marine Biodiversity Biosecurity Report 4*. 22 p.

Graynoth, E; Francis, R; Jellyman, D (2008) Factors influencing juvenile eel (Anguilla spp.) survival in lowland New Zealand streams. *New Zealand Journal of Marine and Freshwater Research 42*: 153–172.

Hall-Spencer, J; Kelly, J; Maggs, C (2008) Assessment of maerl beds in the OSPAR area and the development of a monitoring program. Department of Environment, Heritage and Local Government: Ireland.

Hamer, P; Jenkins, G; Gillanders, B (2005) Chemical tags in otoliths indicate the importance of local and distant settlement areas to populations of a temperate sparid, *Pagrus auratus*. *Canadian Journal of Fisheries and Aquatic Sciences 62*: 623–630.

Helson, J; Leslie, S; Clement, G; Wells, R; Wood, R (2010) Private rights, public benefits: Industry-driven seabed protection. *Marine Policy 34*: 557–566.

Hewitt, J E; Thrush, S F; Halliday, J; Duffy, C (2005) The importance of small-scale habitat structure for maintaining beta diversity. *Ecology 86*: 1619–1626.

Hiscock, K; Tyler-Walters, H (2006) Assessing the seabed species and biotopes – the Marine Life Information Network (MarLIN). *Hydrobiologica 555*: 309–320.

Hurst, R; Bagley, N; Anderson, O; Francis, M; Griggs, L; Clark, M; Paul, L; Taylor, P (2000a) Atlas of juvenile and adult fish and squid distributions from bottom and midwater trawls and tuna longlines in New Zealand waters. *NIWA Technical Report No. 84*. 11 p.

Hurst, R; Stevenson, M; Bagley, N; Griggs, L; Morrison, M; Francis, M (2000b) Areas of importance for spawning, pupping or egg-laying and juveniles of New Zealand coastal fish. Final Research Report for Ministry of Fisheries research project ENV1999/03, Objective 1. 56 p.

Inland Fisheries Service Tasmania (2009) Tasmanian Freshwater Eel Fishery: Application to the Department of the Environment, Water, Heritage and Arts for the re-assessment of the Tasmanian Freshwater Eel Fishery. 22 p.

Jelbart, J; Ross, P; Connolly, R (2007) Fish assemblages in seagrass beds are influenced by the proximity of mangrove forests. *Marine Biology 150*: 993–1002.

Jellyman, D (1994) The fishery for freshwater eels (*Anguilla* spp.) in New Zealand. *New Zealand Fisheries Assessment Research Document 94/14*. 25 p. (Unpublished report held in NIWA library, Wellington.)

Kamenos, N; Moore, P; Hall-Spencer, J (2004) Small-scale distribution of juvenile gadoids in shallow inshore waters; what role does maerl play? *ICES Journal of Marine Science 61*:422–429.

Keefer, M; Peery, C; Wright, N; Daigle, W; Caudill, C; Clabough, T; Griffith, D; Zacharias, M (2008) Evaluating the NOAA Coastal and Marine Ecological Classification Standard in estuarine systems: A Columbia River Estuary case study. *Estuarine, Coastal and Shelf Science 78*: 89–106.

Kerr, L (2008) Cause, consequence, and prevalence of spatial structure of white perch (*Morone americana*) populations in the Chesapeake Bay. Dissertation. University of Maryland, College Park, Maryland, USA.

Kerr, L A; Cadrin, S X; Secor, D H (2010) The role of spatial dynamics in the stability, resilience, and productivity of an estuarine fish population. *Ecological Applications 20*: 497–507.

Kerr, L; Secor, D (2009) Bioenergetic trajectories underlying partial migration in Patuxent River (Chesapeake Bay) white perch *Morone americana*. *Canadian Journal of Fisheries and Aquatic Sciences 66*: 602–612.

Kerr, V (2010) Marine habitat map of Northland: mangawhai to Ahipara (vers 1). Department of Conservation.

Koenig, C; Coleman, F; Grimes, C; FitzHugh, G; Scanlon, K; Gledhill, C; Grace, M (2000) Protection of fish spawning habitat for the conservation of warm-temperate reef-fish fisheries of shelf-edge reefs of Florida. *Bulletin of Marine Science 66*: 593–616.

Leathwick, J R; Dey, K; Julian, K (2006) Development of a demersal fish community classification for New Zealand's Exclusive Economic Zone. *NIWA Client Report HAM2006/062*, prepared for the Department of Conservation. 35 p.

Leathwick, J; Image, K; Snelder, T; Weatherhead, M; Wild, M (2004) Definition and tests of the Marine Environment Classifications of New Zealand's Exclusive Economic Zone and the Hauraki Gulf. *NIWA Client Report CHC2004-085*. 64 p.

Leathwick, J R; Julian, K (2008) Updated conservation rankings for New Zealand's rivers and streams. NIWA report for Department of Conservation. 39 p.

Leathwick, J; Rowden, A; Nodder, S; Gorman, R; Bardsley, S; Pinkerton, M; Baird, S; Hadfield, M; Currie, K; Goh, A (2012) Benthic optimised marine environment classification for New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 89*. 52 p.

Livingston, M (1990) Spawning hoki (*Macruronus novaezelandiae* Hector) concentrations in Cook Strait and off the east coast of the South Island, New Zealand, August–September 1987. *New Zealand Journal of Marine and Freshwater Research 24*: 503–517.

Lundälv, T; Jonsson, L (2003) Mapping of deep-water corals and fishery impacts in the north-east Skagerrak, using acoustical and ROV survey techniques. Proceedings 6th Underwater Science Symposium, Aberdeen, April 2003.

Madden, C; Grossman, D; Goodin, K (2005) Coastal and marine ecosystems of North America; framework for an ecological classification standard: Version II. NatureServe, Virginia, Arlington. 48 p.

Morato, T; Hoyle, S D; Allain, V; Nicol, S J (2010a) Seamounts are hotspots of pelagic biodiversity in the open ocean. *Proceedings of the National Academy of Science USA 107*: 9707–9711. doi:10.1073/pnas.0910290107.

Morato, T; Hoyle, S D; Allain, V; Nicol, S J (2010b) Tuna longline fishing around West and Central Pacific seamounts. *PloS ONE 5 (12)*: e14453. doi:10.1371/journal.pone.0014453.

Morrisey, D; Swales, A; Dittmann, S; Morrison, M; Lovelock, C; Beard, C (2010) The ecology and management of temperate mangroves. *Oceanography and Marine Biology 48*: 43–160.

Morrison, M; Jones, E; Consalvey, M; Berkenbusch, K (2008) Biogenic habitats and their value to New Zealand fisheries. *Water & Atmosphere 16*: 20–21.

Morrison, M; Lowe, M; Parsons, D; Usmar, N; McLeod, I (2009) A review of land-based effects on coastal fisheries and supporting biodiversity in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 37*. 100 p.

Murawski, S A; Brown, R; Lai, H L; Rago, P J; Hendrickson, L (2000) Large-scale closed areas as a fishery-management tool in temperate marine systems: The Georges Bank experience. *Bulletin of Marine Science 66*: 775–798.

Nagelkerken, I; Kleijnen, S; Klop, T; van den Brand, R; de la Moriniere, E; van der Velde, G (2001) Dependence of Caribbean reef fishes on mangroves and seagrass beds as nursery habitats: a comparison of fish faunas between bays with and without mangroves/seagrass beds. *Marine Ecology Progress Series 214*: 225–235.

O'Driscoll, R; Booth, J; Bagley, N; Anderson, O; Griggs, L; Stevenson, M; Francis, M (2003) Areas of importance for spawning, pupping or egg-laying, and juveniles of New Zealand deepwater fish, pelagic fish, and invertebrates. *NIWA Technical Report 119*. 377 p.

O'Driscoll, R L; Clark, M R (2005) Quantifying the relative intensity of fishing on New Zealand seamounts. *New Zealand Journal of Marine and Freshwater Research 39*: 839–850.

Parsons, D; Morrison, M; McKenzie, J; Hartill, B; Bian, R; Francis, R I C C (2011) A fisheries perspective of behavioural variability: differences in movement behavior and extraction rate of an exploited sparid, snapper (*Pagrus auratus*). *Canadian Journal of Fisheries and Aquatic Sciences 68*: 632–642.

Roberts, C M (2000) Selecting marine reserve locations: Optimality versus opportunism. *Bulletin of Marine Science 66*: 581–592.

Rosenberg, A; Bigford, T; Leathery, S; Hill, R; Bickers, K (2000) Ecosystem approach to fishery management through essential fish habitat. *Bulletin of Marine Science 66*: 535–542.

Ross, S; Moser, M (1995) Life-history of juvenile gag, *Mycteroperca-microlepsis*, in North-Carolina estuaries. *Bulletin of Marine Science 56*: 222–237.

Schumacher, J D; Kendall, A W (1991) Some interactions between young walleye Pollock and their environment in the western gulf of Alaska. *California Cooperative Oceanic Fisheries Investigations Reports 32*: 22–40.

Schwarz, A-M; Morrison, M; Hawes, I; Halliday, J (2006) Physical and biological characteristics of a rare marine habitat: subtidal seagrass beds of offshore islands. *Science for Conservation 269*. 30 p.

Thrush, S F; Gray, J S; Hewitt, J E; Ugland, K I (2006) Predicting the effects of habitat homogenization on marine biodiversity. *Ecological Applications 16*: 1636–1642.

Tracey, D M; Rowden, A A; Mackay, K A; Compton, T (2011) Habitat-forming cold-water corals show affinity for seamounts in the New Zealand region. *Marine Ecology Progress Series 430*: 1–22.

Turner, G E; Boesch, D (1987) Aquatic animal production and wetland relationships: insights gleaned following wetland loss or gain. *In*: Hooks, D (Ed.), Ecology and management of wetlands. pp 25–39. Croon Helms, Ltd., Beckenham, Kent, UK.

Turner, S; Schwarz, A-M (2006) Management and conservation of seagrass in New Zealand: an introduction. *Science for Conservation 264*: 90 p.

Valavanis, V D; Pierce, G J; Zuur, A F; Palialexis, Saveliev, A; Katara, I; Wang, J (2008) Modelling of essential fish habitat based on remote sensing, spatial analysis and GIS. *Hydrobiologia 612*: 5–20.

Vooren, C M (1975) Nursery grounds of tarakihi (*Teleostei: Cheilodactylidae*) around New Zealand. *New Zealand Journal of Marine and Freshwater Research 9*: 121–158.

Warner, R R; Swearer, S E; Caselle, J E (2000) Larval accumulation and retention: Implications for the design of marine reserves and essential fish habitat. *Bulletin of Marine Science 66*: 821–830.

Werner, E E; Gilliam, J F (1984) The ontogenetic niche and species interaction in size-structured populations. *Annual Review of Ecology, Evolution, and Systematics 15*: 393–425.

Western Pacific Regional Fishery Management Council (1998) Magnuson-Stevens Act definitions and required provisions. Western Pacific Regional Fishery Management Council, Hawaii.

15 LAND-BASED EFFECTS ON FISHERIES, AQUACULTURE AND SUPPORTING BIODIVERSITY

Status of chapter	This chapter has not been updated for AEBAR 2018.
Scope of chapter	This chapter outlines the main known threats from land-based activities to fisheries, aquaculture and supporting biodiversity. It also describes the present status and trends in land-based impacts.
Area	All of the New Zealand freshwater, EEZ and Territorial Sea.
Focal localities	Freshwater habitats and areas closest to the coast are likely to be most impacted; this will be exacerbated in areas with low water movement. Anthropogenically increased sediment run-off is particularly high from the Waiapu and Waipaoa river catchments on the east coast of the North Island. Areas of intense urbanisation or agricultural use of catchments are also likely to be impacted by bacteria, viruses, heavy metals or nutrients, or some combination of these.
Key issues	Habitat modification, sedimentation, aquaculture, shellfish, terrestrial land-use change (particularly for urbanisation, forestry or agriculture) water quality and quantity, contamination, consequences to seafood production of increased pollutants, freshwater management and demand.
Emerging issues	Impacts on habitats of particular significance to fisheries management (HPSFM), linkages through rainfall patterns to climate change, shellfish bed closures, habitat remediation, domestic animal diseases in protected marine species, proposed aquaculture expansion, water abstraction impacts.
MPI research (current)	ZBD2008/01 *Research on Biogenic Habitat-Forming Biota and their functional role in maintaining Biodiversity in the Inshore Region (5–150 m depths)* – this is also part-funded by Oceans Survey 2020, NIWA and MBIE.
NZ government research (current)	Ministry of Business, Innovation and Employment (MBIE) funded programmes: UOCX0902 *After the outfall: recovery from eutrophication in degraded New Zealand estuaries*; CO1X1005 *Management of Cumulative Effects of Stressors in Aquatic Ecosystems*. NIWA core-funded research on this topic occurs in two areas. Firstly, the 'Managing marine ecosystems' programme, specifically the projects 'Measuring mapping and conserving', 'Ecosystem-based management of coasts and estuaries', 'Coastal management' (C01X0907) and 'Marine Futures' (C01X0227) (Note that the latter two finish 30 September 2014). Secondly, in the 'Fisheries' Centre, the EAFM programme deals with ecosystem-based management approaches in conjunction with the 'Coasts and Oceans' centre. Some funding within these areas will be aligned to the Sustainable Seas Science Challenge in the near future in which the focus is on ecosystem based management of the marine environment.
Related chapters/issues	Habitats of particular significance for fisheries management (HPSFM), marine environmental monitoring.

15.1 CONTEXT

Land-based activities that may have impacts on seafood production are primarily regulated under the Resource Management Act 1991 (and subsequent amendments). Fisheries are controlled under the Fisheries Act 1996, this includes marine and freshwater responsibilities regarding aquatic life (under Part 2 of the Fisheries Act). The MPI

Strategy 'Our Strategy'[1] states that New Zealand's natural resources need to be sustainable, in the primary sector.

The government's 'Fresh Start for Freshwater Programme'[2] (led by MfE and MPI) aims to create a water management system that allows us to make more transparent and better targeted and informed decisions on fresh water. Businesses and water users will have more certainty so that they can plan and invest. All New Zealanders will have a greater say on the water quality they want for their lakes and rivers. The Coastal Policy Statement (2010) also has relevance to matters of fisheries interest, e.g., Policy 20(1) (paraphrased) controls the use of vehicles on beaches where (b) harm to shellfish beds may result. MPI also works with other agencies, principally DOC, MfE and regional councils and through the Natural Resource Cluster to influence these processes to ensure consideration of land-based impacts upon seafood production. The New Zealand aquaculture industry has an objective of developing into a billion dollar industry by 2025.[3] Government supports well-planned and sustainable aquaculture through its *Aquaculture Strategy and Five-year Plan*. One of the desired outcomes of actions by the New Zealand government is to enable more space to be made available for aquaculture. This outcome is likely to heighten the potential for conflict between aquaculture proponents and those creating negative land-based effects.

An MPI-funded survey of scientific experts (MacDiarmid et al. 2012) addressed the vulnerability to a number of threats of marine habitat types within the New Zealand's Territorial Sea and Exclusive Economic Zone (EEZ). Each vulnerability score was based on an assessment of five factors including the spatial scale, frequency and functional impact of the threat in the given habitat as well as the susceptibility of the habitat to the threat and the recovery time of the habitat following disturbance from that threat. The study found that the number of threats and their severity were generally considered to decrease with depth, particularly below 50 m. Reef, sand, and mud habitats in harbours and estuaries and along sheltered and exposed coasts were considered to be the most highly threatened habitats. The study also reported that over half of the 26 top threats fully, or in part, stemmed from human activities external to the marine environment itself. The top six threats in order were:

1. ocean acidification,
2. rising sea temperatures resulting from global climate change,
3rd equal. bottom trawling fishing,
3rd equal. increased sediment loadings from river inputs,
5th equal. change in currents from climate change,
5th equal. increased storminess from climate change.

The reader is guided to MacDiarmid et al. (2012) for more detail including tables of threats-by-habitat and habitats-by-threat. Climate change and ocean acidification, although they can be considered land-based effects, are covered under the chapters in this document called 'New Zealand's Climate and Oceanic Setting' and 'Biodiversity'.

Land-based effects on seafood production and biodiversity in this context are defined as resulting either from the inputs of contaminants from terrestrial sources or through engineering structures (e.g., breakwaters, causeways, bridges), that change the nature and characteristics of coastal habitats and modify hydrodynamics. The major route for entry of land-based contaminants into the marine environment is associated with freshwater flows (rivers, streams, direct runoff and ground water), although contaminants may enter the marine environment via direct inputs (e.g., landslides) or atmospheric transport processes.

The most important land-based effect in New Zealand is arguably increased sediment deposition around our coasts (Morrison et al. 2009, MacDiarmid et al. 2012). This deposition has been accelerated due to increased erosion from land-use, which causes gully and channel erosion and landslides (Glade 2003). Inputs of sediments to our coastal zone, although naturally high in places due to our high

[1] Ministry for Primary Industries. Our Strategy. Retrieved from http://www.mpi.govt.nz/about-mpi/our-strategy.

[2] Ministry for the Environment. Fresh Start for Fresh Water. Retrieved from http://www.mfe.govt.nz/more/cabinet-papers-and-related-material-search/cabinet-papers/freshwater/fresh-start-fresh-water.

[3] Aquaculture New Zealand. Strategy. Retrieved from http://aquaculture.org.nz/about-us/strategy.

rainfall and rates of tectonic uplift (Carter 1975), have been accelerated by human activities (Goff 1997). Sediment inputs are now high by world standards and make up about 1% of the estimated global detrital input to the oceans (Carter et al. 1996). By contrast New Zealand represents only about 0.3% of the land area that drains into the oceans (Griffiths & Glasby 1985, Milliman & Syvitski 1992).

Different land-use effects act over different scales; for example localised effects act on small streams and adjacent estuarine habitats, large scale effects extend to coastal embayments and shelf ecosystems. Associated risks will vary according to location and depend on the relevant ecosystem services (e.g., high value commercial fishery stocks) and their perceived sensitivities. The risk from stormwater pollutants will be more important near urban areas and the effects of nutrient enrichment will be more important near intensively farmed rural areas.

The risk from land-based impacts for seafood production is that they will limit the productivity of a stock or stocks. For example, the bryozoan beds around Separation Point in Golden Bay, were protected from fishing in 1980, partly because of their perceived role as nursery grounds for a variety of coastal fish species (Grange et al. 2003). Recent work has suggested that the main threat to these bryozoans is now sedimentation from the Motueka River, which may inhibit recovery of any damaged bryozoans (Grange et al. 2003, Morrison et al. 2009). Any declines in this bryozoan bed and associated ecological communities could also affect the productivity of adjacent fishery stocks.

MPI mainly manages in the marine environment, therefore this topic area will be dealt with first. The main freshwater fisheries management MPI is involved in is the freshwater eel fishery; this will be dealt in later sections, as relevant.

15.2 GLOBAL UNDERSTANDING

15.2.1 LAND-BASED INFLUENCES

It has been acknowledged for some time now that land-based activities can have important effects on seafood production. The main threats to the quality and use of the world's oceans are (GESAMP 2001):

- alteration and destruction of habitats and ecosystems;
- effects of sewage on human health;
- widespread and increased eutrophication;
- decline of fishstocks and other renewable resources; and
- changes in sediment flows due to hydrological changes.

Coastal development is projected to impact 91% of all inhabited coasts by 2050 and will contribute to more than 80% of all marine pollution (Nellemann et al. 2008). The importance of different land-based influences differ regionally but the South Pacific Regional Environmental Programme (SPREP, which includes New Zealand) defines waste management and pollution control as one of its four strategic priorities for 2011–15 (SPREP 2010).

Influences, including land-based influences, seldom work in isolation; for example the development of farming and fishing over the last hundred years has meant that increased sediment and nutrient runoff has to some degree occurred simultaneously with increased fishing pressure. However, the impact of these influences has often been studied in isolation. In a review on coastal eutrophication, Cloern (2001) stated that *'Our view of the problem [eutrophication] is narrow because it continues to focus on one signal of change in the coastal zone, as though nutrient enrichment operates as an independent stressor; it does not reflect a broad ecosystem-scale view that considers nutrient enrichment in the context of all the other stressors that cause change in coastal ecosystems'*. These influences (in isolation or combination) can also cause indirect effects, such as decreasing species diversity that then lessens resistance to invasion by non-indigenous species or species with different life-history strategies (Balata et al. 2007, Kneitel & Perrault 2006, Piola & Johnston 2008). Studies that research a realistic mix of influences are rare, but valuable.

Sediment deposition can be an important influence, particularly in areas of high rainfall, tectonic uplift, and forest clearances, or areas where these activities coincide. Sediments are known to erode from the land at an increased rate in response to human use, for example, estimates from a largely deforested tropical highland suggest erosion rates 10–100 times faster than pre-clearance rates (Hewawasam et al. 2003). Increased sediment either deposited on the seafloor or suspended in the water column can negatively impact invertebrates in a number of ways including: burial, scour, inhibiting settlement, decreasing filter-feeding efficiency and decreasing light penetration, generally leading to less

diverse communities, with a decrease in suspension feeders (Thrush et al. 2004). These impacts can affect the structure, composition and dynamics of benthic communities (Airoldi 2003, Thrush et al. 2004). Effects of this increased sediment movement and deposition on finfish are mostly known from freshwater fish and can range from behavioural (such as decreased feeding rates) to sublethal (e.g., gill tissue disruption) and lethal as well as having effects on habitat important to fishes (Morrison et al. 2009). These effects differ by species and life-stage and are dependant upon factors that include the duration, frequency and magnitude of exposure, temperature, and other environmental variables (Servizi & Martens 1992).

Increased nutrient addition to the aquatic environment can initially increase production, but with increasing nutrients there is an increasing likelihood of harmful algal blooms and cascades of effects damaging to most communities above the level of the plankton (Kennish 2002, Heisler et al. 2008). This excess of nutrients is termed eutrophication. Eutrophication can stimulate phytoplankton growth, which can decrease the light availability and subsequently lead to losses in benthic production from seagrass, microalgae or macroalgae and their associated animal communities. Algal blooms then die and their decay depletes oxygen and blankets the seafloor. The lack of oxygen in the bed and water column can lead to losses of finfish and benthic communities. These effects are likely to be location specific and are influenced by a number of factors including: water transparency, distribution of vascular plants and biomass of macroalgae, sediment biogeochemistry and nutrient cycling, nutrient ratios and their regulation of phytoplankton community composition, frequency of toxic/harmful algal blooms, habitat quality for metazoans, reproduction/growth/survival of pelagic and benthic invertebrates, and subtle changes such as shifts in the seasonality of ecosystems (Cloern 2001). The effects of eutrophication abound in the literature, for example, the formation of dead (or anoxic) zones is exacerbated by eutrophication, although oceanographic conditions also play a key role (Diaz & Rosenberg 2008). Dead zones have now been reported from more than 400 systems, affecting a total area of more than 245 000 km^2 (Diaz & Rosenberg 2008). This includes anoxic events from New Zealand in coastal north-eastern New Zealand and Stewart Island (Taylor et al. 1985, Morrissey 2000).

Other pollutants such as heavy metals and organic chemicals can have severe effects, but are more localised in extent than sediment or nutrient pollution (Castro and Huber 2003, Kennish 2002). Fortunately the concentration of these pollutants in most New Zealand aquatic environments is relatively low, with a few known exceptions. Examples of this include naturally elevated levels of arsenic in Northland,[4] cadmium levels in Foveaux Strait oysters (Frew et al. 1996) and levels of nickel and chromium within the Motueka river plume in Tasman Bay (Forrest et al. 2007). The high cadmium levels have caused market access issues for Foveaux Strait oysters. Some anthropogenically generated pollutants such as copper, lead, zinc and PCBs are high in localised hotspots within urban watersheds. In the Auckland region these hotspots tend to be in muddy estuarine sites and tidal creeks that receive runoff from older urban catchments (Auckland Regional Council 2010). There is a lack of knowledge on the impacts of these pollutants upon fisheries.

Climate change is likely to interact with the effect of land-based impacts as the main delivery of land-based influences is through rainfall and subsequent freshwater flows. Global climate change projections include changes in the amount and regional distribution of rainfall over New Zealand (IPCC 2007). More regional predictions include increasing frequency of heavy rainfall events over New Zealand (Whetton et al. 1996). This is likely to exacerbate the impact of some land-based influences as delivery peaks at times of high rainfall, e.g., sediment delivery (Morrison et al. 2009).

Physical alterations of the coast are generally, but not exclusively (e.g., wetland reclamation for agriculture), concentrated around urban areas and can have a number of consequences on the marine environment (Bulleri & Chapman 2010). Changes in diversity, habitat fragmentation or loss and increased invasion susceptibility have all been identified as consequences of physical

[4] NIWA. Ocean Survey 2020. Retrieved from https://www.niwa.co.nz/coasts-and-oceans/research-projects/oceans-2020.

alteration. The effects of physical alterations upon fisheries remain largely unquantified; however the habitat loss or alteration portion of physical alterations will be dealt with under the habitats of particular significance for fisheries management (HPSFM) section.

An area of emerging interest internationally is infectious diseases from land-based animals affecting marine populations. Perhaps the most well-known example of this is the canine distemper outbreak in Caspian seals that caused a mass mortality in the Caspian Sea in 2000 (Kennedy et al. 2000).

15.2.2 HABITAT RESTORATION

Habitat restoration or rehabilitation has been the subject of much recent research. Habitat restoration or rehabilitation rarely, if ever, replaces what was lost and is most applicable in estuarine or enclosed coastal areas as opposed to exposed coastal or open ocean habitats (Elliott et al. 2007). Connectivity of populations is a key consideration when evaluating the effectiveness of any marine restoration or rehabilitation (Lipcius et al. 2008). In the marine area, seagrass replanting methodologies are being developed to ensure the best survival success (Bell et al. 2008) and artificial reefs can improve fisheries catches, although whether artificial reefs boost population numbers or merely attract fish is unclear (Seaman 2007). In addition, the incorporation of habitat elements in engineering structures, e.g., artificial rockpools in seawalls, shows promise in terms of ameliorating the impacts of physical alterations (Bulleri 2006). Spatial approaches to managing land-use impacts, such as marine reserves, will be covered under the section about HPSFM.

Freshwater rehabilitation has been reviewed by Roni et al. (2008). Habitat reconnection, floodplain rehabilitation and instream habitat improvement are all suggested for improving habitat and local fish abundances. Riparian rehabilitation, sediment reduction, dam removal, and restoration of natural flood regimes have shown promise for restoring natural processes that create and maintain habitats, but there is a lack of long-term studies to gauge their success. Wild eel fisheries in America and Europe have declined over time (Allen et al. 2006, Atlantic States Marine Fisheries Commission 2000, Haro et al. 2000). Declines in wild eel fisheries have been linked to a number of factors including: barriers to migration; hydro turbine mortality; and habitat loss or alteration. Information to quantitatively assess these linkages is however often lacking (Haro et al. 2000).

15.3 STATE OF KNOWLEDGE IN NEW ZEALAND

Land-based effects will be most pronounced closest to the land, therefore freshwater, estuarine, coastal, middle depths and deepwater fisheries, will be affected in decreasing order. The scale of land-use effects will, however, differ depending upon the particular influence. The most localised are likely to be direct physical impacts; for example, the replacement of natural shorelines with seawalls; although even direct physical impacts can have larger-scale impacts, such as affecting sediment transport and hence beach erosion, or contributing to cumulative effects upon ecosystem responses. Point-source discharges are likely to have a variable scale of influence, and this influence is likely to increase where a number of point-sources discharge, particularly when this occurs into an embayed, low-current environment. An example of this is Waitemata Harbour in Auckland where there are multiple stormwater discharges (Hayward et al. 2006). The influences on the largest scale can be from diffuse-source discharges such as nutrients or sediment (Kennish 2002). For example, the influence of diffuse-source materials from the Motueka river catchment in Golden Bay on subtidal sediments and assemblages and shellfish quality can extend up to tens of kilometres offshore (Tuckey et al. 2006; Forrest et al. 2007), with even a moderate storm event extending a plume greater than 6 km offshore (Cornelisen et al. 2011). Terrestrial influences on New Zealand's marine environment can, at times, be detected by satellites from differences in ocean colour and turbidity extending many kilometres offshore from river mouths (Gibbs et al. 2006).

All coastal areas are unlikely to suffer from land-based impacts in the same way. The quantities of pollutants or structures differ spatially. Stormwater pollutants, seawalls and jetties are more likely to be concentrated around urban areas. Nutrient inputs are likely to be concentrated either around sewage outlets or associated with areas of intensive agriculture or horticulture. Sediment production has been mapped around the country and is greatest around the west coast of the South Island and the east coast of the North Island (Griffiths & Glasby 1985, Hicks & Shankar 2003, Hicks et al. 2011). Notably the catchments where improved land management may result in the biggest changes to sediment delivery to coastal environments are likely to be the Waiapu and Waipaoa river catchments on the East

coast of the North Island. In addition to this, the sensitivity of receiving environments is also likely to differ; this will be covered in subsequent sections.

An MPI-funded project (IPA2007/07) reviewed the impacts of land-based influences on coastal biodiversity and fisheries (Morrison et al. 2009). This review used a number of lines of evidence to conclude that in this context, sedimentation is probably New Zealand's most important pollutant. The negative impacts of sediment include decreasing efficiency of filter-feeding shellfish (such as cockles, pipi, and scallops), reduced settlement success and survival of larval and juvenile phases (e.g., paua, kina), and reductions in the foraging abilities of finfish (e.g., juvenile snapper). Indirect effects include the modification or loss of important nursery habitats, particularly biogenic habitats (green-lipped and horse mussel beds, seagrass meadows, bryozoan and tubeworm mounds, sponge gardens, kelps/seaweeds, and a range of other structurally complex species). Inshore filter-feeding bivalves and biogenic habitats were identified as the most likely to be adversely affected by sedimentation. Eutrophication was also identified as a potential threat from experience overseas. This review identified knowledge gaps and made suggestions for more relevant research on these influences:

- identification of fisheries species/habitat associations for different life stages, including consideration of how changing habitat landscapes may change fisheries production;
- better knowledge of connectivity between habitats and ecosystems at large spatial scales;
- the role of river plumes;
- the effects of land-based influences both directly on fished species, and indirectly through impacts on nursery habitats;
- a better spatially based understanding, mapping and synthesis of the integrated impacts of land-based and marine-based influences on coastal marine ecosystems.

The locations where addressing land-based impacts is likely to result in a lowering in risk to seafood production or increased seafood production, excluding those already mentioned, are undefined.

A national-scale threat analysis has been completed for biogenic habitats, given their likely importance for fisheries management as nursery areas (Morrison et al. 2014b). The sparse data available (often anecdotal accounts), shows that strong declines in biogenic habitats have occurred, which appear largely attributable to land-based effects (e.g., sedimentation and elevated nutrient levels), and fishing impacts. Examples include the extensive loss of seagrass meadows (e.g., large areas in Whangarei, Waitemata, Manukau, Tauranga and Avon-Heathcote estuaries), green-lipped mussel beds (about 500 km^2 in the Hauraki Gulf), bryozoan beds (about 80 km^2 in Torrent Bay, about 800 km^2 in Foveaux Strait), and deepwater coral thickets on seamounts. Cumulatively, the magnitude and extent of biogenic habitat losses are likely to have been very substantial, but are unknown, and probably will never be able to be calculated. Other biogenic habitat species for which evidence points to historical losses include horse mussels, kelp forests, oyster beds, and sponges, both in assemblages where they tend to dominate, and as part of mixed biogenic habitat assemblages. A better understanding of the threats to these biogenic habitats is recommended.

The Kaipara Harbour has been identified as a system that supports important fisheries functions both for the harbour proper, and for the wider west coast North Island ecosystem (Morrison et al. 2014a). This report detailed fish-habitat associations in the harbour and concluded that increased sedimentation, and to a lesser extent the possibility of eutrophication, was probably the greatest threat to these fisheries.

The threat of sedimentation has prompted much concern and action by land managers and local communities (Morrison et al. 2014a). For example, in the Kaipara Harbour the southern subtidal seagrass meadows area is especially important as a juvenile nursery for snapper and trevally and based on its high value as a juvenile fish nursery habitat, the Auckland Council has listed this area as an Ecologically Significant Area (ESA) in its draft unitary plan. There are significant collaborative CRI/Northland Regional Council/Auckland Council sediment erosion and transport research programmes currently under way in Kaipara Harbour catchment and the harbour itself. There are also local initiatives around tree planting and the improvement of riparian and other forms of land management. The fish/fisheries habitat work described here engages and collaborates with the IKHMG and Kaipara Research Advisory Group (KRAG), and this type of collaboration/interaction between fisheries habitat research, other scientific research programmes, and

management agencies is one promising way for these issues to be addressed.

Another study investigated correlations between environmental variables and flounder abundance for the Manukau and Mahurangi harbours (McKenzie et al. 2013). Consistent correlations were obtained for a variety of environmental variables for juvenile sand and yellowbelly flounder (YBF) in the Manukau, but not in Mahurangi Harbour. The influence of environmental variables on adult YBF catch in the Manukau Harbour was even more evident. These correlations suggested that decreasing oxygen and increasing ammonia and turbidity may have negatively affected yellowbelly flounder recruitment success. When these results were considered alongside the declining trends in flatfish abundance in the FLA 1 fishery, estuarine water quality may be a significant factor affecting the sustainability of the flatfish fishery.

Marine restoration studies published in New Zealand have focused on the New Zealand cockle *Austrovenus stutchburyi*. The first of these studies identified a tagging methodology to aid relocation of transplanted individuals (Stewart & Creese 2002). Subsequent studies stressed the use of adults in restoration and the importance of site selection, either from theoretical or modelling viewpoints (Lundquist et al. 2009, Marsden & Adkins 2009). Detailed restoration methodology has been investigated in Whangarei Harbour and recommends replanting adults at densities between 222 and 832 m-2 (Cummings et al. 2007).

Multiple influences in areas relevant to seafood production in New Zealand have been addressed by three studies. A field experiment near Auckland showed greater effects on infaunal colonisation of intertidal estuarine sediments when three heavy metals (copper, lead and zinc) were in combination compared to each in isolation (Fukunaga et al. 2010). A survey approach looking at the interaction of sediment grain size, organic content and heavy metal contamination upon densities of 46 macrofaunal taxa across the Auckland region also showed a predominance of multiplicative effects (Thrush et al. 2008). However influences can work in unexpected directions; as in a study on large suspension feeding bivalves off estuary mouths where the anticipated negative impacts from sediment were not observed and these species benefitted from food resources generated from the estuaries (Savage et al. 2012).

Toheroa populations are currently closed to all but customary harvesting but have failed to recover to former population levels even though periodic (and sometimes substantial) pulses in young recruits have been detected in both Northland and Southland (Beentjes 2010, Morrison & Parkinson 2008). Current thinking suggests that a mix of influences are probably responsible for these declines including overharvesting, land-use changes leading to changes in freshwater seeps on the beaches, and vehicle traffic (Morrison et al. 2009, Williams et al. 2013). A number of discrete pieces of research have been completed in this area. A review of the wider impact of vehicles on beaches and sandy dunes has been completed, and suggested that more research was needed on the impacts of vehicle traffic on the intertidal (Stephenson 1999). A four-day study over a fishing contest on Ninety Mile Beach showed the potential of traffic to produce immediate mortalities of juvenile toheroa, but the temporal importance of this could not be gauged (Hooker & Redfearn 1998). Mortalities of toheroa from the Burt Munro Classic motorcycle race on Oreti beach have been quantified and recommendations made for how to minimise these, but again the importance of vehicle traffic for toheroa survival over longer time periods was unclear (Moller et al. 2009). Notably, similar negative impacts from driving were observed on juvenile tuatua (*Paphies donacina*) on a Pegasus Bay beach (Marsden & Taylor 2010). The impact of a range of influences upon toheroa at Ninety Mile Beach has been investigated by Williams et al. (2013). The main factors identified that potentially affect toheroa abundance were food availability, climate and weather, sand smothering/sediment instability, toxic algal blooms, predation, harvesting, vehicle impacts, and land-use change. To investigate the causal mechanisms operating, a combination of monitoring, experimental, and modelling studies may be necessary.

Rhodolith beds have been surveyed in the Bay of Islands and high diversity was reported even in areas of abundant fine sediments (Nelson et al. 2012). It is unclear if the increasing sedimentation occurring in the Te Rawhiti Reach is negatively impacting rhodoliths and whether this atypical rhodolith bed (i.e., with abundant fine sediments) is at risk if current sedimentation and mobilisation rates continue.

The protozoan *Toxoplasma gondii* has been identified as the cause of death for 7 of 28 Hector's and Māui dolphins examined since 2007 (W. Roe, Massey University, unpubl. data, 31 July 2012). Land-based runoff containing cat faeces

is believed to be the means by which *Toxoplasma gondii* enters the marine environment (Hill & Dubey 2002). A Hector's dolphin has also tested positive for *Brucella abortus* (or a similar organism) a pathogen of terrestrial mammals that can cause late pregnancy abortion, and has been seen in a range of cetacean species elsewhere. This resulted in the Department of Conservation's suggested research priorities in the 'Review of the Maui's dolphin Threat Management Plan: Consultation paper', including objectives to determine the presence, pathways and possible mitigation of the threat from *Toxoplasmosis gondii* (Department of Conservation and Ministry for Primary Industries 2012). The recently established Māui dolphin Research Advisory Group[5] confirmed risk factors to Māui dolphin from *Toxoplasma gondii* as a priority area for future research.

The effects of large-scale habitat loss and modification on eels in New Zealand are clearly significant, but difficult to quantify (Beentjes et al. 2005). Significant non-fisheries mortality of New Zealand freshwater longfin and shortfin eels are caused by mechanical clearance of drainage channels, and damage by hydro-electric turbines and flood control pumping. Eels prefer habitat that offers cover and in modified drains aquatic weed provides both daytime cover and nighttime foraging areas. Loss of weed and natural debris can thus result in significant displacement of eels to other areas. In addition, wetlands drainage has resulted in greatly reduced available habitat for eels, particularly shortfins, which prefer slower-flowing coastal habitats such as lagoons, estuaries, and lower reaches of rims. Water abstraction is one of a number of information requirements identified in Beentjes et al. (2005) to better define the effects on eel populations.

A number of Integrated Catchment Management (ICM) projects are underway in New Zealand. These take a holistic view to land management incorporating aquatic effects; this approach could help restore water quality of both fresh and coastal waters. An overview of these projects is given in a Ministry for the Environment Report on integrated catchment management (Environmental Communications Limited 2010). Many of these projects employ restoration techniques such as riparian planting, but few assessments of the effectiveness of riparian planting exist. One assessment of the effect of nine riparian zone planting schemes in the North Island on water quality, physical and ecological indicators concluded that riparian planting could improve stream quality; in particular, rapid improvements were seen in terms of visual clarity and channel stability (Parkyn et al. 2003). Nutrient and faecal contamination results were more variable. Improvement in macroinvertebrate communities did not occur in most streams and the three factors needed for these were canopy closure (which decreased stream temperature), long lengths of riparian planting and protection of headwater tributaries. A modelling study also demonstrated the long time lag needed to grow large trees, which then provide wood debris to structure channels, which achieves the best stream rehabilitation results (Davies-Colley et al. 2009). Although some of these studies extend into the marine realm (at least in terms of monitoring) it is difficult to gauge the impact of these activities upon fisheries or aquaculture, particularly on wider scales because ICM studies have been localised at small scales.

15.3.1 CURRENT RESEARCH

An MPI biodiversity project also has components that address land-based effects; the threats to biogenic habitats are addressed in project ZBD2008/01 (for more detail see the Biodiversity chapter).

A Ministry of Business, Innovation and Employment (MBIE)-funded project [6] of particular relevance is 'Nitrogen reduction and benthic recovery' (UOCX0902, University of Canterbury). This research aims to determine the trajectories and thresholds of coastal ecosystem recovery following removal of excessive nutrient loading (called 'eutrophication') and earthquake impacts. This will be achieved by monitoring the effects of diverting all of Christchurch's treated wastewater discharge from the eutrophied Avon-Heathcote (Ihutai) Estuary and the

[5] Department of Conservation. Māui Dolphin. Retrieved from http://www.doc.govt.nz/nature/native-animals/marine-mammals/dolphins/maui-dolphin.

[6] Ministry of Business, Innovation and Employment. Who got funded? Retrieved from http://www.msi.govt.nz/update-me/who-got-funded.

subsequent earthquake induced disturbances to this diversion.

15.4 INDICATORS AND TRENDS

A national view of the impacts of land-based influences upon seafood production does not exist; this could be facilitated by better coordination and planning of the many disparate marine monitoring programmes operating around the country. Monitoring of marine water quality and associated communities is carried out through a variety of organisations, including universities, regional councils and aquaculture or shell fisheries operations. Regional council monitoring of water quality and associated biological communities is often reported through websites such as the Auckland Regional Council environmental monitoring data, or summary reports such as the Hauraki Gulf state of the Environment 2011 report (Auckland Regional Council 2011). Water quality and associated marine communities may also be monitored for a regional council as part of a consent application or as a stipulation for a particular marine development. However the data from aquaculture and shellfisheries water quality monitoring are not generally available.

Improved coordination and planning of marine monitoring has been achieved in some countries, e.g., the United Kingdom. [7] The Marine Environmental Monitoring Programme (ZBD2010-42), is a step towards this goal, more information is available on this project in the Biodiversity chapter of this document. This project identifies remote sensing of sea surface particulate matter in nearshore waters as a possible indicator of changes in sediment inputs in the future, but this requires algorithm validation for New Zealand waters. Possible national-scale proxies for coastal faecal contamination may exist after collating information from sanitation area monitoring for shellfish harvesting and/or coastal bathing beaches.

High faecal coliform counts (primarily from mammal or bird faeces) can impact upon the value gained from shellfish fisheries and aquaculture. Area closures to commercial harvesting usually depend on an area's rainfall/runoff relationship and areas closer to significant farming areas or urban concentrations are likely to be closed more frequently, due to high faecal coliform counts, than areas where the catchment is unfarmed or not heavily populated. For example, Inner Pelorus sound is likely to be closed more frequently than outer Pelorus Sound (Marlborough Sounds). For coastal areas of the Marlborough Sounds, the Coromandel Peninsula and Northland closures can range from a few days to over 50% of the time in a given year (Brian Roughan, New Zealand Food Safety Authority, pers. comm.). Certain fisheries may be limited by the amount of time where water quality is sufficient to allow harvesting, e.g., the cockle fishery in COC 1A (Snake bank in Whangarei harbour) was closed for 101, 96, 167, 86, 117 and 118 days for the 2006–07, 2007–08, 2008–09, 2009–10, 2010–11 and 2011–12 fishing years, respectively, due to high faecal coliform counts from sewage spills or runoff.[8] Models also now exist that allow real-time prediction of *E. coli* pulses associated with storm events (e.g., Wilkinson et al. 2011), which may help harvesters to better cope with water quality issues.

The Ministry for the Environment (MfE) also reports on freshwater quality. River water quality indicators that have been assessed have direct relevance to the eel, and other freshwater fisheries, and this water will flow through estuaries and enter the marine environment. The National River Water Quality Network (NRWQN) has national coverage, and has been running for over 20 years and has recently reported upon the following eight variables: temperature, dissolved oxygen, visual clarity, dissolved reactive and total phosphorous, and ammoniacal, oxidised and total nitrogen (Ballantine & Davies-Colley 2009). Dissolved oxygen showed few meaningful trends and the ammoniacal nitrogen data suffered from a processing artefact. An upward, although not significant trend in temperature and an improvement of water clarity were seen at the national scale. However, a negative correlation was seen between water clarity and percent of catchment in pasture, which suggests that any expansion of pasture lands may have impacts on clarity. Strong increasing trends over time were seen in oxidised nitrogen, total nitrogen,

[7] CEFAS. Marine monitoring. Retrieved from http://wavenet.cefas.co.uk/Smartbuoy.

[8] Statistics supplied by New Zealand Food Safety Authority in Whangarei. Notably the fishery has not been operating since November 2012.

total phosphorous and dissolved reactive phosphorous. These latter trends all signify deteriorating water quality and are mainly attributable to increased diffuse-source pollution from the expansion and intensification of pastoral agriculture.

Total nitrogen and phosphorous loads to the coast in New Zealand have been modelled and were estimated at 167 300 and 63 100 t yr-1, respectively (Elliot et al. 2005).[9] The main sources of nitrogen and phosphorous were from pastoralism (70%) and erosion (53%), respectively. Dairying contributes 37% of the nitrogen load from only 6.8% of the land. The total amount of land used for dairy farms increased by 47% (1.4 to 2.0 million ha) from 1986 to 2002. [10] These statistics provide strong circumstantial evidence that the expansion in dairying is primarily responsible for the observed declines in water quality from agricultural sources.

15.5 REFERENCES

Airoldi, L (2003) The effects of sedimentation on rocky coast assemblages. *Oceanography and Marine Biology Annual Review 41*: 161–236.

Allen, M; Rosell, R; Evans, D (2006) Predicting catches for the Lough Neagh (Northern Ireland) eel fishery based on stock inputs, effort and environmental variables. *Fisheries Management and Ecology 13*: 251–260.

Atlantic States Marine Fisheries Commission (2000) Interstate Fishery Management Plan for American Eel Fishery Management. Report No. 36 of the Atlantic States Marine Fisheries Commission. 93 p.

Auckland Regional Council (2010) State of the Auckland Region: Chapter 4.4 Marine. Retrieved from http://www.aucklandcity.govt.nz/council/documents/technicalpublications/Chapter%204_0%20-%20State%20of%20the%20environment%20and%20biodiversity%20overview%20-%20%20and%204_1%20-%20Air.pdf.

Auckland Regional Council (2011) State of our Gulf Tikapa Moana – Hauraki Gulf. State of the environment report 2011. Retrieved from http://knowledgeauckland.org.nz/publication/?mid=1078&.

Balata, D; Piazzi, L; Cinelli, F (2007) Increase of sedimentation in a subtidal system: Effects on the structure and diversity of macroalgal assemblages. *Journal of Experimental Marine Biology and Ecology 351*: 73–82.

Ballantine, D; Davies-Colley, R (2009) Water quality trends at National River Water Quality Network sites for 1989–2007. *NIWA Client Report HAM2009-026*, prepared for Ministry for the Environment. 43 p.

Beentjes, M (2010) Toheroa survey of Oreti Beach, 2009, and review of historical surveys. *New Zealand Fisheries Assessment Report 2010/06*. 40 p.

Beentjes, M; Boubée, J; Jellyman, J D; Graynoth, E (2005) Non-fishing mortality of freshwater eels (*Anguilla* spp.). *New Zealand Fisheries Assessment Report 2005/34*. 38 p.

Bell, S S; Tewfik, A; Hall, M O; Fonseca, M S (2008) Evaluation of seagrass planting and monitoring techniques: Implications for assessing restoration success and habitat equivalency. *Restoration Ecology 16*: 407–416.

Bulleri, F (2006) Is it time for urban ecology to include the marine realm? *Trends in Ecology & Evolution 21*: 658–659.

Bulleri, F; Chapman, M (2010) The introduction of coastal infrastructure as a driver of change in marine environments. *Journal of Applied Ecology 47*: 26–35.

Carter, L (1975) Sedimentation on the continental terrace around New Zealand: A Review. *Marine Geology 19*: 209–237.

Carter, L; Carter, R; McCave, I; Gamble, J (1996) Regional sediment recycling in the abyssal Southwest Pacific Ocean. *Geology 24*: 735–738.

Castro, P; Huber, M (2003) Marine Biology. Volume 4. McGraw Hill Higher Education, New York.

Cloern, J (2001) Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series 210*: 223–253.

Cornelisen, C; Gillespie, P; Kirs, M; Young, R; Forrest, R; Barter, P; Knight, B; Harwood, V (2011) Motueka River plume facilitates transport of ruminant faecal contaminants into shellfish growing waters, Tasman Bay, New Zealand. *New Zealand Journal of Marine and Freshwater Research 45*: 477–495.

Cummings, V; Hewitt, J; Halliday, J; Mackay, G (2007) Optimizing the success of *Austrovenus stutchburyi* restoration: preliminary

[9] This is a known underestimate because streams with catchments less than 10 km^2 were excluded from this calculation.

[10] Statistics NZ (2006) Fertiliser Use and the Environment. Retrieved from http://www3.stats.govt.nz/environment/Fertiliser_use_and_the_environment_Aug06.pdf.

investigations in a New Zealand estuary. *Journal of Shellfish Research 26*: 89–100.

Davies-Colley, R J; Meleason, M A; Hall, G M J; Rutherford, J C (2009) Modelling the time course of shade, temperature, and wood recovery in streams with riparian forest restoration. *New Zealand Journal of Marine and Freshwater Research 43*: 673–688.

Department of Conservation and Ministry for Primary Industries (2012) Review of the Maui's dolphin Threat Management Plan: Consultation paper. MPI and DOC joint Discussion Paper No: 2012/18. Retrieved from http://www.doc.govt.nz/Documents/conservation/native-animals/marine-mammals/maui-tmp/mauis-tmp-discussion-document-full.pdf.

Diaz, R J; Rosenberg, R (2008) Spreading dead zones and consequences for marine ecosystems. *Science 321*: 926–929.

Elliot, A; Alexander, R; Schwarz, G; Shankar, U; Sukias, J; McBride, G (2005) Estimation of nutrient sources and transport for New Zealand using the hybrid mechanistic-statistical model SPARROW. *Journal of Hydrology: New Zealand 44*: 1–27.

Elliott, M; Burdon, D; Hemingway, K L; Apitz, S E (2007) Estuarine, coastal and marine ecosystem restoration: Confusing management and science - A revision of concepts. *Estuarine Coastal and Shelf Science 74*: 349–366.

Environmental Communications Limited (2010) Integrated catchment management - a review of literature and practice. A report for the Ministry for the Environment. 158 p.

Forrest, B; Gillespie, P; Cornelisen, C; Rogers, K (2007) Multiple indicators reveal river plume influence on sediments and benthos in a New Zealand coastal embayment. *New Zealand Journal of Marine and Freshwater Research 41*: 13–24.

Frew, R; Hunter, K; Beyer, R (1996) Cadmium in Oysters and Sediments From Foveaux Strait, New Zealand. Proceedings of the Trace Element Group of New Zealand, Waikato University.

Fukunaga, A; Anderson, M J; Webster-Brown, J G; Ford, R B (2010) Individual and combined effects of heavy metals on estuarine infaunal communities. *Marine Ecology Progress Series 402*: 123–136.

GESAMP (2001) Protecting the oceans from land-based activities. Land-based sources and activities affecting the quality and uses of the marine, coastal and associated freshwater environment. *GESAMP Reports and Studies No. 71*. 168 p.

Gibbs, M; Hobday, A; Sanderson, B; Hewitt, C (2006). Defining the seaward extent of New Zealand's coastal zone. *Estuarine Coastal and Shelf Science 66*: 240–254.

Glade, T (2003) Landslide occurrence as a response to land use change: A review of evidence from New Zealand. *Catena 51*: 297–314.

Goff, J R (1997) A chronology of natural and anthropogenic influences on coastal sedimentation, New Zealand. *Marine Geology 138*: 105–117.

Grange, K; Tovey, A; Hill, A F (2003) The spatial extent and nature of the bryozoan communities at Separation Point, Tasman Bay. *Marine Biodiversity Biosecurity Report No. 4*. 22 p.

Griffiths, G; Glasby, G (1985) Input of River-derived sediments to the New Zealand Continental Shelf: I. Mass. *Estuarine, Coastal and Shelf Science 21*: 773–784.

Haro, A; Richkus, W; Whalen, K; Hoar, A; Busch, W D; Lary, S; Brush, T; Dixon, D (2000) Population decline of the American eel: Implications for research and management. *Fisheries 25*: 7–16.

Hayward, B W; Grenfell, H R; Sabaa, A T; Morley, M S; Horrocks, M (2006) Effect and timing of increased freshwater runoff into sheltered harbor environments around Auckland City, New Zealand. *Estuaries and Coasts 29*: 165–182.

Heisler, J; Glibert, P; Burkholder, J; Anderson, D; Cochlan, W; Dennison, W; Dortch, Q; Gobler, C; Heil, C; Humphries, E; Lewitus, A; Magnien, R; Marshall, H; Sellner, K; Stockwell, D; Stoecker, D; Suddleson, M (2008) Eutrophication and harmful algal blooms: A scientific consensus. *Harmful Algae 8*: 3–13.

Hewawasam, T; von Blanckenburg, F; Schaller, M; Kubik, P (2003) Increase of human over natural erosion rates in tropical highlands constrained by cosmogenic nuclides. *Geology 31*: 597–600.

Hicks, D; Shankar, U (2003) Sediment from New Zealand rivers. NIWA Chart, Miscellaneous Series No. 79.

Hicks, D; Shankar, U; McKerchar, A I; Basher, L; Lynn, I; Page, M; Jessen, M. (2011) Suspended sediment yields from New Zealand rivers. *Journal of Hydrology (New Zealand) 50*: 81–142.

Hill, D; Dubey, J P (2002) *Toxoplasma gondii*: transmission, diagnosis and prevention. *Clinical Microbiology and Infection 8 (10)*: 634–640.

Hooker, S; Redfearn, P (1998) Preliminary survey of toheroa (*Paphies ventricosa*) populations on Ninety Mile Beach and possible impacts of vehicle traffic. *NIWA Client Report AK98042*. 34 p.

IPCC (2007) Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Pachauri, R; Reisinger A E (Eds.). Geneva, Switzerland. 104 p.

Kennedy, S; Kuiken, T; Jepson, P; Deaville, R; Forsyth, M; Barrett, T; Wilson, S (2000) Mass Die-Off of Caspian Seals caused by Canine Distemper Virus. *Emerging Infectious Diseases 6*: 637–639.

Kennish, M J (2002) Environmental threats and environmental future of estuaries. *Environmental Conservation 29*: 78–107.

Kneitel, J M; Perrault, D (2006) Disturbance-induced changes in community composition increase species invasion success. *Community Ecology 7*: 245–252.

Lipcius, R N; Eggleston, D B; Schreiber, S J; Seitz, R D; Shen, J; Sisson, M; Stockhausen, W T; Wang, H V (2008) Importance of metapopulation connectivity to restocking and restoration of marine species. *Reviews in Fisheries Science 16*: 101–110.

Lundquist, C J; Oldman, J W; Lewis, M J (2009) Predicting suitability of cockle *Austrovenus stutchburyi* restoration sites using hydrodynamic models of larval dispersal. *New Zealand Journal of Marine and Freshwater Research 43*: 735–748.

MacDiarmid, A; McKenzie, A; Sturman, J; Beaumont, J; Mikaloff-Fletcher, S; Dunne, J (2012) Assessment of anthropogenic threats to New Zealand marine habitats. *New Zealand Aquatic Environment and Biodiversity Report No.93*. 255 p.

McKenzie, J; Parsons, D M; Bian, R (2013) Can juvenile yellowbelly and sand flounder abundance indices and environmental variables predict adult abundance in the Manukau and Mahurangi Harbours? *New Zealand Fisheries Assessment Report 2013/10*. 31 p.

Marsden, I; Adkins, S (2009) Current status of cockle bed restoration in New Zealand. *Aquaculture International 18*: 83–97.

Marsden, I D; Taylor, G F (2010) Impact of vehicles on juvenile tuatua, *Paphies donacina* on Pegasus Bay surf beaches. *Environment Canterbury Estuarine Research Report 38*. 23 p.

Milliman, J; Syvitski, J (1992) Geomorphic/Tectonic Control of Sediment Discharge to the Ocean: The Importance of Small Mountainous Rivers. *The Journal of Geology 100*: 525–544.

Moller, J; Moller, S; Futter, J; Moller, J; Harvey, J; White, H; Stirling, F; Moller, H (2009) Potential impacts of vehicle traffic on recruitment of Toheroa (*Paphies ventricosa*) on Oreti Beach, Southland, New Zealand. Dunedin, University of Otago. 61 p.

Morrisey, D J; Gibbs, M M; Pickmere, S E; Cole, R G (2000) Predicting impacts and recovery of marine farm sites in Stewart Island, New Zealand, from the Findlay-Watling model. *Aquaculture 185*: 257–271.

Morrison, M; Jones, E; Consalvey, M; Berkenbusch, K (2014a) Linking marine fisheries species to biogenic habitats in New Zealand: a review and synthesis of knowledge. *New Zealand Aquatic Environment and Biodiversity Report No. 130*. 156 p.

Morrison, M; Lowe, M; Jones, E; Makey, L; Shankar, U; Usmar, N; Miller, A; Smith, M; Middleton, C (2014b) Habitats of particular significance for fisheries management: the Kaipara Harbour. *New Zealand Aquatic Environment and Biodiversity Report No. 129*. 169 p.

Morrison, M; Lowe, M; Parsons, D; Usmar, N; McLeod, I (2009) A review of land-based effects on coastal fisheries and supporting biodiversity in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 37*. 100 p.

Morrison, M; Parkinson, D (2008) Distribution and abundance of toheroa (*Paphies ventricosa*) on Ninety Mile Beach, March 2006. *New Zealand Fisheries Assessment Report 2008/26*. 27 p.

Nellemann, C; Hain, S; Alder, J E (2008) In Dead Water – Merging of climate change with pollution, over-harvest, and infestations in the world's fishing grounds. United Nations Environment Programme, GRID-Arendal. 64 p.

Nelson, W; Neill, K; Farr, T; Barr, N; D'Archino, R; Miller, S; Stewart, R (2012) Rhodolith Beds in Northern New Zealand: Characterisation of Associated Biodiversity and Vulnerability to Environmental Stressors. *New Zealand Aquatic Environment and Biodiversity Report No. 99*. 102 p.

Parkyn, S M; Davies-Colley, R; Halliday, N J; Costley, K J; Croker, G F (2003) Planted Riparian Buffer Zones in New Zealand: Do They Live Up to Expectations? *Restoration Ecology 11*: 436–447.

Piola, R F; Johnston, E L (2008) Pollution reduces native diversity and increases invader dominance in marine hard-substrate communities. *Diversity and Distributions 14*: 329–342.

Roni, P; Hanson, K; Beechie, T (2008) Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. *North American Journal of Fisheries Management 28*: 856–890.

Savage, C; Thrush, S; Lohrer, A; Hewitt, J (2012) Ecosystem services transcend boundaries: estuaries provide resource subsidies and influence functional diversity in coastal benthic communities. *PLoS ONE 7*: e42708.

Seaman, W (2007) Artificial habitats and the restoration of degraded marine ecosystems and fisheries. *Hydrobiologia 580*: 143–155.

Servizi, J A; Martens, D W (1992) Sublethal responses of coho salmon, *Oncorhynchus kisutch*, to suspended sediments. *Canadian Journal of Fisheries & Aquatic Sciences 49 (7)*: 1389–1395.

SPREP (2010) South Pacific Regional Environment Programme, Strategic Plan 2011–2015. 29 p.

Stephenson, G (1999) Vehicle impacts on the biota of sandy beaches and coastal dunes: A review from a New Zealand perspective. *Science for Conservation No. 121*. 48 p.

Stewart, M J; Creese, R G (2002) Transplants of intertidal shellfish for enhancement of depleted populations: preliminary trials with the New Zealand little neck clam. *Journal of Shellfish Research 21 (1)*: 21–27.

Taylor, F J; Taylor, N J; Walsby, J R (1985) A bloom of planktonic diatom *Ceratulina pelagica* off the coastal northeastern New Zealand in 1983, and its contribution to an associated mortality of fish and benthic fauna. *Internationale Revue der gesamten Hydrobiologie und Hydrographie 70*: 773–795.

Thrush, S; Hewitt, J; Cummings, V; Ellis, J; Hatton, C; Lohrer, A; Norkko, A (2004) Muddy waters: elevating sediment input to coastal and estuarine habitats. *Frontiers in ecology and the environment 2*: 299–306.

Thrush, S F; Hewitt, J E; Hickey, C W; Kelly, S (2008) Multiple stressor effects identified from species abundance distributions:

Interactions between urban contaminants and species habitat relationships. *Journal of Experimental Marine Biology and Ecology 366*: 160–168.

Tuckey, B; Gibbs, M; Knight, B; Gillespie, P (2006) Tidal circulation in Tasman and Golden Bays: implications for river plume behaviour. *New Zealand Journal of Marine and Freshwater Research 40*: 305–324.

Whetton, P; Mullan, A B; Pittock, A (1996) Climate change scenarios for Australia and New Zealand. *In*: Bouma, W; Pearman, G; Manning, M (Eds.), Greenhouse: coping with climate change. pp 145–168. CSIRO Publishing, Collingwood, Australia.

Wilkinson, R; McKergow, L; Davies-Colley, R; Ballantine, D; Young, R (2011) Modelling storm-event *E. coli* pulses from the Motueka and Sherry Rivers in the South Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research 45*: 369–393.

Williams, J; Paterson, C; Sim-Smith, C (2013) Review of factors affecting the abundance of toheroa (*Paphies ventricosa*). *New Zealand Aquatic Environment and Biodiversity Report No. 114*. 76 p.

16 ECOLOGICAL EFFECTS OF MARINE AQUACULTURE

Status of chapter	This chapter has not been updated for AEBAR 2018.
Scope of chapter	The known effects of current impacts from aquaculture operations in New Zealand.
Area	All of the New Zealand EEZ and territorial sea, although presently aquaculture operations are located coastally.
Focal localities	Northland, Coromandel, Auckland, Marlborough Sounds, Tasman and Golden Bays, Canterbury, Southland.
Key issues	Uncertainty in predictions, cumulative effects, levels of nitrogen loading in coastal areas that will cause adverse effects
Emerging issues	Marine spatial planning, Integration of monitoring datasets.
MPI research (current)	ENV2012-01 *Nitrogen levels and adverse marine ecological effects* Aquaculture Planning Fund 12/03 *Marine Management Model (Waikato Regional Council)*; 12/04 *Guidance for aquaculture monitoring in the Waikato region*; 13/01 *Marlborough Sounds Hydrodynamic & Ecological Modelling*; 13/02 *Aquaculture Zoning in the Southland Region.*
NZ research (current)	C01X0904 *NIWA Sustainable Aquaculture*
Related issues	Land-based effects, marine biodiversity, habitats of particular significance for fisheries management.

16.1 CONTEXT

Aquaculture is the world's fastest growing primary industry and in 2011 supplied 41.2% of the supply of seafood globally, including 12.5% from marine aquaculture in the same year (FAO 2012). Fish convert a greater proportion of the food they eat into body mass than livestock and therefore the environmental demands per unit biomass or protein produced are lower (Hall et al. 2011). The production of 1 kg of finfish protein requires less than 14 kg of grain compared to 62 kg of grain for beef protein and 38 kg for pork protein. However, although farmed fish may convert food more efficiently than livestock there are important issues globally with respect to farming carnivorous fish species, which places demands on the use of capture fisheries for animal feeds.

In 2011 the Oceania region (which includes New Zealand and Australia) produced only 0.3% of the world's aquaculture production (183 516 t); globally nearly 60 million t were produced (FAO 2012). The average annual value of New Zealand aquaculture exports from 2008 to 2012 has been dominated by green-lipped mussels ($197 million), Salmon ($61 million) and Pacific oysters ($16 million) (Aquaculture New Zealand 2012). As of December 2011, aquaculture activities in New Zealand take place within approximately 19 268 ha of allocated water space (Aquaculture New Zealand 2012). This space can be categorised as below (Aquaculture New Zealand 2012):

- 7743 ha is granted to the aquaculture industry with the right to farm for a defined term, and is in known productive growing areas;
- 8960 ha is in open-ocean sites where productivity is yet to be proven;
- 1195 ha is in near shore sites yet to be developed;
- 1370 ha is undeveloped space in interim Aquaculture Management Areas (AMAs).

In New Zealand, the majority of aquaculture activities are located in the coastal marine environment, and the main current aquaculture locations are shown in Figure 16.1.

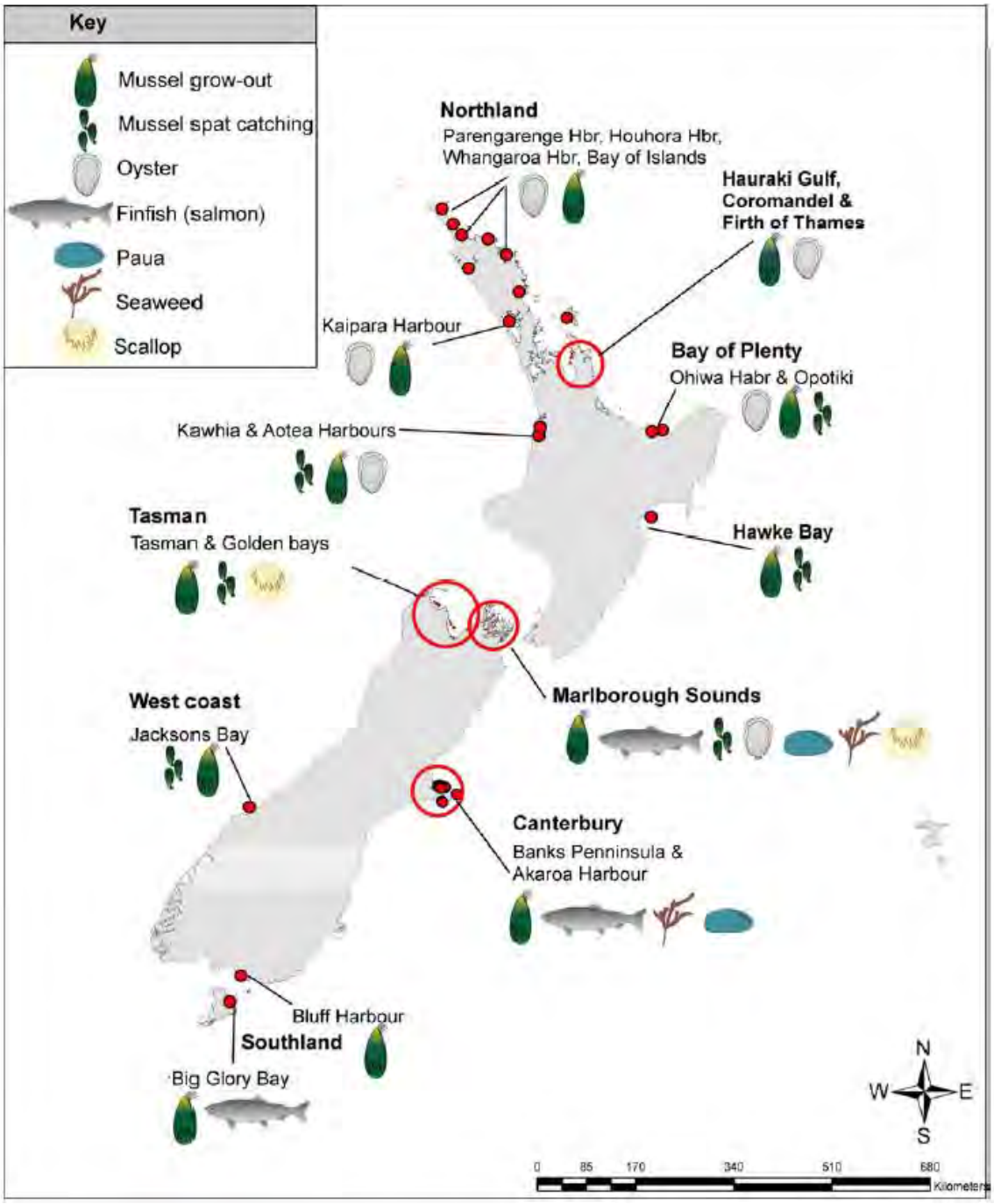


Figure 16.1: Geographic locations of main marine farming areas in New Zealand (Keeley et al. 2009).

The New Zealand aquaculture industry has a current estimated value in excess of $400 million and an objective of developing into a billion dollar industry by 2025 (Aquaculture New Zealand 2012). This ambition has been supported by the New Zealand government through the establishment of the Aquaculture Unit (now within the Ministry for Primary Industries (MPI)), the release of Government's Aquaculture Strategy and 5-Year Action Plan to support aquaculture, the 2011 aquaculture legislation reforms, and ongoing reforms of the Resource Management Act (RMA). One of the desired outcomes of these actions was to improve the consenting process to enable more space to be made available for aquaculture. To this end a number of Aquaculture Planning Fund projects have been initiated to address factors limiting aquaculture growth regionally. It is however recognised that aquaculture development, along with all other activities controlled by the RMA, needs to be ecologically sustainable.

Sustainable development of aquaculture in New Zealand needs to be supported by good quality information on

ecological effects to enable appropriate decision making. The aquaculture unit of MPI therefore funded a collaborative project between NIWA and the Cawthron Institute to review the ecological effects of aquaculture (PRM2010-36). This chapter largely summarises the findings of that larger document (MPI 2013), which should be referred to for further details, references or clarification.

16.2 GLOBAL UNDERSTANDING

It is known that the environmental effects of aquaculture vary by country, region, production system and species (Hall et al. 2011). Ninety-one percent of the world's aquaculture production comes from Asia and only 0.3% from Oceania (Hall et al. 2011); therefore global reports on the environmental impacts of aquaculture tend to focus on Asia. The relevant (as judged by the authors of MPI 2013) references to New Zealand from overseas literature will hence be included in Section 16.3.

16.3 STATE OF KNOWLEDGE IN NEW ZEALAND

A 2009 survey of experts assessed the relative importance of 62 threats on 65 of New Zealand's marine habitats (MacDiarmid et al. 2012). Threat scores were categorised as extreme if the score was 3 or more, major if the score was 2–2.9, moderate if the score was 1–1.9, minor if the score was 0.5–1.0, and trivial if the score was less than 0.5. For example, the three top threats identified across all habitats were ocean acidification, increased sea temperatures from climate change and bottom trawling, which scored mean impacts across all habitats of 2.6 (major), 1.6 (moderate) and 1.5 (moderate), respectively. The study considered three threats posed by aquaculture activities: benthic accumulation of debris (shells, faeces, and food material), a decrease in the availability of primary production downstream of the marine farm (particularly mussel farms) and an increase in habitat complexity that may be detrimental to some species. The benthic accumulation of shells, food and faeces from aquaculture ranked 19th equal with a score of 0.7 (minor). The two other aquaculture threats were ranked 36th equal with a score of only 0.4 (trivial). Notably this is an average score across all habitats, however the highest scores attained for any of these aquaculture threats in particular habitats were 2.6 and 2.3 for the benthic accumulation of debris (shells, faeces, food material) in muddy sediment on sheltered coasts (2–9 m) and seagrass meadows in harbours and estuaries, respectively. The benthic accumulation of debris was the fourth most highly scoring threat in sheltered muddy coasts (2–9 m deep) and the third most highly scoring threat in seagrass meadows in harbours and estuaries.

The actual and potential effects of filter feeding and feed-added culture are shown diagrammatically in Figure 16.2 and Figure 16.3.

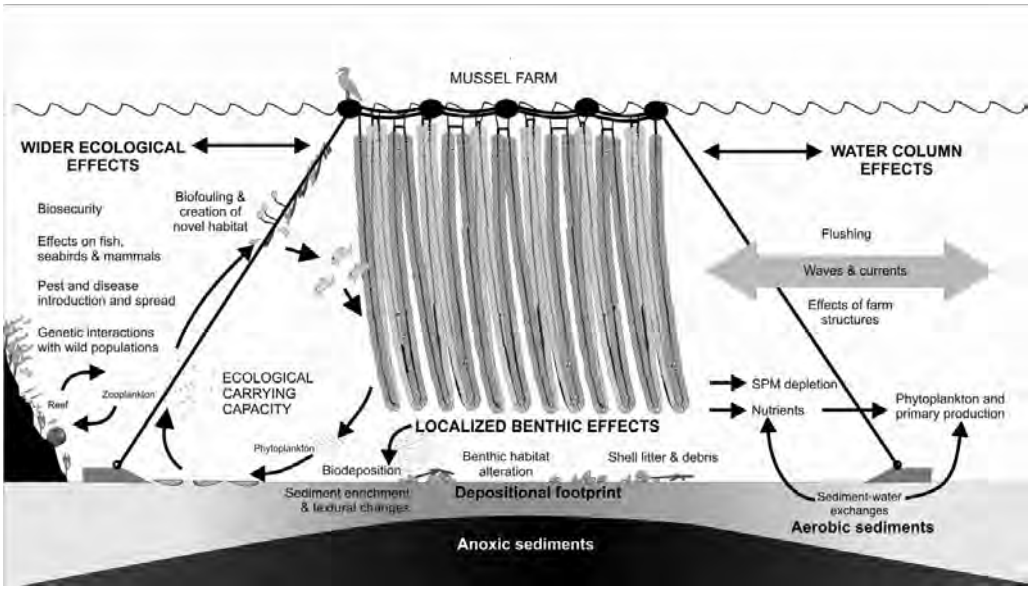


Figure 16.2: Schematic of actual and potential ecological effects from mussel farming (Keeley et al. 2009).

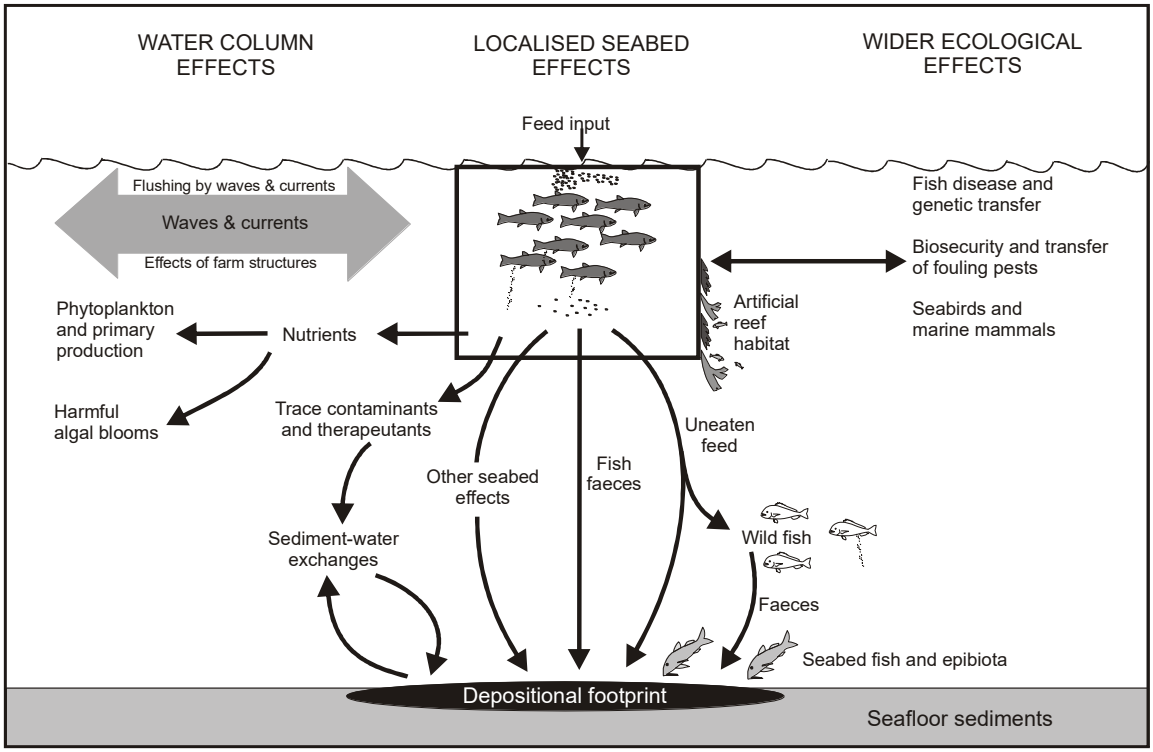


Figure 16.3: Schematic of actual and potential ecological effects from feed-added farming (Forrest et al. 2007c).

An expert panel approach was also used to trial a method for prioritising the ecological threats from aquaculture (Stoklosa et al. 2012). This process brought together 17 knowledgeable participants from across a range of interested parties (central and local government, aquaculture industry and scientists), to attempt to gain consensus on the relative importance of a range of ecological threats from aquaculture. The results of this process are only indicative but for both feed-added and filter-feeding species the same three issues were identified as most important; these were (in decreasing order of importance): biosecurity threats, pelagic effects and marine mammal interactions (Table 16.1). Notably the score for the threat from biosecurity was more than 50% greater than the next highest score and the threat of pelagic effects was rated as markedly higher for feed-added species than it was for filter-feeders. Other potential ecological threats considered were of lesser importance and are listed bullet pointed below the top three, along with an explanatory sentence about what was considered under each term (in no particular order). Interactions between threats and large scale effects were not covered within this prioritisation exercise.

1. Biosecurity threats – how aquaculture may influence risks associated with pests and diseases.
2. Pelagic effects – aquaculture effects on the water column (excluding those explicitly dealt with by other chapters in the MPI 2013 literature review) at approximately the scale of the farm.
3. Marine mammal interactions – aquaculture effects on marine mammals.

- Benthic effects – aquaculture effects on the seafloor.
- Seabird interactions – aquaculture effects on birds.
- Effects from additives – The effect of chemicals used in aquaculture upon the environment.
- Escapee effects – the effects of escaped farmed species upon the environment.
- Wild fish interactions – aquaculture effects on non-farmed fish populations.
- Hydrodynamic alteration of flows – aquaculture effects on the water movement at scales greater than the farm scale.

Table 16.1: Trial prioritisation of potential classes of aquaculture effects from Stoklosa et al. (2012). Results of pair-wise comparisons using the Analytical Hierarchy Process (Saaty 1987) from the phase two workshop of the Aquaculture Ecological Guidance Project. RIW = relative importance weight. Order is decreasing in importance for the feed-added species.[1]

	Feed-added species		Filter-feeder species	
Potential ecological effects	RIW	Rank	RIW	Rank
Biosecurity threats	0.360	1	0.373	1
Pelagic effects	0.236	2	0.143	2
Marine mammal interactions	0.118	3	0.135	3
Benthic effects	0.090	4	0.088	5
Seabird interactions	0.079	5	0.092	4
Additive effects	0.042	6	0.019	9
Escapee effects	0.029	7	0.088	5
Wild fish interactions	0.026	8	0.021	8
Hydrodynamic alteration of flows	0.019	9	0.041	7

These topic areas will be discussed further under each of their headings below (in the order above). In addition, note that stressors do not act in isolation, and any aquaculture impacts will occur within the context of (and potentially interacting with) other anthropogenic stressors and natural ongoing natural processes (see Figure 13.4 for an example of this). The interacting and cumulative effects of aquaculture will be discussed in Section 16.3.10 of this chapter.

16.3.1 BIOSECURITY THREATS

Aquaculture biosecurity has recently been covered by the reviews of Forrest et al. (2011) for finfish and Keeley et al. (2009) for other species, and then compiled and summarised in MPI (2013), this section draws heavily from those sources, and the reader is referred to them for more detail.

16.3.1.1 INTRODUCTION

The Ministry of Agriculture and Forestry (MAF) Biosecurity Strategy defines biosecurity as 'the exclusion, eradication or effective management of risks posed by pests and diseases' (Biosecurity Council 2003). Biosecurity risk organisms include animals, plants and micro-organisms capable of causing diseases (e.g., the ostreid herpes virus in Pacific oysters) or otherwise adversely affecting New Zealand's natural, traditional or economic values (e.g., the sea squirt *Styela clava*, and the red seaweed *Grataloupia turuturu*). In an aquaculture context, biosecurity also encompasses the protection of hatchery or culture operations from parasites, microscopic pathogens [2] or biotoxin-producing microalgae. These organisms may include indigenous species already present in the environment that become enhanced as a result of culture operations (Forrest et al. 2011).

The primary source of entry for biosecurity risk organisms into New Zealand is through international shipping (Cranfield et al. 1998, Kospartov et al. 2010). However, aquaculture production systems may increase biosecurity risk, through acting as reservoirs or exacerbators (Okamura & Feist 2011, Peeler & Taylor 2011). Reservoirs host risk-organisms that can then spread by either natural or human-mediated mechanisms. Exacerbators create incubators/stepping stones for otherwise benign or low impact pests, pathogens or parasites (both native and exotic species).

Considerable effort is placed on preventing incursions of pests, parasites and diseases into the New Zealand environment. This is because the introduction, proliferation and spread of risk species in New Zealand can have effects on marine and freshwater environments that are often difficult to manage, resulting in permanent and irreversible impacts (Forrest et al. 2011). The few successful efforts to eradicate aquatic invasive species (AIS) have several

[1] Notably there was a chapter in MPI (2013) on the potential effects from genetic manipulation and polyploidy. However, genetic manipulation is controlled by the Environmental Protection Authority (EPA) and is not authorised for use in aquaculture. Polyploidy was also considered by the risk assessment workshop participants to be relatively rare in aquaculture and therefore this topic area was not considered by the prioritisation.

[2] Defined here as an agent of disease, e.g., a bacterium or virus.

common elements (Locke et al. 2009b), which are unlikely to occur in combination:

- early detection and correct identification of the invader,
- pre-existing authority to take action,
- the ability to sequester the AIS to prevent dispersal, (or else the AIS had very limited dispersal capabilities),
- political and public support for eradication,
- acceptance of some collateral environmental damage,
- follow-up monitoring to verify the completeness of the eradication.

Environmental factors including depth, wave climate, temperature regime, and currents that influence dispersal of waste, disease agents, and pests play a significant role in determining the potential biosecurity risk for a given site.

The hydrodynamics (water movement patterns that are dependent on depth, wave climate and currents) at a site play an important role on several levels. Hydrodynamics can influence the mineralisation of wastes and nutrient release through oxygen supply to the sediment and also dispersion of pathogens and pests and parasites in the water column (Zeldis et al. 2011b). For example, individual farms within any one Aquaculture Management Area (AMA) in Nelson Bays could function as a source of infection to other AMAs in Golden Bay (Zeldis et al. 2011b) via the transfer of viral or bacterial pathogens. Dispersion potential (within farms, between farms or between blocks of farms), which is largely controlled by hydrodynamics, will also be influenced by temperature, as temperature can regulate metabolic growth and the proliferation of bacteria/viruses etc. that are shed as free-living single-celled organisms (Zeldis et al. 2011b).

Temperature and salinity can also affect the associated biosecurity risks associated with individual species by controlling their range. For example in the case of the proliferation of invasive Pacific oysters, the southern distribution is limited to Nelson/Marlborough, as water temperatures further south are too low for successful reproduction (Quale 1969, Askew 1972, Dinamani 1974). Salinity can vary with season, climatic variation (Scavia et al. 2002), and the catchment rainfall, with catchments that are dry in summer producing less runoff, elevating coastal salinities, which then affect the distribution of fouling species (Handley, unpub. data). Farm stocks that may be susceptible to biosecurity risks are usually at greatest risk in summer. Summer is when temperatures, and hence metabolic rates of farmed animals, are highest, dissolved oxygen levels in the water are lowest (hence the risk of oxygen deprivation is highest), and the proliferation of fouling populations is also greatest (Handley, unpub. data.).

Over the last decade aquaculture space allocation in New Zealand has predominantly been driven by constraint mapping, allocating space in areas that do not conflict with other users and stakeholders (e.g., Handley & Jeffs 2002). This strategy increases potential biosecurity risks by encouraging development of aquaculture at environmentally less favourable sites. The use of ecosystem-based approaches to aquaculture development that incorporate tools like GIS can incorporate biosecurity risks (if known) to optimise site selection even in cases of data poor environments (Aguilar-Manjarrez et al. 2010, Soto et al. 2008, Silva et al. 2011).

16.3.1.2 SIGNIFICANCE OF EFFECTS

It is generally recognised that adverse ecological effects arising from pests, parasites and pathogenic species associated with aquaculture can result in a range of level of threat including (Molnar et al. 2008):

a. disruptions to entire ecosystem processes with wider abiotic influences,
b. disruptions to wider ecosystem function, and/or keystone species or species/assemblages of high conservation value (e.g., threatened species),
c. disruptions to single species with little or no wider ecosystem impact,
d. little or no disruption.

The infection of marine farms by pest organisms can lead to the development of significant infestations on farm structures, which may then:

1. act as a reservoir for subsequent spread to natural ecosystems,
2. increase drag on cages and anchoring systems in high current areas, which in turn increases the chance of escapee effects if stocks are infected with pathogens or parasites (Forrest et al. 2011),

3. significantly reduce the flow of water (in areas of lower current velocity), carrying vital food and oxygen to cultured species.

Examples of significant effects from pest fouling organisms on aquaculture activities in New Zealand include documented impacts from infestation of marine farms with *Undaria* and the colonial tunicate *Didemnum vexillum* (e.g., Forrest & Taylor 2002 and L. Fletcher, Cawthron, unpub. data). As well as attached fouling organisms, aquaculture structures may also act as recruitment substrata for mobile pelagic or benthic species (e.g., jellyfish, ctenophores, sea star *Asterias amurensis*, sea cucumbers, or the crab *Carcinus maenas*; Forrest et al. 2009, 2011).

Any attempt to assess the significance of potential effects of invasive pests, pathogens or parasites in terms of their magnitude will be limited by the lack of robust information on the affected environments, inherent difficulties in making reliable predictions regarding the invasiveness of difference species, and hence inferences regarding their direct or indirect effects (Forrest et al. 2011). An example of the ecological effects stemming from a pathogen is the outbreak of pilchard herpes virus that was thought to have stemmed from pilchards imported for tuna aquaculture feed in South Australia. This event caused starvation and the recruitment failure of little penguins, which prey on pilchards (Dann et al. 2000). The potential effects of pests and pathogens are illustrated in Table 16.2 for finfish aquaculture in the Waikato region.

Table 16.2: Matrix illustrating the often unknown effects of pests, pathogens and parasites associated with finfish aquaculture in the Waikato Region. Examples are given of direct interactions (shaded cells) between potential biosecurity hazards and values in the Waikato region, and indirect effects (I). Direct interactions designated as: likely to be new and important (***), may be an important incremental risk above that already occurring (**), and probably a minor incremental risk (*). ? = direct interaction possible but significance unknown. From Forrest et al. (2011).

Potentially affected uses and values	Component directly affected	Marine pests			Pathogens or parasites		
		Fouling	Predation	HABS	Virus	Monogeanean	Digenean
Ecological							
Habitats and their biodiversity	Unstructured soft-sediment habitats	*	**	?			
	Structured soft-sediment habitats (physical or biogenic)	**	**	?			
	Zostera meadows	*		?			
	Saltmarsh			?			
	Rocky reef	**	**	?			
	Water column (plankton communities)			?			
Wildlife of conservation importance	Wading and seabirds	I	I	I	?+I		?
	Marine mammals	I	I	I	?+I		?
Wild fishery resources and fishing							
Finfish populations of commercial, recreational or customary importance	Conspecific finfish populations (kingfish or hapuku)			?	?	*	*
	Pelagic finfish populations (e.g. snapper, kahawai)			?	?	*	*
	Benthic finfish (e.g. flatfish) or reef-fish populations	I	I	?	?	*	*
Shellfish populations of commercial, recreational or customary importance	Infaunal soft-sediment shellfish (e.g. cockles, tuatua)	*	?	?	?		?
	Epibenthic soft-sediment shellfish (e.g. scallops)	**	?	?	?		?
	Reef-associated non-finfish species (e.g. paua, crayfish)	**	?	?	?		?
Harvesting of fish/shellfish (interference)	Pelagic finfish populations (e.g. snapper, kahawai)						
	Benthic finfish (e.g. flatfish) or reef-fish populations	*	*				
	Infaunal soft-sediment shellfish (e.g. cockles, tuatua)		*				
	Epibenthic soft-sediment shellfish (e.g. scallops)	**	*				
	Reef-associated non-finfish species (e.g. paua, crayfish)	*	*				
Harvesting of fish/shellfish (contamination)	Finfish or shellfish harvestability for human consumption			?	?	?	?

16.3.1.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Biosecurity control of aquaculture activities currently occurs through: resource consent conditions, farm practices and import health standards. The resource consenting process under the Resource Management Act (RMA) considers biosecurity via factors such as farm spacing, zoning,[3] staged development and epidemiological units. Best farm practices are often described by industry codes of practice (NZMIC 2001, NZOIA 2007, NZSFA 2007). Import health standards are controlled by the Ministry for Primary Industries (MPI) and include requirements that must be met in the exporting country, during transit and on arrival. For example, existing standards cover:

- import of juvenile yellowtail kingfish (*Seriola lalandi*) from Australia,
- import of fish food and fish bait from all countries.

Possible prevention approaches that could be considered are summarised here as pathway management or on-farm management Forrest et al. (2011).

Pathway management should focus on controls and surveillance on pathways from:

i. international source regions or pathways that are novel,
ii. pathways from domestic source regions known to be infected by recognised high-risk pests,
iii. pathways along which the frequency of transfers is considerably greater than that occurring as a result of other human activities.

Broadly there are two approaches to management of pathway risk (Forrest & Blakemore 2002), either a) avoid transfers on high risk pathways, or b) treat pathways to minimise risk. Both pathway management strategies have been used, for example, in relation to the New Zealand mussel industry (Forrest et al. 2011). Surveillance strategies for pathways can focus on entry surveillance, routine surveillance or targeted surveillance of high-risk areas. Entry surveillance includes activities such as routine screening at airports, ports and mail centres. MPI also commissions routine surveillance in ports and harbours around New Zealand. Targeted surveillance may be undertaken when activities such as harvest, grading or transfer of stock from hatcheries or between sites is undertaken.

Good on-farm management is often guided by industry codes of practice (NZMIC 2001, NZOIA 2007, NZSFA 2007). These should include farm cleaning and surveillance (MPI 2013). Farm cleaning guidelines should deal with factors such as frequency and waste disposal. Routine surveillance, undertaken on and around marine farms is often the first point of detection of pests, pathogens and diseases.

Recent New Zealand experience suggests that even when pest organisms become well established, the benefits gained from even limited management success have the potential to greatly outweigh the consequences of uncontrolled fouling (Forrest et al. 2007a). To be effective, however, management requires buy-in from all marine stakeholders whose activities can spread pest organisms. Aquaculture companies can assist by:

a. identifying existing and future pests that threaten the aquaculture industry,
b. implementing surveillance of farm structures and associated vessels and infrastructure,
c. developing coordinated response plans for high-risk species before they become established,
d. preventing incursions of new pests onto aquaculture structures.

For vectors of spread such as service vessels and farm equipment, preventative management options include:

i. maintenance of effective antifouling coatings,
ii. hull inspections and hull cleaning as necessary,

[3] The World Organisation for Animal Health's (OIE) online aquatic animal health code (http://www.oie.int/en/international-standard-setting/aquatic-code/access-online) suggests establishing zones and using compartmentalisation (through geographical separation) to manage biosecurity and epidemiological risks.

iii. early eradication of pests from farm structures before they become well established.

However, once incursions have occurred, the use of eradication treatments is only advised if the risk of re-invasion can be managed. Many eradication treatments have been used in an attempt to control fouling and pests either directly (Carver et al. 2003, Coutts & Forrest 2005, Locke et al. 2009a, Morrisey et al. 2009), indirectly (Handley & Jeffs 2002, Handley 2002, Handley & Bergquist 1997) or via biological control agents (NRC 2010, Hidu et al. 1981, Enright et al. 1983, 1993, Cigarria et al. 1998).

Perhaps the best method for controlling the spread of disease is through the use of management practices that call for the pathological inspection of animals to ensure that infected animals are not moved into areas that do not currently have endemic infections (WWF 2010). In New Zealand, in the absence of enforced stock transfer protocols, management of gear and vessel transfers between geographic zones by voluntary codes of practice developed by industry could be used to minimise risks, e.g., the New Zealand Mussel Industry Council Ltd. code of practice for transfer of mussel seed (NZMIC 2001).

The different prospective farmed groups: feed-added (referred to as finfish), filter-feeders (referred to as shellfish), and lower trophic level species (*Undaria* and sea cucumbers) and their potential impacts and management measures were covered in the literature review (MPI 2013) and are summarised in Table 16.3.

Table 16.3: Matrix of biosecurity management options and their relevance to key aquaculture groups (MPI 2013). [Continued on next page]

Management measure	Description	Finfish	Shellfish	Undaria	Sea cucumbers
Harvest					
Isolate waste streams from growing areas	Prevent reintroduction of pests/pathogens to harvested sites	y	y	y	y
Fallow Sites	Reduce opportunities for reintroduction of pests/pathogens from intermediate hosts	y	y	y	y
Education					
Codes of practice	Educate and alert staff to biosecurity requirements	y	y	y	y
Public notification	Alert public to biosecurity risks	y	y	y	y
Eradication					
Culling	Cull diseased stock to remove pathogen/ pest	y	y	y	y
Fallowing	Remove stock from an area to allow host mediated pathogen to die out	y	y	y	y
Manual removal of macroscopic organisms	Eradication of individual pest organisms early in the invasion process	y	y	y	y
Treatment technologies	Treatment of whole farms or bays to remove pests	y	y	y	y
Pharmaceutical treatment	Treatment of individual affected stocks to remove pathogen/parasite	y	n	n	n

Table 16.3 [Continued]:

Management measure	Description	Finfish	Shellfish	Undaria	Sea cucumbers
Import					
Import health standards	For import of seedstock	y	n	n	n
Boarder Surveillance	Prevent import of macroscopic pests	y	y	y	y
Regulations on fouling on vessels/bilge water release	Prevent import of macroscopic pests/ fouling organisms/ harmful algae	y	y	y	y
Planning and development					
Site selection	Sites with appropriate environment for biological requirements of stock	y	y	y	y
Zoning	Sites location in relation to pathogen risks – other farms, processing plants, rivers, sewerage discharge	y	y	y	y
Vessel berthing	Segregate local vessels from vessels that move regionally (commercial or recreational)	y	y	y	y
Targeted surveillance	Routine monitoring for pre-determined range of species	y	y	y	y
Farm practices					
Fouling					
Management of nets, and equipment to minimise fouling	Regularly remove fouling organisms from equipment	y	y	n	n
Anti-fouling	Treat equipment with chemicals to prevent fouling	y	?	n	n
Transfer of equipment between sites/ regions	Prevent transfer of potentially contaminated equipment between sites				
Husbandry					
Appropriate stock husbandry	Minimise stress = reduce risk of disease becoming established	y	y	y	y
Management of feed so as not to attract birds/fish	Limit opportunity for transfer between sites/wild stocks through direct contact	y	n	n	n
Routine environmental monitoring linked to husbandry activities	Manage stock within environmental limits	y	y	y	y
Remove mortalities	Limit opportunity for reservoir of disease to accumulate	y	n	n	n
	Reduce attraction of predators	y	n	n	n
Use of processed feeds	Feeds heat treated to kill pests/pathogens	y	n	n	y
Surveillance	Observe and record mortality causes, unusual fouling etc.	y	y	y	y
Stock transfer		y	y	y	y
Hatchery testing for disease	Prevent diseased stock being sent to sites	y	y	y	y
Single year-class sites	Prevent disease transmission between year classes	y	n	y	y

16.3.2 PELAGIC EFFECTS

There is a large volume of international literature on the effects of shellfish and salmon farming on the pelagic environment and much of this material is referenced in three local reviews: finfish (Forrest et al. 2007a), shellfish (Keeley et al. 2009) and oysters (Forrest et al. 2007b) and summarised in MPI (2013), the reader is referred to these for more detail.

16.3.2.1 INTRODUCTION

This section deals with near-field (approximately at the scale of the farm) pelagic effects (those seen in the water column). This should be read in conjunction with the benthic effects (where wastes from the pelagic zone settle) and the cumulative effects sections (where far-field pelagic effects are seen).

The pelagic zone is the zone where:

- Filter-feeders extract phytoplankton, microzooplankton and organic particulates from the water column, which can reduce food available to other consumers (Zeldis et al. 2004).
- Dissolved oxygen (DO) is extracted by respiration of farmed organisms and this can potentially lead to DO depletion when cages are heavily stocked or where they are located in shallow sites with weak flushing (La Rosa et al. 2002). Excessive DO depletion in the water column could potentially stress or kill the fish and other animals, with sediment DO depletion resulting in the release of toxic by-products (e.g., hydrogen sulphide) into the water, which can also have adverse effects on fish and other organisms (Forrest et al. 2007a).
- Fish pellets and the excretory products and waste products of cultured and fouling organisms are received. Wastes excreted can either be as a particulate 'cloud' that disperses rapidly, in the case of fin-fish, or be bound in long strands composed of digested and undigested plankton, in the case of filter-feeders (Reid 2007). The difference in shellfish and finfish faeces can result in different biochemical impacts on the pelagic zone (Reid 2007). Dissolved farm waste has the potential to increase ambient DIN (Dissolved Inorganic Nitrogen), the potential effects of this are usually experienced away from the farm so will be dealt with in the cumulative effects section.

16.3.2.2 SIGNIFICANCE OF EFFECTS

The significance of these key primary impacts depends on the assimilation capacity (or carrying capacity) of the environment. Local hydrodynamics, water depth and ambient oxygen levels are the most critical criteria for determining the pelagic impacts of aquaculture (Zeldis 2008a, Zeldis et al. 2010, 2011a). In shallow areas with slow currents, effects will be more pronounced compared to a deep site with strong flow and good flushing. In the New Zealand situation where most shellfish farms are located in well-flushed areas, nutrient enrichment beyond the farm boundaries is presently difficult to detect (Zeldis 2008a). In addition there are a number of design and management factors that will greatly influence potential impacts:

- Density of farms in a unit volume of water; more farms will generally have more effect.
- Stocking density; higher stocking densities will generally have more effect, this may differ seasonally.
- Feed conversion ratio (FCR for feed-added species): FCR is a measure of the efficiency of growth relative to feed used, the global range is 1.1 to 1.7 on average (Reid 2007). The lower the FCR the less waste will be produced.
- Cage designs and orientation to prevailing current direction. This will impact on drag on passing water masses, flushing of cages and settlement of biofouling organisms.

Undaria and sea cucumbers have less significant ecological effects on the pelagic environment since seaweeds utilise dissolved nutrients for growth (mainly dissolved inorganic nutrients (DIN)) and sea cucumbers feed on organic material on the surface of the seabed (MPI 2013). The reader is guided to the document MPI (2013) for coverage of the specific threats created via farming *Undaria* and sea cucumbers.

16.3.2.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Pelagic effects can be partially controlled through carefully selecting sites, deep sites (more than 25 m) with high currents are preferable. The farm design, orientation and stocking rates should then be appropriate to that site. Good farm management (e.g., compliance with The New Zealand Finfish Aquaculture Environmental Code of Practice (2007)[4]) should include reducing biofouling on nets by regular cleaning and removal of biofouling waste. Monitoring, adaptive management and the use of

[4] A copy of these codes can be obtained from Aquaculture New Zealand (www.aquaculture.org.nz).

Integrated Multi Trophic aquaculture (IMTA) are also potential mitigation measures (see the cumulative effects section for more discussion of these). Notably pelagic effects are reversible upon removal of the farm.

Models are an important component in determining pelagic effects at a site and a number of potential model improvements are identified in MPI (2013), including improved methods for determining ecological carrying capacity.

16.3.3 MARINE MAMMALS

The reader is referred to MPI (2013) (and references therein) for more detail.

16.3.3.1 INTRODUCTION

Several overseas studies (Würsig & Gailey 2002, Kemper et al. 2003, Wright 2008) have characterised the possible interactions between marine mammals and aquaculture, which include:

- competition for space (habitat modification or exclusion),
- potential for entanglement,
- underwater noise disturbance,
- attraction to artificial lighting,
- possible flow-on effects due to alterations in trophic pathways.

The physical location of the farm within important habitats or migration routes of New Zealand marine mammal species is the main factor that leads to potentially adverse interactions or avoidance issues. Once a farm is within the habitat or migration route of a species, the types of gear and equipment employed, as well as operational procedures around regular farm activities, influence the probability and scale of the impacts discussed above.

16.3.3.2 SIGNIFICANCE OF EFFECTS

Incidences of marine mammal entanglement with aquaculture operations are very few in New Zealand despite over 25 years of sea-cage salmon farming, due in part to the relatively small scale of this industry and operational procedures that minimise entanglement risk at New Zealand farms (Forrest et al. 2007c). Studies in New Zealand have so far only addressed interactions between mussel farms with Hector's (Slooten et al. 2001) and dusky dolphins (Markowitz et al. 2004, Vaughn & Würsig 2006, Duprey 2007, Pearson et al. 2007). Collectively, these works suggest that while some marine mammal species are not completely displaced from regions as a whole, they do not appear to be utilising habitats occupied by shellfish farms in the same manner as prior to the farms' establishment.

These effects may need to be reconsidered in relation to any larger scale and offshore developments in New Zealand waters (MPI 2013). For instance, as multiple farms or several types of aquaculture begin to overlap or enlarge in their locations, marine mammal populations may be excluded from particular bays or regions depending on the species and its sensitivity to such activities. In the case of depleted populations (e.g., southern right whales), the issues of low population size and a fairly isolated population structure make these species more vulnerable to such impacts than other species. This large variation in the significance of aquaculture impacts (depending on the size of the affected populations) on New Zealand marine mammals makes developing and implementing one set of effective management guidelines or standards extremely difficult.

16.3.3.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Farm locations need to be carefully selected to minimise the likelihood of overlap with marine mammal migration routes and/or known habitats. In Admiralty Bay, where overlap with dusky dolphins was a concern, and distribution patterns were not well known, three years' worth of presence monitoring was required prior to commencement of aquaculture development (Mulcahy & Peart 2012). The risks associated with physical interactions can be further minimised by adopting maintenance and operational guidelines and standards for farm structures as well as any noise-generating equipment (British Columbia Shellfish Growers Association 2001, SAD 2011). Some examples include enclosing predator nets at the bottom, keeping nets taut, using mesh sizes of less than 6 cm (Kemper et al. 2003), keeping nets well maintained (e.g., repairing holes), and reducing feed waste. In Admiralty Bay surface lines were removed from the water over winter to minimise interactions when dolphins are more active foragers (Mulcahy & Peart 2012).

Unfortunately, detailed information on abundance, distribution and critical habitats is available for only a handful of New Zealand's marine mammals. Monitoring records of the presence (and absence) of marine mammal species in the vicinity or general region of the farm site along with any detailed observations of their time spent under or around the farm structure should be compiled when possible. Future research needs to focus on those species most likely to come in contact with aquaculture in the future. In addition, ongoing research into the types of design and maintenance features and operational procedures that minimise entanglement risk should be supported. For example, cage technology in South Australia has developed and improved to the point where predators are excluded by the cage structures themselves (Taylor et al. 2010).

16.3.4 BENTHIC EFFECTS

This area is covered by the review of Forrest et al. (2007c) and summarised in MPI (2013), the reader is referred there for more detail.

16.3.4.1 INTRODUCTION

The benthic effects of aquaculture can be classified as:

- Organic enrichment and smothering, which can lead to (Forrest et al. 2007c):
  - localised biodeposition leading to enrichment of the seabed and associated microbial processes, and chemical and biological changes (including to infauna and epifauna, e.g., Christensen et al. 2003, Keeley et al. 2009);
  - in the case of intensive filter-feeder cultivation widespread biodeposition can potentially lead to a reduction in natural deposition rates;
  - smothering of benthic organisms and changes in sediment physical composition;
  - widespread biodeposition leading to mild enrichment in naturally depositional areas which has the potential for effects on reefs, inshore habitats and sensitive taxa;
  - sediment contamination (copper and zinc, covered in the additives section).
- Biofouling and drop-off of debris, which can lead to:
  - smothering and changes to physical composition of sediments (Keeley et al. 2009);
  - creation of habitat structure (Davidson & Brown 1999) and aggregations of predators and scavengers (Inglis & Gust 2003).
- Seabed shading by structures, which can change localised productivity under the farm (Huxham et al. 2006).

The magnitude and spatial extent of seabed effects from finfish farms are a function of a number of inter-related factors, which can be broadly considered as farm attributes and physical environment attributes.

Farm attributes that can affect the mass load of organic material deposited to the seabed include the following:

- fish stocking density and settling velocities of fish faeces (Magill et al. 2006);
- the type of feed and feeding systems, the feeding efficiency of the fish stock and the settling velocities of waste feed pellets;
- the type of cage structure can also influence depositional effects through differences in fish holding capacity, which affects feed loadings and may affect feeding efficiencies. Furthermore, cage design and position may affect the site's hydrodynamics; any reductions in flow will reduce waste dispersal and flushing, potentially resulting in depositional effects that are more localised but also more pronounced.

The capacity of the environment to disperse and assimilate farm wastes is a function of the attributes of the site (primarily water depth and current speeds), although assimilative capacity may also vary seasonally in relation to factors such as water temperature. Consequently, sites located in deep water (more than 30 m) and exposed to strong water currents (more than 15 cm s-1 on average) will have more widely dispersed depositional footprints with less intense enrichment than shallow, less well-flushed sites (e.g., Molina Dominguez et al. 2001, Pearson & Black 2001, Aguado-Gimenez & Garcia-Garcia 2004).

16.3.4.2 SIGNIFICANCE OF EFFECTS

In general, benthic effects from feed-added and filter-feeder aquaculture are similar as they are caused by debris and waste falling to the seafloor generally in close proximity to the farm. However the higher volume of waste and the uneaten food involved in feed-added farming and its more

particulate nature generally means that effects from feed-added aquaculture are greater than those seen from filter-feeder aquaculture, and can be seen further away (within 1 km for feed-added species as opposed to within 100 m for filter-feeders; Forrest et al. 2007c). In extreme cases this can lead to anoxia and outgassing of hydrogen sulphide and methane. At low-flow sites very little resuspension occurs and effects are largely constrained to the local environment (Forrest et al. 2007c). At high-flow sites, however, the majority of the biodeposits are resuspended, exported and eventually deposited in a very diffuse form in neighbouring low flow areas (e.g., in blind bays). If depositional inputs are sufficiently elevated then there is potential for effects in the form of increased far-field deposition. This may result in very mild, but potentially spatially extensive organic enrichment. The ecological effects of farming *Undaria* and sea cucumbers are likely to be less severe on the benthos then those from feed-added or filter-feeding species (Keeley et al. 2009).

Fish farm and mussel farm studies in New Zealand and overseas indicate timescales of recovery ranging from a few months in well-flushed areas where effects are minor, to a few years in poorly flushed areas where moderate/strong enrichment has occurred (references within MPI 2013).

16.3.4.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Management measures for mitigating benthic impacts for aquaculture are similar to those for mitigating pelagic impacts (Section 16.3.2.3). Site selection is important for the same reasons, to maximise the dispersive properties of the site, but should also try to avoid potentially sensitive/valuable benthic habitats (conservation areas, reefs etc.). The fine-scale positioning of the cages should optimise the dispersal of wastes and minimise impacts on potentially sensitive habitats. Depositional modelling should be used to predict benthic effects from a range of farming scenarios to inform decisions regarding optimum (sustainable) site-specific feed capacities. The application of Environmental Quality Standards (EQS) can be useful. For example, best management practices (BMP) have been developed for benthic impacts from salmon farms in the Marlborough Sounds. These provide consistent and clear requirements for the management and monitoring of existing farms. Staged development and a Modelling-Ongrowing-Monitoring (MOM) approaches are also potentially beneficial (MPI 2013).

16.3.5 SEABIRD INTERACTIONS

The reader is referred to MPI 2013 (and references therein) for more detail.

16.3.5.1 INTRODUCTION

In New Zealand, the generally perceived negative effects of both feed-added aquaculture and filter feeder aquaculture have centred on entanglement (resulting in birds drowning) and habitat exclusion and displacement from feeding grounds. The location of the farm within the range of seabirds and the conservation status (which is a measure of the risk of extinction) of these seabird species are the main factors that may lead to issues of sustainability and conservation concern. Of particular concern are the location of farms in relation to breeding and feeding sites and the operational procedures of regular farm activities (which can affect things like likelihood of entanglement).

Potential negative effects may include disturbance of breeding colonies and birds feeding, blockage of the digestive tract following ingestion of foreign objects, injury or death following collision with farm structures and the spread of pathogens or pest species. In contrast, a potential beneficial effect includes the provision of roost sites closer to foraging areas (Lalas 2001), saving energy and enabling more efficient foraging; this is most likely to benefit shags, gulls and terns (MPI 2013). Likewise, the attraction and aggregation of small fish around marine farm structures (Grange 2002) may provide enhanced feeding opportunities for piscivorous seabirds.

16.3.5.2 SIGNIFICANCE OF EFFECTS

Siting of a farm close to a seabird breeding colony is very likely to have an immediate adverse effect that will continue as long as the duration of the farm. However, there are no reports of seabird deaths as a result of entanglement in aquaculture facilities in New Zealand (Butler 2003, Lloyd 2003) as the use of top-nets over sea cages in New Zealand appears to effectively exclude seabirds (MPI 2013). The potential effects of habitat exclusion by feed-added farms in New Zealand are considered to be insignificant given the small area occupied in relation to the large total area of suitable habitat available for foraging seabirds (MPI 2013).

16.3.5.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

At present, potential risks are identified on a case-by-case basis. The most obvious is the choice of site for a farm to avoid disturbance to sensitive breeding colonies of seabirds. Good operating practices (for feed-added farms) such as enclosing predator nets above and below cages, controlling litter, minimising the use of lights at night, keeping nets taut and using mesh sizes less than 6 cm, all minimise the chances of negative seabird interactions. Given the current relatively small size of the aquaculture industry in New Zealand, the overlap of farming activities with the feeding areas of seabirds is unlikely to present significant issues (MPI 2013).

There are significant knowledge gaps concerning almost all seabird species in New Zealand. Detailed information on the time-specific distribution, abundance and critical habitats is lacking. Also missing is information on key prey species of seabirds, particularly those that may be affected by aquaculture. In addition, there should be ongoing monitoring (where an issue is identified) and research into the operation, design and maintenance of farm structures that minimise disturbance and entanglement risks. Little is known about the exclusion distance needed from different species of foraging and feeding seabirds, for example, proposed exclusion distances for king shags in the Marlborough Sounds range from 100 to 1000 m (Davidson et al. 1995, Taylor 2000), but more recently, Lalas (2001) noted that king shags resting ashore or on emergent objects only flew off when approached to within 30 m.

16.3.6 EFFECTS FROM ADDITIVES

Background data on the use and impact of chemicals locally are from research on salmon aquaculture and have been reviewed previously (Forrest et al. 2007c, 2011, Wilson et al. 2009, Burridge et al. 2010, Clement et al. 2010, MPI 2013), the reader is referred there for more detail.

16.3.6.1 INTRODUCTION

The main intentional use of additives is as antibiotics, antibacterials and other therapeutants (MPI 2013). The concern with therapeutants is their potential to affect non-target organisms (phyto- and zooplankton, sediment bacteria) and the rise of resistant bacteria and/or parasites (GESAMP 1997, Forrest et al. 2007c, 2011). The main unintentional additions are from zinc in fish feed and copper when used as an antifouling agent on structures (MPI 2013). The main concern with metals is their toxicity to animals (Forrest et al. 2007c, 2010, Clement et al. 2010).

16.3.6.2 SIGNIFICANCE OF EFFECTS

Currently, there is minimal use of chemicals such as antibiotics, antibacterials and other therapeutants intentionally added to the marine environment by the New Zealand aquaculture industry; however, culture of native species may lead to the emergence of diseases that may require new treatments.

Recent assessments at salmon farming sites in the Marlborough Sounds revealed locally elevated copper and zinc levels (with maxima exceeding ANZECC (2000) sediment quality guideline values between 2005 and 2010; Hopkins et al. 2006). Potential adverse effects from high zinc exposures range from interference with growth at low concentrations to behavioural abnormalities at high concentrations (Eisler 1993, Burridge et al. 2010); but elevated metal concentrations do not necessarily indicate adverse ecological effects as they may not be bioavailable (Forrest et al. 2007c).

16.3.6.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

All species cultured for human consumption from aquaculture have to meet strict food safety standards, which regulate the acceptable concentrations of metals, chemicals and additives in food products. New Zealand salmon farmers must also comply with the New Zealand Salmon Farmers Association's Finfish Aquaculture Environmental Code of Practice, with harvesting and processing in accordance with New Zealand food safety standards.

No chemical/additives are known to be used in the farming of bivalves and lower trophic level species. If these are used in the future 'best management practice', should minimise food wastage and the use of therapeutants, and hence help mitigate potential effects. The most important means to reduce and manage the overall antibiotic usage would be to support development of targeted disease management strategies and alternative therapies, in particular vaccines, which are not presently licensed for use, nor used, in New Zealand.

The potential for environmental issues from therapeutant use in the future will need to be assessed on a case-by-case basis. Use of therapeutants in New Zealand is low, but their persistence in the environment, the induction of resistance of targeted organisms and the effects on non-target organisms are the main knowledge gaps. Studies on the bioavailability and forms of the metals will give better understanding of their toxicity; a focus is needed on sub-lethal effects on individual species and the broader effects on benthic communities.

16.3.7 ESCAPEE EFFECTS

The subject of escapee effects from aquaculture is well covered for finfish by the reviews of Forrest et al. (2007c) for New Zealand and Jensen et al. (2010) for Norway, and for shellfish by Keeley et al. (2009) and summarised in MPI (2013). The reader is referred to these sources for more detail.

16.3.7.1 INTRODUCTION

It is useful to recognise that the human-mediated transfer of numerous marine organisms to New Zealand and around the coastline is an issue with a long history that continues today. Historically, this reflects deliberate transplants of marine organisms (including salmon), and more recently the inadvertent transfer of a range of native and non-indigenous marine species (including fish), especially via vessel movements (e.g., Hayward 1997, Cranfield et al. 1998). The alteration to marine ecosystems and transfer of fish diseases via these unmanaged mechanisms is well recognised (Ruiz et al. 2000, Hilliard 2004), and hence any incremental risk from finfish culture should be considered within this broader context.

The effects of escapees from aquaculture vary considerably in relation to the following factors (Forrest et al. 2007c):

- the numbers involved in the escape episode,
- the location of the farm in relation to wild populations and its size, distribution and health,
- whether the species is native (hāpuku, kingfish) or introduced (salmon),
- whether the brood stock is hatchery bred or wild sourced,
- the fish harvest size in relation to reproductive maturity and the ability of gametes to survive and develop in the wild,
- the ability of escapees to survive and reproduce in the wild, as determined by their ability to feed successfully and interbreed with wild stocks.

The main effects of escapees (Forrest et al. 2007c) for feed-added species are in terms of:

- competition for resources with wild fish and related ecosystem effects from escapee fish (e.g., through predation),
- alteration of the genetic structure of wild fish populations by escapee fish and potential loss of genetic integrity in the wild populations,
- transmission of pathogens from farmed stocks to wild fish populations.

The main factors controlling the number of fish escaping, and their subsequent effects are the integrity of the nets used to contain the fish and the amount of difference between the wild fish and farmed fish in terms of their genetics and their pests and diseases.

16.3.7.2 SIGNIFICANCE OF EFFECTS

The likelihood of escapee effects in New Zealand is low, based on the current small size of the industry, limited overlap of wild and farmed populations (in terms of salmon; Deans et al. 2004) and the broad home range (in terms of kingfish and hāpuku) and likelihood of high genetic diversity in these native species (Paul 2002, Forrest et al. 2007c). If escapee effects are seen on wild populations they are, however, likely to be irreversible and could potentially be at a national scale.

16.3.7.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Management strategies to minimise escapees are usually based upon maintaining net integrity. In Norway reporting of escapes, and estimation of numbers escaped is mandatory and therefore provides a baseline to improve upon (Jensen et al. 2010). In New Zealand escapee events are not reported to any central authority. At this time no knowledge is available on the potential effect that escaped farmed kingfish or hāpuku could have upon the wild populations.

16.3.8 EFFECTS ON WILD FISH

The reader is referred to MPI 2013 (and references therein) for more detail.

16.3.8.1 INTRODUCTION

A potential immediate effect on wild fish populations from the development of a finfish farm is the degradation or loss of habitat beneath or within close proximity to new farm structures (e.g., spatial overlap with species' critical spawning grounds and/or migration routes). By adding three-dimensional structures to the marine environment, finfish farms provide habitat for colonisation by fouling organisms and associated biota (Glasby 1999, Connell 2000, Dealteris et al. 2004). These newly colonised structures and the habitat they create tend to attract wild fish species seeking foraging habitat, detrital food sources and/or refuge from predators (e.g., Dealteris et al. 2004). Submerged artificial lighting at night is frequently used on finfish farms to control maturation and increase productivity (e.g., Porter et al. 1999). The lighting can enhance the attraction of wild fish to farm structures (Cornelisen & Quarterman 2010).

The main effects associated with the creation of artificial habitats, and attraction of wild fish species to aquaculture structures, include the following:

- enhanced predation on wild fish by higher trophic level predators (e.g., seals) and predation by cultured fish on wild fish trapped within cage structures,
- consumption of waste feed by wild fish (Felsing et al. 2004, Dempster et al. 2005),
- changes in recreational fishing patterns and pressure (N. Keeley, pers. comm.), which could affect wild fish populations differently than in the absence of the structures,
- larval fish depletion by filter-feeders (as observed by Davenport et al. 2000 and Lehane & Davenport 2002) and/or potential trophic interactions (e.g., alteration of plankton composition and food availability).

16.3.8.2 SIGNIFICANCE OF EFFECTS

In general, the effects of aquaculture on wild fish populations are likely to be small in comparison with the effects on other aspects of the marine ecosystem, such as effects on the seabed. The effects of farming hāpuku or kingfish on wild fish are expected to be generally similar to those from farming of king salmon already in New Zealand. Modelling of larval egg depletion (Broekhuizen et al. 2002) and other work suggest that while the feeding of fish in farms could have an impact on recruitment to fisheries; the scale of this effect will largely be governed by the extent of the culture, the behaviour and characteristics of larvae and the flow dynamics of the regions in question (MPI 2013).

The effects of farming filter-feeders are likely to be less than those of farming feed-added species (due to the lack of food added as an attractant), but shell-drop is likely to create a (lesser) attraction. The extent of impacts from the farming of *Undaria* and sea cucumbers is likely to have a lesser impact than feed-added or filter-feeding aquaculture, as they neither require feed nor exhibit shell drop (MPI 2013).

16.3.8.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Management options identified in MPI (2013) for minimising effects on wild fish include proper site selection, which requires assessment of potential impacts of farm developments on wild fish stocks. Assessments should identify proximity and impact to critical, sensitive or protected habitats and species, with particular reference to potential impacts on spawning grounds or juvenile habitats. Careful management of feed quality and feeding practices should minimise waste feed inputs to the surrounding environment and minimise effects on wild fish populations. The effects of finfish farms on wild fish populations in New Zealand are not well documented and knowledge gaps exist, particularly with regard to the effects of finfish farms on fish movements and various reproductive stages (e.g., larval settlement).

16.3.9 HYDRODYNAMIC EFFECTS

The reader is referred to MPI 2013 (and references therein) for more detail.

16.3.9.1 INTRODUCTION

Hydrodynamic conditions are an important determinant of the suitability of a site for aquaculture, as well as the spatial size and magnitude of the environmental effects. Here,

hydrodynamics refers to the physical attributes of the water including:

- currents,
- stratification,
- waves.

Current speed is a key factor determining the exchange of water through the cage, areas over which deposition occurs, where the dissolved material is transported and how it is dispersed and the resuspension of material. Stratification refers to the layering of water caused by differences in temperature and salinity. Stratification can play a strong role in oxygen depletion by restricting vertical transport of oxygen from the surface to deeper waters. Waves can break up stratification, play a key role in determining which species can inhabit an area, and resuspend material.

16.3.9.2 SIGNIFICANCE OF EFFECTS

Aquaculture operations can have a number of effects on hydrodynamics. The drag from cages can affect currents, causing wakes, turbulence and flow diversion (Helsley & Kim 2005, Venayagamoorthy et al. 2011). Low-velocity areas have a higher probability of issues of deposition, oxygen depletion and ammonium build-up. There are likely to be interactions between stratification and fish cages in the form of selective blocking, restricted underflow, generation of internal waves and vertical mixing (Plew et al. 2006). Fish swimming may also play a role in enhancing mixing and causing upwelling within cages (Chacon-Torres et al. 1988). Wave energy is attenuated by fish cages, and this will result in a shadow of reduced wave activity behind the farmed areas (Chan & Lee 2001, Lader et al. 2007).

While some physical effects may affect other physical processes directly, for example attenuation of wave energy affecting surf or coastal sediment transport; it is generally more important to consider how physical effects influence ecological processes. For example, the physical effect of reduced current speeds caused by drag from aquaculture structures (Helsley & Kim 2005, Venayagamoorthy et al. 2011) may result in an increase in the flushing time of a bay (Plew 2011). This in turn may lead to increased nutrient concentrations. Reductions in wave energy near the coast may change the mix of species inhabiting an area.

16.3.9.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

The physical hydrodynamic effects will interact strongly with pelagic and benthic processes. Selection of suitable indicators for physical changes should ideally be based on their relative importance in determining the habitat for ecological communities in an area. However, it is this link between the physical and ecological changes that is often the least understood area of hydrodynamic impacts.

16.3.10 CUMULATIVE IMPACTS

The following section draws heavily on previous reviews of the environmental effects of finfish (Forrest et al. 2007c) and non-finfish aquaculture (Keeley et al. 2009). Complementary information on the wider ecosystem effects of aquaculture in relation to the water column is provided in Section 16.3.2: Pelagic effects. The reader is referred to MPI 2013 (and references therein) for more detail.

16.3.10.1 INTRODUCTION

The previous sections (16.3.1–16.3.9) have focused on issue-specific ecological effects of aquaculture developments on the marine environment. Our understanding of these effects is largely based on farm-scale assessments and monitoring; the potential for wider-ecosystem effects (e.g., far-field benthic enrichment, effects on fish populations, migrating mammals, etc.) is acknowledged but is far less well understood. As aquaculture develops and the number of farms in coastal waters increases, wider-ecosystem issues become more important to consider due to the cumulative environmental effects that could arise from multiple farms combined with additional anthropogenic stressors affecting, and possibly interacting with natural marine processes (see Figure 16.4 for an example of multiple stressors interacting with natural processes).

Within the context of aquaculture development in the marine environment, cumulative effects are defined here as:

> *Ecological effects in the marine environment that result from the incremental, accumulating and interacting effects of an aquaculture development when added to other stressors from anthropogenic activities affecting the marine environment (past, present and future*

activities) and foreseeable changes in ocean conditions (i.e., in response to climate change).

A number of examples of potential cumulative impacts of aquaculture exist, three of these will be given here to illustrate the definition above:

- Drop-off of mussels, shells and biofouling organisms onto the seabed beneath mussel farms, can lead to the creation of reef-like habitat, and alter the composition and abundance of benthic organisms beneath farms (see Section 16.3.4). Where this occurs in high densities such as the ribbon-like developments in the Marlborough Sounds, this could lead to additive (cumulative) effects on the wider ecosystem due to alteration of a larger proportion of the benthos.
- In the case of farm structures, aquaculture involving numerous farms situated along the coast could also have cumulative effects on nearshore currents and waves, which in turn could affect important processes (e.g., larval transport, nutrient exchange) along the shoreline (see Section 16.3.9). As aquaculture development intensifies, there is likely to be an increase in man-made structures and boat traffic, increasing the risk of invasion and establishment of pests. Cumulative degradation of the marine environment from multiple stressors compromises habitat quality and could enhance biosecurity risks by increasing productivity and proliferation of pest species such as invasive macroalage (e.g., *Undaria*) and invertebrates (e.g., the bivalve *Theora lubrica* and tunicate *Styela clava*) that thrive on the benthos under conditions of high organic enrichment (Section 16.3.1 provides comprehensive information on methods for minimising biosecurity risk that are applicable to wider, regional scales).

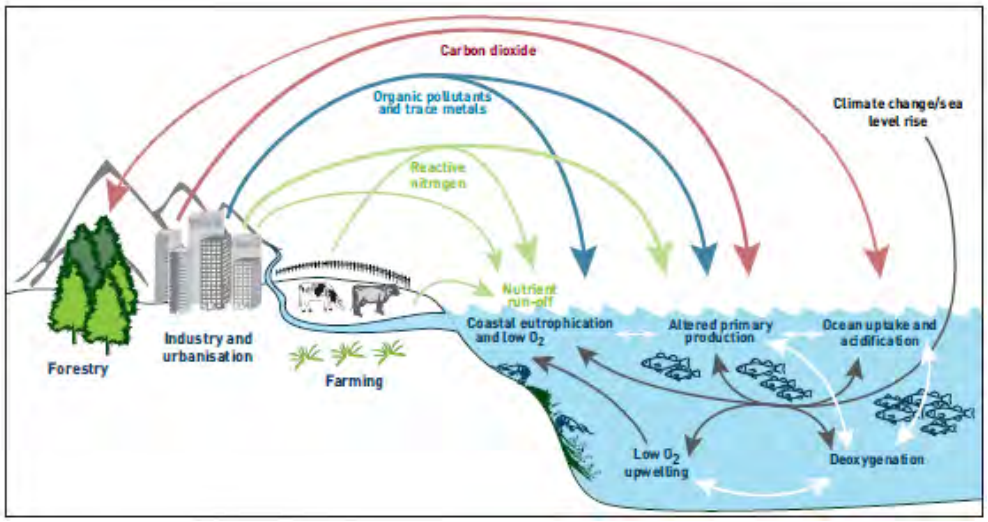


Figure 16.4: Conceptual diagram of anthropogenic influence in marine ecosystems.

Limited resources and uncertainty in understanding all of the potentially complex interactions between aquaculture, other stressors and the environment necessitates the need to focus on those aspects of aquaculture most likely to contribute to cumulative environmental change. Hence, increasing emphasis has been placed on assessing the contribution of aquaculture to cumulative changes in nutrient conditions and primary production, and in turn the knock-on effects on the wider ecosystem (see Hargrave et al. 2005, Volkman et al. 2009 and chapters therein). All forms of aquaculture addressed in this report contribute to these nutrient effects, whether through nutrient emissions to the water column and seabed, or the net extraction of plankton (filter-feeding bivalves) and nutrients (nutrient uptake by macroalgae) from the water column. The following sections focus on the potential far-field nutrient implications of aquaculture.

16.3.10.2 SIGNIFICANCE OF EFFECTS

The particular concern with the potential expansion of fish farms is the potential risk of eutrophication (SEPA 2000, Hargrave et al. 2005, Díaz et al. 2012). Eutrophication is the process where excessive nutrient inputs to a water body result in accelerated primary production (phytoplankton and macroalgae growth) and flow-on effects to the wider environment such as reduced water clarity, physical smothering of biota, or extreme reductions in dissolved oxygen because of microbial decay (Degobbis 1989, Cloern 2001, Paerl 2006). On a global scale, runoff from land-based agriculture has been identified as the primary driver of intense eutrophication of coastal environments, however, feed-added forms of aquaculture have been singled out as an important emerging contributor to nutrient enrichment (Díaz et al. 2012).

Nutrients of varying particulate and dissolved organic and inorganic forms are added to the environment as a result of feed-added aquaculture. Particulate organic nitrogen (PON) and phosphorus (POP) are primarily deposited onto the seabed as fish faeces but also as waste feed pellets and particles. Farmed fish also excrete dissolved inorganic nutrients such as ammonium (NH4). Smaller particles of feed in the water column (through the addition of feed and/or via resuspension) can be consumed by other organisms such as zooplankton and shellfish, which, through subsequent excretion, in turn contribute to the dissolved nutrient pool. The dissolved inorganic nutrients from feed-added aquaculture combined with other sources of nutrient inputs can fuel the growth of phytoplankton (Wu et al. 1994) and at high concentrations can cause harmful phytoplankton blooms (Sorokin et al. 1996). In New Zealand's temperate waters, nitrogen may be the nutrient limiting phytoplankton growth under certain conditions e.g., when concentrations are generally low and light is plentiful (MacKenzie 2004, Howarth & Marino 2006). Complicating matters is the fact that nutrients from finfish farms are only one source of nutrients in the marine environment, and, like other sources, their inputs vary over time, e.g., salmon farms in the Marlborough Sounds increase feed levels by about 50% during summer months, which is also the period of greatest light availability for primary production. Internationally there have been experiences of blooms of species that produce biotoxins, some of which can be directly toxic to fish, and others which can accumulate in shellfish and affect consumers. As far as is known to date salmon farming in New Zealand has not given rise to any harmful phytoplankton blooms and such effects are unlikely in the near future unless considerable new development occurs (Forrest et al. 2007c).

The risk of exceeding the assimilative capacity and accelerating eutrophication will be dictated by the physical characteristics of a region, such as retention time, water depth and ambient nutrient concentrations, combined with the intensity and types of existing and planned aquaculture and upstream land-based developments. There is compelling evidence that bivalve aquaculture can affect nutrient cycling and the quantity and quality of food (plankton) across a range of spatial scales from local to system-wide (Prins et al. 1998, Cerco & Noel 2007, Coen et al. 2007). In turn, the quantity and quality of food available to other consumers could be affected (Prins et al. 1998, Dupuy et al. 2000, Pietros & Rice 2003, Leguerrier et al. 2004), with consequences for local populations of higher trophic level organisms such as fish.

In some regions where numerous farms with high-density cultures occur, there is the potential risk of exceeding the region's capacity to sustain high shellfish production and the wider ecosystem itself. An example is Pelorus Sound, where questions around the concept of carrying capacity arose following observed decreases of about 25% in Greenshell mussel yields between 1999 and 2002 (Zeldis et al. 2008). These reductions were attributed to climatic forcing conditions and inter-annual variability in phytoplankton biomass over multi-year time scales (Zeldis et al. 2008). This suggests that this region is close to sustainable production limits during years of naturally low primary production.

16.3.10.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

The management of cumulative effects in the marine environment can be addressed using a two-tiered approach that not only considers the contribution of effects from individual developments, but also an overall regional assessment of wider environmental change in response to the many stressors impacting on the marine environment (e.g., Dubé 2003). Critical to regional assessments of cumulative effects in the marine environment is accessibility and coordination of datasets, including those derived from consent monitoring at individual farms, and long-term State of the Environment (SoE) monitoring programmes. Standardised monitoring requirements for

aquaculture is an important step in ensuring the usefulness of consent monitoring datasets within broader-scale assessments. The requirements for assessing and managing cumulative effects fall beyond the scope of a single consent applicant or industry and are best dealt with through regional councils (e.g., Dubé 2003, Hargrave et al. 2005, Zeldis 2008a, 2008b) or central government departments (Morrisey et al. 2009, Zeldis et al. 2011a, 2011b).

Two ongoing projects will help address monitoring requirements for aquaculture. An ongoing MPI Biodiversity project 'Marine Environmental Monitoring Programme' (ZBD2010-42) is seeking to address the following two objectives:

1. prepare an online inventory of repeated biological and abiotic marine observations/datasets in New Zealand,

2. review, evaluate fitness for purpose, and identify gaps in the utility and interoperability of these datasets for inclusion in a Marine Environmental Monitoring Programme (MEMP) from both science and policy perspectives.

Therefore any attempts to standardise monitoring datasets for aquaculture should try to learn from the experience or recommendations of this project. In addition the Aquaculture Planning Fund project 12/04 'Guidance for aquaculture monitoring in the Waikato Region' will develop an environmental monitoring framework to manage environmental change from aquaculture growth that will incorporate SOE monitoring, consent monitoring and predictive monitoring and have application to other regions.

Spatial modelling tools offer a way of estimating the extent to which the cumulative effects of aquaculture may be approaching ecological carrying capacity on 'bay-wide' and 'regional' scales. However, knowledge gaps are still evident in these models; particularly in the biological aspects (e.g., feeding behaviour and growth of the shellfish), which are still areas of active research (particularly within the Sustainable Aquaculture MBIE funded programme CO10X0904).

Some generalisations have been proposed in terms of carrying capacity, but these are not always in agreement. Using 'sustainability performance indicators', Gibbs (2007) suggests that the retention (flushing) time for a water body should not exceed 5% of the clearance time of farmed mussels in order to minimise cumulative effects on the wider ecosystem. Whilst recently proposed bivalve aquaculture standards suggest that if the clearance time for the farmed bivalves divided by the retention time of the water body is less than 1 and the area occupied by the farms is less than 10% of the total area of the water body then ecological impacts are likely to be acceptable (Bivalve Aquaculture Dialogue 2010).

ECOPATH modelling (Christensen et al. 2000) was applied to assess the potential of Tasman Bay for mussel aquaculture development. This indicated that significant ecosystem energy flow changes occurred at mussel biomass levels less than 20% of a mussel-dominated ecosystem, thus implying that ecological carrying capacity limits may be much lower than production carrying capacity limits (Jiang & Gibbs 2005). Typically modelling is therefore used to determine the ecological carrying capacity of each system. An ongoing MPI project 'Nitrogen levels and adverse marine ecological effects' (ENV2012-01) is seeking to determine to what extent knowledge from overseas about the adverse effects of nitrogen on the marine environment can be applied here.

In the case of cumulative effects related to eutrophication, there is currently a very limited scientific understanding of the transport, fate and ecological consequences of nutrient loading from different sources and, in turn, how they cumulatively affect marine ecosystems (Olsen et al. 2008). Managing cumulative effects to achieve sustainability ultimately requires regional approaches to managing developments and activities in a holistic, ecosystem-based management (EBM) framework which utilises spatial planning (Crain et al. 2008).

In the absence of over-arching EBM programmes and a robust scientific base for adaptive management in response to cumulative effects, a precautionary approach is warranted in future developments of feed-added aquaculture. Using a precautionary approach, development should be conducted in a staged manner based on conservative limits of expansion. Important tools and components of a precautionary approach include:

1. The use of models and existing data to gauge limits to development[5] within the context of a region's assimilation capacity (i.e., ecological carrying capacity).
2. Establishment of wider-ecosystem, long-term monitoring programmes that include establishment of baseline conditions of a region and adoption of limits of acceptable change.
3. Mitigation of effects through continual improvement of on-farm practices, potentially including improved feed technologies and the use of Integrated Multitrophic Aquaculture (IMTA, Figure 16.5). IMTA combines farming of different species to potentially ameliorate environmental effects.
4. Targeted monitoring and research for validating and improving accuracy of predictive models and understanding the role of feed-added aquaculture in driving cumulative effects.

In New Zealand the Limits of Acceptable Change (LAC) adaptive framework has been applied in the 3000 ha Wilson Bay Aquaculture Management Area (AMA), in the eastern Firth of Thames.[6] This involved stakeholders agreeing both to levels of acceptable change in indicators, and to management responses to apply if monitoring showed that these changes have been exceeded. An overseas example of the precautionary approach is the M-O-M system (Modelling–Ongrowing fish farms–Monitoring), which has been undertaken in Norway to provide information for adaptive management of salmon farming (Ervik et al. 1997, Hansen et al. 2001).

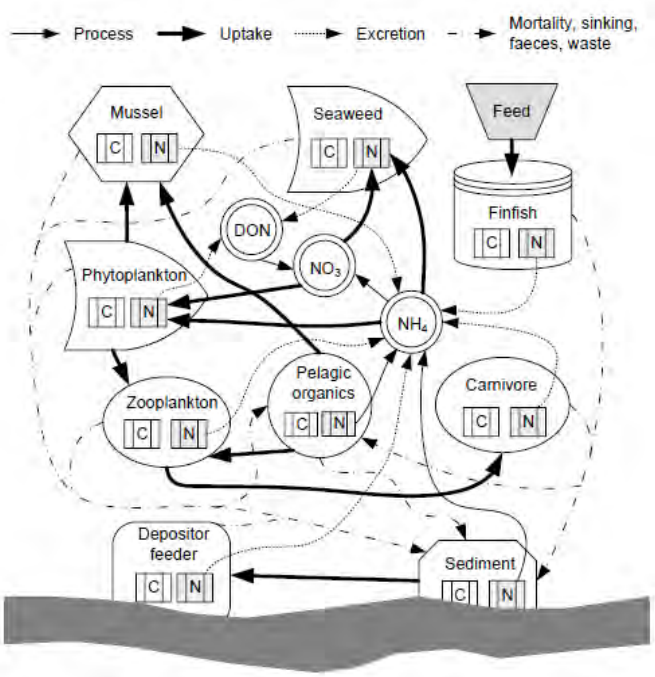


Figure 16.5: Conceptual diagram of IMTA model in terms of carbon (C) and nitrogen (N) biomass (from Ren, pers. comm.).

A precautionary approach necessitates establishment of conservative thresholds or limits to minimise risks and the extent of cumulative effects. Minimising risk of eutrophication by setting a limit (or cap) on nutrient loads in a coastal receiving environment would be similar to the approach taken in restoring the Rotorua Lakes. Nutrient mass-balance models can provide guidance on nutrient loading rates in a region under various scenarios, and on gauging proximity to conservative critical nutrient loading rates or CNLRs (Olsen et al. 2008). The mass-balance approach has facilitated the development of system-wide nutrient budgets and estimates of carrying capacity for feed-added aquaculture in Golden and Tasman Bays (Zeldis 2008b, Zeldis et al. 2011a, 2011b) and the Firth of Thames (Zeldis 2008a, Zeldis et al. 2010).

Internationally, there is a very limited understanding of the cumulative effects of multiple stressors on marine ecosystems in the long-term. A critical requirement for

[5] In some cases, areas may not be suitable for any development of aquaculture.

[6] NIWA (2006) Limits of acceptable change: a framework for managing marine farming. Retrieved from http://www.niwa.co.nz/publications/wa/vol14-no2-june-2006/limits-of-acceptable-change-a-framework-for-managing-marine-farming.

understanding these effects is having good information on existing environmental conditions, and continued monitoring to provide long time series datasets from which to validate models and quantify and forecast changes occurring in the wider environment.

Modelling has an important role to play in understanding, predicting and managing cumulative effects and New Zealand has access to extensive modelling capability; yet in most cases the uncertainty in model accuracy remains high due to insufficient field data for their calibration and validation. For example, underlying hydrodynamic models require sufficient time-series data on currents and water column stratification, while more advanced biogeochemical models require validated estimates of inputs (e.g., surface water, groundwater, marine) and losses (denitrification, burial rates) of nutrients specific to New Zealand's coastal waters.

16.4 REFERENCES

Aguado-Gimenez, F; Garcia-Garcia, B (2004) Assessment of some chemical parameters in marine sediments exposed to offshore cage fish farming influence: a pilot study. *Aquaculture 242*: 283–296.

Aguilar-Manjarrez, J; Kapetsky, J M; Soto, D (2010) The potential of spatial planning tools to support the ecosystem approach to aquaculture. Expert Workshop, 19–21 November 2008, Rome, Italy. *FAO Fisheries and Aquaculture Proceedings No. 17*. FAO, Rome. 176 p.

ANZECC (2000) Australian and New Zealand Guidelines for Fresh and Marine Water Quality 2000. Volume 2. Aquatic Ecosystems - Rationale and Background Information.

Aquaculture New Zealand (2012) New Zealand Seafood Exports: Calendar Year to December 2012 – Report 10B Seafood Exports Country by Species. 251 p.

Askew, C G (1972) The growth of the oysters *Ostrea edulis* and *Crossostrea gigas* in Emsworth Harbour. *Aquaculture 1 (2)*: 237–259.

Biosecurity Council (2003) Tiakina Aotearoa Protect New Zealand – The Biosecurity Strategy for New Zealand. 67 p. Retrieved from http://www.biosecurity.govt.nz/files/biosec/sys/strategy/biosecurity-strategy.pdf.

Bivalve Aquaculture Dialogue (2010) Bivalve Aquaculture Dialogue Standards. 50 p.

Broekhuizen, N; Zeldis, J; Stephens, S; Oldman, J; Ross, A; Ren, J; James, M (2002) Factors related to the sustainability of shellfish aquaculture operations in the Firth of Thames: A preliminary analysis. *NIWA Client Report EVW02243*, *Environment Waikato Technical Report 02/09* and *Auckland Regional Council Technical Publication TP 182*, prepared for Environment Waikato and Auckland Regional Council.

British Columbia Shellfish Growers Association (2001) Environmental management system code of practice. Retrieved from http://bcsga.netfirms.com/wp-content/uploads/2007/08/enviro-mgmt-code-of-practice_02feb7.pdf.

Burridge, L; Weis, J S; Cabello, F; Pizarro, J; Bostick, K (2010) Chemical use in salmon aquaculture: a review of current practices and possible environmental effects. *Aquaculture 306*: 7–23.

Butler, D (2003) Possible impacts of marine farming of mussels (*Perna canaliculus*) on king shag (*Leucocarbo carunculatus*). *DOC Science Internal Series 111*. Department of Conservation, Wellington, New Zealand.

Carver, C E; Chisholm, A; Mallet, A L (2003) Strategies to mitigate the impact of *Ciona intestinalis* (L.) biofouling on shellfish production. *Journal of Shellfish Research 22*: 621–631.

Cerco, C F; Noel, M R (2007) Can oyster restoration reverse cultural eutrophication in Chesapeake Bay? *Estuaries and Coasts 30*: 331–343.

Chacon-Torres, A; Ross, L G; Beveridge, M C M (1988) The effects of fish behaviour on dye dispersion and water exchange in small net cages. *Aquaculture 73*: 283–293.

Chan, A T; Lee, S W C (2001) Wave characteristics past a flexible fishnet. *Ocean Engineering 28*: 1517–1529.

Christensen, P B; Glud, R N; Dalsgaard, T; Gillespie, P (2003) Impacts of long-line mussel farming on oxygen and nitrogen dynamics and biological communities of coastal sediments. *Aquaculture 218*: 567–588.

Christensen, V; Walters, C J; Pauly, D (2000) Ecopath with EcoSim: a User's Guide, October 2000 Edition. Fisheries Centre, University of British Columbia. Vancouver, Canada.

Cigarria, J; Fernandez, J; Magadan, L P (1998) Feasibility of biological control of algal fouling in intertidal oyster culture using periwinkles. *Journal of Shellfish Research 17 (4)*: 1167–1169.

Clement, D; Keeley, N; Sneddon, R (2010) Ecological Relevance of Copper (Cu) and Zinc (Zn) in Sediments beneath Fish Farms in New Zealand. *Report No. 1805*, prepared for Marlborough District Council. 48 p. Retrieved from http://www.envirolink.govt.nz/PageFiles/584/877-MLDC48%20Ecological%20relevance%20of%20Cu%20and%20Zn%20in%20sediments%20beneath%20fish%20farms%20in%20NZ.pdf.

Cloern, J E (2001) Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series 210*: 223–253.

Coen, L D; Brumbaugh, R D; Busek, D; Grizzle, R; Luckenback, M W; Posey, M H; Powers, S P; Tolley, S G (2007) Ecosystem services related to oyster restoration. *Marine Ecology Progress Series 341*: 303–307.

Connell, S D (2000) Floating pontoons create novel habitats for subtidal epibiota. *Journal of Experimental Marine Biology and Ecology 247*: 183–194.

Cornelisen, C; Quarterman, A (2010) Effects of artificial lighting on the marine environment at the Clay Point and Te Pangu Bay salmon farms. *Cawthron Report No. 1851*, prepared for New Zealand King Salmon Company Limited. Cawthron Institute, Nelson, New Zealand.

Coutts, A D M; Forrest, B M (2005) Evaluation of eradication tools for the clubbed tunicate *Styela clava*. *Cawthron Report No. 1110*. 48 p.

Crain, C M; Kroeker, K; Halpern, B S (2008) Interactive and cumulative effects of multiple human stressors in marine systems. *Ecology Letters 11*: 1304–1315.

Cranfield, H J; Gordon, D P; Willan, R C; Marshall, B A; Battershill, C N; Francis, M P; Nelson, W A; Glasby, C J; Read, G B (1998) Adventive marine species in New Zealand. *NIWA Technical Report 34*.

Dann, P; Norman, F I; Cullen, J M; Neira, F J; Chiaradia, A (2000) Mortality and breeding failure of little penguins, *Eudyptual minor*, in Victoria, 1995–96, following a widespread mortality of pilchard, *Sarinops sagax*. *Marine Freshwater Research 51*: 355–362.

Davenport, J; Smith, R W; Packer, M (2000) Mussels *Mytilus edulis*: Significant consumers and destroyers of mesozooplankton. *Marine Ecology Progress Series 198*: 131–137.

Davidson, R J; Brown, D A (1999) Ecological report of potential marine farm areas located offshore of the west coast of Coromandel Peninsula. *Survey and Monitoring Report No. 177*, prepared by Davidson Environmental Ltd for Environment Waikato. 39 p.

Davidson, R J; Courtney, S P; Millar, I R; Brown, D A; Deans, N A; Clerke, P R; Dix, J C; Lawless, P F; Mavor, S J; McRae, S M (1995) Ecologically important marine, freshwater, island and mainland areas from Cape Soucis to the Ure River, Marlborough, New Zealand: Recommendations for protection. *Occasional Publication 16*. Nelson/Marlborough Conservancy, Department of Conservation, Nelson, New Zealand.

Dealteris, J T; Kilpatrick, B D; Rheault, R B (2004) A comparative evaluation of the habitat value of shellfish aquaculture gear, submerged aquatic vegetation and a non-vegetated seabed. *Journal of Shellfish Research 23*: 867–874.

Deans, N; Unwin, M; Rodway, M (2004) Sports fishery management, Chapter 41. *In*: Harding, J; Mosley, P; Pearson, C; Sorrell, B (Eds.), Freshwaters of New Zealand. NZ Hydrological Society.

Degobbis, D (1989) Increased eutrophication of the northern Adriatic Sea: second act. *Marine Pollution Bulletin 20*: 452–457.

Dempster, T; Fernandez-Jover, D; Sanchez-Jerez, P; Tuya, F; Bayle-Sempere, J; Boyra, A; Haroun, R (2005) Vertical variability of wild fish assemblages around sea-cage fish farms: Implications for management. *Marine Ecology Progress Series 304*: 15–29.

Díaz, R; Rabalais, N N; Breitburg, D L (2012) Agriculture's Impact on Aquaculture: Hypoxia and Eutrophication in Marine Waters. Report under Directorate for Trade and Agriculture for the OECD (Organisation for Economic Co-operation and Development). 45 p.

Dinamani, P (1974) Pacific oyster may pose threat to rock oyster. *Catch 74*: 5–9.

Dubé, M (2003) Cumulative effect assessment in Canada: a regional framework for aquatic ecosystems. *Environmental Impact Assessment Review 23*: 723–245.

Duprey, N M T (2007) Dusky dolphin (*Lagenorhynchus obscurus*) behaviour and human interactions: Implications for tourism and aquaculture. MSc Thesis, Texas A&M University Department of Wildlife and Fisheries Science, Texas, USA.

Dupuy, C; Vaquer, A; Lam-Höai, T; Rougier, C; Mazouni, N; Lautier, J; Collos, Y; Gall, S L (2000) Feeding rate of the oyster *Crassostrea gigas* in a natural planktonic community of the Mediterranean Thau lagoon. *Marine Ecology Progress Series 205*: 171–184.

Eisler, R (1993) Zinc hazards to fish, wildlife and invertebrates: a synoptic review. US Department of the Interior Fish and Wildlife Service. *Biological Report No. 10*. Contaminant Hazard Reviews 26. 126 p.

Enright, C T; Elner, R W; Griswold, A; Borgese, E M (1993) Evaluation of crabs as control agents for biofouling in suspended culture of European oysters. *World Aquaculture 24*: 49–51.

Enright, C; Krailo, D; Staples, L; Smith, M; Vaughan, C; Ward, D; Gaul, P; Borgese, E (1983) Biological control of fouling algae in oyster aquaculture. *Journal of Shellfish Research 3*: 41–44.

Ervik, A; Hansen, P K; Aure, J; Stigebrandt, A; Johannessen, P; Jahnsen, T (1997) Regulating the local environmental impact of intensive marine fish farming I. the concept of the MOM system (modelling – ongrowing fish farms – monitoring). *Aquaculture 158*: 85–94.

FAO (2012) The State of World Fisheries and Aquaculture 2012. Fisheries and Aquaculture Department, Rome. 230 p.

Felsing, M; Glencross, B; Telfer, T (2004) Preliminary study on the effects of exclusion of wild fauna from aquaculture cages in a shallow marine environment. *Aquaculture 243*: 159–174.

Forrest, B M; Blakemore, K (2002) Inter-regional marine farming pathways for the Asian kelp *Undaria pinnatifida*. *Cawthron Report No. 726*. 27 p. plus appendices.

Forrest, B; Dunmore, R; Keeley, N (2010) Seabed impacts of the Te Pangu Bay salmon farm: Annual monitoring 2009. *Cawthron Report No. 1733*, prepared for New Zealand King Salmon Company Limited. 32 p.

Forrest, B; Elmetri, I; Clark, K (2007a) Review of the ecological effects of intertidal oyster aquaculture. *Cawthron Report No. 1275*, prepared for Northland Regional Council. 25 p. Retrieved from http://www.nrc.govt.nz/upload/1742/Oyster%20Effects_Final%20 %28web%29.pdf.

Forrest, B M; Hopkins, G A; Dodgshun, T J; Gardner, J P A (2007b) Efficacy of acetic acid treatments in the management of marine biofouling. *Aquaculture 262*: 319–332.

Forrest, B; Hopkins, G; Webb, S; Tremblay, L (2011) Overview of marine biosecurity risks from finfish aquaculture development in the Waikato region. *Cawthron Report No. 1871*, prepared for Waikato Regional Council. 78 p.

Forrest, B; Keeley, N; Gillespie, P; Hopkins, G; Knight, B; Govier, D (2007c) Review of the Ecological Effects of Marine Finfish Aquaculture: Final Report. Cawthron Report for the Ministry of Fisheries. 80 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Forrest, B; Keeley, N; Hopkins, G; Webb, S; Clement, D (2009) Bivalve aquaculture in estuaries: Review and synthesis of oyster cultivation effects. *Aquaculture 298*: 1–15.

Forrest, B M; Taylor, M D (2002) Assessing Invasion Impact: Survey Design Considerations and Implications for Management of An Invasive Marine Plant. *Biological Invasions 4*: 375–386.

GESAMP (1997) Towards safe and effective use of chemicals in coastal aquaculture. IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection, Reports and Studies No. 65. 40 p.

Gibbs, M T (2007) Sustainability performance indicators for suspended bivalve aquaculture activities. *Ecological indicators 7 (1)*: 94–107.

Glasby, T M (1999) Differences between subtidal epibiota on pier pilings and rocky reefs at marinas in Sydney, Australia. *Estuarine, Coastal and Shelf Science 48*: 281–290.

Grange, K R (2002) The effects of mussel farms on benthic habitats and fisheries resources within and outside marine farms, Pelorus Sound. *NIWA Client Report NEL2002-003*, prepared for Mussel Industry Council.

Hall, S; Delaporte, A; Phillips, M; Beveridge, M; O'Keefe, M (2011) Blue Frontiers: Managing the Environmental Costs of Aquaculture. The World Fish Center, Penang, Malaysia. 103 p.

Handley, S J (2002) Optimizing intertidal Pacific oyster (Thunberg) culture, Houhora Harbour, northern New Zealand. *Aquaculture Research 33*: 1019–1030.

Handley, S J; Bergquist, P R (1997) Spionid polychaete infestations of intertidal pacific oysters *Crassostrea gigas* (Thunberg), Mahurangi Harbour, northern New Zealand. *Aquaculture 153*: 191–205.

Handley, S; Jeffs, A (2002) Assessment for Future Expansion of Pacific Oyster Farming in Northland. *NIWA Client Report AL2003-027*, prepared for Enterprise Northland, Aquaculture Development Group. 36 p.

Hansen, P K; Ervik, A; Schaanning, M; Johannessen, P; Aure, J; Jahnsen, T; Stigebrandt, A (2001) Regulating the local environmental impact of intensive, marine fish farming: II. The monitoring programme of the MOM system (Modelling-Ongrowing fish farms-Monitoring). *Aquaculture 194 (1–2)*: 75–92.

Hargrave, B T; Silvert, W; Keizer, P D (2005) Assessing and managing environmental risks associated with marine finfish aquaculture. *Environmental Effects of Marine Finfish Aquaculture 5*: 433–461.

Hayward, B (1997) Introduced marine organisms in New Zealand and their impact in the Waitemata Harbour, Auckland. *Tane 36*: 197–223.

Helsley, C E; Kim, J W (2005) Mixing downstream of a submerged fish cage: a numerical study. *IEEE Journal of Oceanic Engineering 30*: 12–19.

Hidu, H; Conary, C; Chapman, S R (1981) Suspended culture of oysters: Biological fouling control. *Aquaculture 22*: 189–192.

Hilliard, R (2004) Best practice for the management of introduced marine pests: a review. GISP: the Global Invasive Species Program, GISP Secretariat. 173 p.

Hopkins, G A; Butcher, R; Clarke, M (2006) Fisheries Resource Impact Assessments (FRIAs) for three proposed marine farm extensions in East Bay Queen Charlotte Sound. *Cawthron Report No. 1125*. 51 p.

Howarth, R W; Marino, R (2006) Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: Evolving views over three decades. *Limnology and Oceanography 51 (1)*: 364–376.

Huxham, M; Gilpin, L; Mocogni, M; Harper, S (2006) Microalgae, macrofauna and sediment stability: an experimental test of a reciprocal relationship. *Marine Ecology Progress Series 310*: 55–63.

Inglis, G T; Gust, N (2003) Potential Indirect Effects of Shellfish Culture on the Reproductive Success of Benthic Predators. *Journal of Applied Ecology 40*: 1077–1089.

Jensen, Ø; Dempster, T; Thorstad, E; Uglem, I; Fredheim, A (2010) Escapes of fish from Norwegian sea-cage aquaculture: causes, consequences and methods to prevent escape. *Aquaculture and Environmental Interactions 1*: 71–83.

Jiang, W M; Gibbs, M T (2005) Predicting the carrying capacity of bivalve shellfish culture using a steady, linear food web model. *Aquaculture 244*: 171–185.

Keeley, N; Forrest, B; Hopkins, G; Gillespie, P; Knight, B; Webb, S; Clement, D; Gardener, J (2009) Sustainable aquaculture in New Zealand: Review of the Ecological Effects of farming shellfish

and other Non-finfish species. *Cawthron Report No. 1476*, prepared for the Ministry of Fisheries. 150 p. plus appendices. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Kemper, C M; Pemberton, D; Cawthorn, M; Heinrich, S; Mann, J; Würsig, B; Shaughnessy, P; Gales, R (2003) Aquaculture and marine mammals: Co-existence or conflict? *In*: Gales, N; Hindell, M; Kirkwood, R (Eds.), Marine mammals: Fisheries, tourism and management issues. pp 208–224. CSIRO Publishing, Collingwood, Victoria, Australia.

Kospartov, K; Inglis, G; Seaward, K; Brink, A; D'Archino, R; Ahyong, S (2010) Non-indigenous and cryptogenic marine species in New Zealand. Current state of knowledge. Interim report prepared for MAFBNZ Post Border Directorate. MAF Biosecurity New Zealand, Wellington, New Zealand.

Lader, P F; Olsen, A; Jensen, A; Sveen, J K; Fredheim, A; Enerhaug, B (2007) Experimental investigation of the interaction between waves and net structures--Damping mechanism. *Aquacultural Engineering 37*: 100–114.

Lalas, C (2001) Evidence presented for Kuku Mara Partnership, Forsyth Bay. Environment Court Hearing, Blenheim. Statements of Evidence Volume 1.

La Rosa, T; Mirto, S; Favaloro, E; Savona, B; Sarà, G; Danovaro, R; Mazzola, A (2002) Impact on the water column biogeochemistry of a Mediterranean mussel and fish farm. *Water Research 36*: 713–721.

Leguerrier, D; Niquil, N; Petiau, A; Bodoy, A (2004) Modeling the impact of oyster culture on a mudflat food web in Marennes-Oléron Bay (France). *Marine Ecology Progress Series 273*: 147–162.

Lehane, C; Davenport, J (2002) Ingestion of mesozooplankton by three species of bivalve: *Mytilus edulis*, *Cerastoderma edule* and *Aequipecten opercularis*. *Journal of the Marine Biological Association of the United Kingdom 82*: 3999/1–6.

Lloyd, B D (2003) Potential effects of mussel farming on New Zealand's marine mammals and seabirds: A discussion paper. Department of Conservation, Wellington, New Zealand.

Locke, A; Doe, K G; Fairchild, W L; Jackman, P M; Reese, E J (2009a) Preliminary evaluation of effects of invasive tunicate management with acetic acid and calcium hydroxide on non-target marine organisms in Prince Edward Island, Canada. *Aquatic Invasions 4 (1)*: 221–236.

Locke, A; Hanson, J M; MacNair, N G; Smith, A H (2009b) Rapid response to non-indigenous species. 2. Case studies of non-indigenous tunicates in Prince Edward Island. *Aquatic Invasions 4*: 249–258.

MacDiarmid, A; McKenzie, A; Sturman, J; Beaumont, J; Mikaloff-Fletcher, S; Dunne, J (2012) Assessment of anthropogenic threats to New Zealand marine habitats. *New Zealand Aquatic Environment and Biodiversity Report No. 93*. 255 p.

MacKenzie, L (2004) River inputs, re-mineralisation and the spatial and temporal distribution of inorganic nutrients in Tasman Bay, New Zealand. *New Zealand Journal of Marine and Freshwater Research 38*: 681–704.

Magill, S H; Thetmeyer, H; Cromey, C J (2006) Settling velocity of faecal pellets of gilthead sea bream (*Sparus aurata* L.) and sea bass (*Dicentrarchus labrax* L.) and sensitivity analysis using measured data in a deposition model. *Aquaculture 251*: 295–305.

Markowitz, T M; Harlin, A D; Würsig, B; Mcfadden, C J (2004) Dusky dolphin foraging habitat: Overlap with aquaculture in New Zealand. *Aquatic Conservation: Marine and Freshwater Ecosystems 14*: 133–149.

Ministry for Primary Industries (2013) Literature Review of Ecological Effects of Aquaculture. Ministry for Primary Industries, Cawthron Institute and National Institute for Water and Atmospheric Research Ltd. Ministry for Primary Industries, Wellington, New Zealand. 260 p.

Molina Dominguez, L; Lopez Calero, G; Vergara-Martin, J M; Robaina, L (2001) A comparative study of sediments under a marine cage farm at Gran Canaria Island (Spain): Preliminary results. *Aquaculture 192*: 225–231.

Molnar, J L; Gamboa, R L; Revenga, C; Spalding, M D (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment 6 (9)*: 485–492.

Morrisey, D; Page, M; Handley, S; Middleton, C; Schick, R (2009) Biology and ecology of the introduced ascidian *Eudistoma elongatum*, and trials of potential methods for its control. MAF Biosecurity New Zealand Technical Paper.

Mulcahy, K; Peart, R (2012) Wonders of the Sea: The protection of New Zealand's marine mammals. 334 p.

NRC (2010) Ecosystem Concepts for Sustainable Bivalve Mariculture. By Committee on Best Practices for Shellfish Mariculture and the Effects of Commercial Activities in Drakes Estero, Pt. Reyes National Seashore, California. 191 p.

NZMIC (2001) New Zealand Mussel Industry Council Ltd. code of practice for transfer of mussel seed. Aquaculture New Zealand. 3 p.

NZOIA (2007) New Zealand Oyster Industry Code of Practice. Aquaculture New Zealand. 51 p.

NZSFA (2007) New Zealand Salmon Farmers Association Inc. Finfish Aquaculture Code of Practice. Aquaculture New Zealand. 27 p.

Okamura, B; Feist, S W (2011) Emerging diseases in freshwater systems. *Freshwater Ecosystem 56*: 627–637.

Olsen, L M; Holmer, M; Olsen, Y (2008) Perspectives of nutrient emission from fish aquaculture in coastal waters: Literature review with evaluated state of knowledge. Final report for The Fishery and Aquaculture Industry Research Fund. 87 p.

Paerl, H W (2006) Assessing and managing nutrient-enhanced eutrophication in estuarine and coastal waters: Interactive effects of human and climatic perturbations. *Ecological Engineering 26 (1)*: 40–54.

Paul, L (2002) Can existing data describe the stock structure of the two New Zealand groper species hapuku (*Poliprion oxygeneios*) and bass (*P. ainericanus*)? *New Zealand Fisheries Assessment Report 2002/14*. 24 p.

Pearson, H; Srivivasan, M; Vaughn, R; Würsig, B (2007) Cetacean abundance, habitat use, and behaviour in winter and spring 2006, Admiralty Bay, New Zealand, with comparisons to 2005. Report to Marlborough District Council and Department of Conservation.

Pearson, T H; Black, K D (2001) The environmental impact of marine fish cage culture. *In*: Black, K D (Ed.), Environmental impacts of aquaculture. pp 1–31. Academic Press, Sheffield.

Peeler, E J; Taylor, N G (2011) The application of epidemiology in aquatic animal health – opportunities and challenges. *Veterinary Research 42 (1)*: 94.

Pietros, J M; Rice, M A (2003) The impacts of aquacultured oysters, *Crassostrea virginica* (Gmelin, 1791) on water column nitrogen and sedimentation: results of as mesocosm study. *Aquaculture 220*: 407–422.

Plew, D R (2011) Shellfish farm-induced changes to tidal circulation in an embayment, and implications for seston depletion. *Aquaculture Environment Interactions 1*: 201–214.

Plew, D R; Spigel, R H; Stevens, C L; Nokes, R I; Davidson, M J (2006) Stratified flow interactions with a suspended canopy. *Environmental Fluid Mechanics 6*: 519–539.

Porter, M J R; Duncan, N J; Mitchell, D; Bromagea, N R (1999) The use of cage lighting to reduce plasma melatonin in Atlantic salmon (*Salmo salar*) and its effects on the inhibition of grilsing. *Aquaculture 176 (3–4)*: 237–244.

Prins, T C; Smaal, A C; Dame, R F (1998) A review of the feedbacks between bivalve grazing and ecosystem processes. *Aquatic Ecology 31*: 349–359.

Quale, D B (1969) Pacific oyster culture in British Columbia. *Fisheries Research Board of Canada Bulletin 169*.

Reid, G (2007) Nutrient releases from Salmon Aquaculture, Chapter 1. *In*: Buschmann, A; Costa-Pierce, A; Cross, S; Iriarte, L; Olsen, Y; Reid, G (committee members). Nutrient impacts of farmed Atlantic salmon (*Salmo salar*) on pelagic ecosystems and implications for carrying capacity. pp 7–23. Report of the Technical Working Group (TWG) on nutrients and carrying capacity of the salmon aquaculture dialogue.

Ruiz, G M; Rawlings, T K; Dobbs, F C; Drake, L A; Mullady, T; Huq, A; Colwell, R R (2000) Global spread of microorganisms by ships. *Nature 408*: 49–50.

Saaty, T (1987) Risk, its priority and probability: the analytic hierarchy process. *Risk Analysis 7 (2)*: 159–172.

SAD (Salmon Aquaculture Dialogue) (2011) Second draft standards for responsible salmon aquaculture: Revised draft standards for public comment, 16 May 2011. Retrieved from www.worldwildlife.org/salmondialogue.

Scavia, D; Field, J; Boesch, D; Buddemeier, R; Burkett, V; Cayan, D; Fogarty, M; Harwell, M; Howarth, R; Mason, C; Reed, D; Royer, T; Sallenger, A; Titus, J (2002) Climate change impacts on U.S. Coastal and Marine Ecosystems. *Estuaries and Coasts 25 (2)*: 149–164.

SEPA (2000) Policy on regulation and expansion of caged fish farming of salmon in Scotland. Policy No 40, Scottish Environmental Protection Agency.

Silva, C; Ferreira, J G; Bricker, S B; DelValls, T A; Martín-Díaz, M L; Yáñez, E (2011) Site selection for shellfish aquaculture by means of GIS and farm-scale models, with an emphasis on data-poor environments. *Aquaculture 318 (3–4)*: 444–457.

Slooten, E; Dawson, S M; DuFresne, S (2001) Report on interactions between Hector's dolphins (*Cephalorynchus hectori*) and a Golden Bay mussel farm. Unpublished report for Environment Canterbury.

Sorokin, Y I; Sorokin, P Y; Ravagnan, G (1996) On an extremely dense bloom of the dinoflagellate *Alexandrium tamarense* in lagoons of the Po river delta: impact on the environment. *Journal Sea Research 35*: 251–255.

Soto, D; Aguilar-Manjarrez, J; Brugère, C; Angel, D; Bailey, C; Black, K; Edwards, P; Costa-Pierce, B; Chopin, T; Deudero, S; Freeman, S; Hambrey, J; Hishamunda, N; Knowler, D; Silvert, W; Marba, N; Mathe, S; Norambuena, R; Simard, F; Tett, P; Troell, M; Wainberg, A (2008) Applying an ecosystem based approach to aquaculture: principles, scales and some management measures. *In*: Soto, D; Aguilar-Manjarrez, J; Hishamunda, N (Eds.), Building an Ecosystem Approach to Aquaculture. pp 15–35. FAO/Universitat de les Illes Balears Expert Workshop, 7–11 May 2007, Palma de Mallorca, Spain. *FAO Fisheries and Aquaculture Proceedings No. 14*. FAO, Rome.

Stoklosa, R; Ford, R; Pawson, M; Nielsen, M (2012) Phase Two Report of the MAF Aquaculture Ecological Guidance Project-Risk-based ecological assessment of New Zealand aquaculture. Workshop Report, 21–22 February 2012, Nelson. Prepared for the Aquaculture Unit of the New Zealand Ministry of Agriculture and Forestry (E-Systems Pty Limited, Hobart Tasmania, Australia). 95 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Taylor, D I; Keeley, N B; Heasman, K; Clement, D; Knight, B (2010) AEE for offshore finfish farming within the four Kahungunu blocks of the Napier AMA. Cawthron Client Report. Cawthron Institute, Nelson, New Zealand.

Taylor, G A (2000) Action plan for seabird conservation in New Zealand. Part A: Threatened seabirds. *Threatened Species Occasional*

Publication 16. Department of Conservation, Wellington, New Zealand.

Vaughn, R; Würsig, B (2006) Dusky dolphin distribution, behaviour and predator associations in spring 2005, Admiralty Bay, New Zealand. A report to Marlborough District Council and New Zealand Department of Conservation.

Venayagamoorthy, S K; Ku, H; Fringer, O B; Chiu, A; Naylor, R L; Koseff, J R (2011) Numerical modeling of aquaculture dissolved waste transport in a coastal embayment. *Environmental Fluid Mechanics 11 (4)*: 329–352. doi:10.1007/s10652-011-9209-0.

Volkman, J; Thompson, P; Herzfeld, M; Wild-Allen, K; Blackburn, S; MacLeod, C; Swadling, K M; Foster, S; Bonham, P; Holdsworth, D (2009) A whole-of-ecosystem assessment of environmental issues for salmonid aquaculture. Aquafin CRC Final Report, for CRC Project 4.2 (2)/FRDC Project 2004/074.

Wilson, A; Magill, S; Black, K D (2009) Review of environmental impact assessment and monitoring in salmon aquaculture. Environmental impact assessment and monitoring in aquaculture. *FAO Fisheries and Aquaculture Technical Paper 527*: 455–535.

Wright, A (2008) What's all the noise about? A call for lowering underwater noise from ships. *Marine Technology Reporter 51 (5)*: 22–25.

Wu, R S S; Lam, K S; MacKay, D W; Lau, T C; Yam, V (1994) Impact of marine fish farming on water quality and bottom sediment: a case study in the sub-tropical environment. *Marine Environmental Research 38*: 115–145.

Würsig, B; Gailey, G A (2002) Marine mammals and aquaculture: Conflicts and potential resolutions. *In*: Stickney, R R; McVay, J P (Eds.), Responsible marine aquaculture. pp 45–59. CAP International Press, New York, USA.

WWF (2010) Bivalve Aquaculture Dialogue Standards Environmental and Social Standards for Bivalve Aquaculture. 23 p. Retrieved from http://www.worldwildlife.org/what/globalmarkets/aquaculture/WWFBinaryitem13945.pdf.

Zeldis, J (2008a) Wilson Bay marine farm monitoring bi-annual report for 2006 and 2007. *NIWA Client Report CHC2008-086*, for NIWA Project WBC08501, prepared for Wilson Bay Group A Consortium.

Zeldis, J R (2008b) Origin and processing of nutrients in Golden and Tasman Bays. *Envirolink Advice Grant and Client Report CHC2008-052*, prepared for Tasman District Council.

Zeldis, J; Broekhuizen, N; Forsythe, A; Morrisey D, Stenton- Dozey, J M E (2010) Waikato Marine Finfish Farming: Production and Ecological Guidance. *NIWA Client Report CHC2010-147*, for NIWA Project PRM 201016.

Zeldis, J; Broekhuizen, N; Forsythe, A; Morrissey, D; Stenton-Dozey, J (2011a) Waikato marine finfish farming: Production and ecological guidance. *NIWA Client Report CHC2010-147*. 112 p.

Zeldis, J; Hadfield, M; Broekhuizen, N; Morrisey, D; Stenton-Dozey, J (2011b) Tasman Aquaculture – Guidance on farming additive species (Stage 1). *NIWA Report CHC 2011-005*, prepared for Ministry of Fisheries Aquaculture Unit. 72 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Zeldis, J R; Howard-Williams, C; Carter, C M; Schiel, D R (2008) ENSO and riverine control of nutrient loading, phytoplankton biomass and mussel aquaculture yield in Pelorus Sound, New Zealand. *Marine Ecology Progress Series 371*: 131–142.

Zeldis, J; Robinson, K; Ross, A; Hayden, B (2004) First observations of predation by New Zealand Greenshell mussels *Perna canaliculus* on zooplankton. *Journal of Experimental Marine Biology and Ecology 311 (2)*: 287–299.

17 ANTARCTIC SCIENCE

Status of chapter	This chapter has been fully updated for AEBAR 2018.
Scope of chapter	This chapter outlines the ecosystem structure in the Ross Sea sector of the Southern Ocean (Ross Sea and Amundsen Sea), toothfish fishery in this region, nature of ecosystem-fishing interactions, the management approach, trends in key indicators of fishing effects, major sources of uncertainty and research priorities.
Area	Ross Sea region; Amundsen Sea region (CCAMLR areas 88.1 and 88.2).
Focal localities	Areas with significant fisheries interactions include Ross Sea shelf and slope, Amundsen Sea seamounts and shelf, areas of the Pacific-Antarctic Ridge.
Key issues	Fisheries in the region are managed by CCAMLR according to the principles of conservation given in Article 2 of the Convention for the Conservation of Antarctic Marine Living Resources. Effects of fishing are considered in the following categories: (1) bycatch species, (2) prey of target species, (3) predators of target species, (4) ecosystem at the system-level, (5) benthic habitat.
Emerging issues	Role of Marine Protected Areas in the Ross Sea region.
FNZ research (current)	ANT2018.
NZ government research (current)	MBIE: C01X1226 *Ross Sea Climate and Ecosystem*. MBIE: END18301 *Ross Sea Region Research and Monitoring Programme*. NIWA Core Funding, Coasts & Oceans Programme 4: 'Structure and function of marine ecosystems' Antarctica New Zealand: Antarctic SIF
Related chapters/issues	Benthic (seabed) impacts; Trophic and ecosystem-level effects; Biodiversity. Fisheries New Zealand 2018 Fisheries Plenary Report May 2018 Chapter 99: Toothfish.

17.1 CONTEXT

17.1.1 THIS CHAPTER

This chapter discusses the ecosystem effects of fishing for toothfish (principally Antarctic toothfish) in the Ross Sea region (150°E to 150°W) and the Amundsen Sea region (150°W to 105°W) (Figure 17.1).There is currently no krill fishing in the Ross Sea and Amundsen Sea regions.

The focus is on the ecosystem effects of fishing rather than the management of the toothfish stock itself. The stock assessment for Antarctic toothfish in the Ross Sea region is updated every two years (most recently in 2017, Mormede 2017) and a summary is available as part of the May Plenary Report (Fisheries New Zealand 2018). Research towards a stock assessment for Antarctic toothfish in the Amundsen Sea region is in progress (Large et al. 2016).

Section 17.1 presents a brief history of Southern Ocean fisheries, the present management framework for toothfish, and overviews of the life history, fishery and management of toothfish fisheries in the Ross Sea and Amundsen Sea regions, including the Ross Sea MPA.

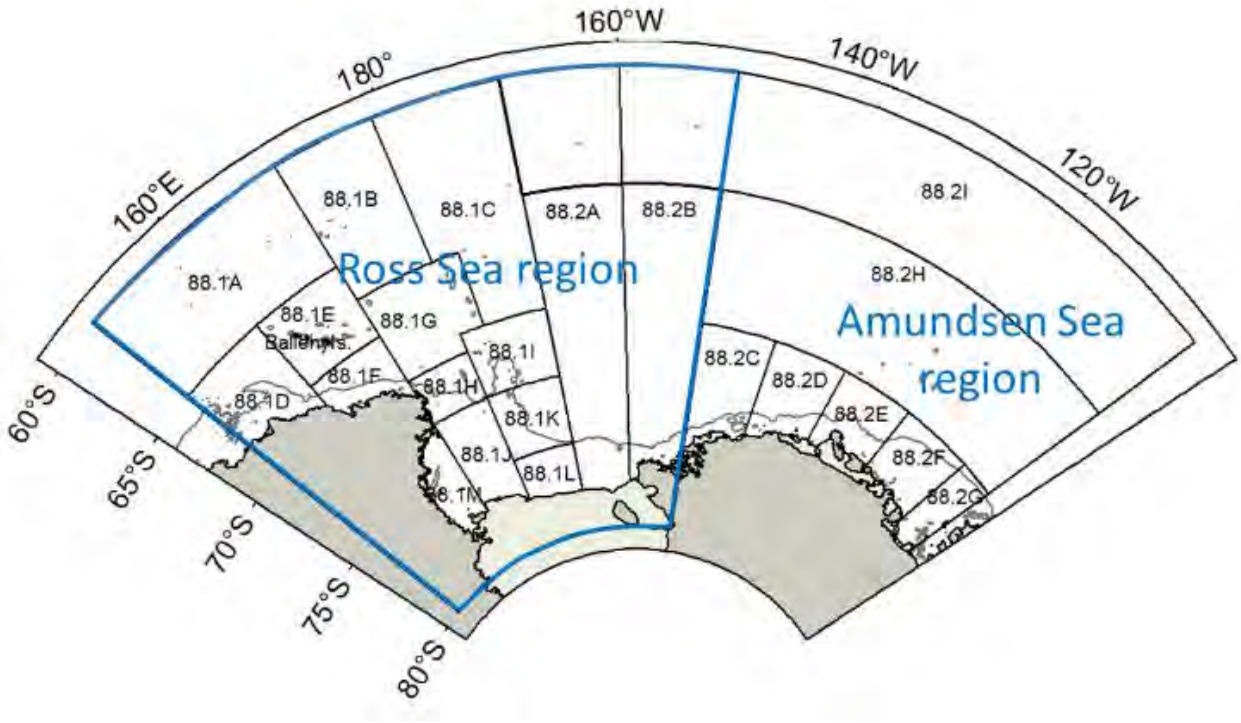


Figure 17.1: The Ross Sea and Amundsen Sea regions, which span CCAMLR (Commission for the Conservation of Antarctic Marine Living Resources) Subareas 88.1 and 88.2. CCAMLR small scale research units (SSRUs) and the depth contour at 1000 m are shown. The Ross Sea region is made up of 88.1 and 88.2A, B. The Amundsen Sea region includes SSRUs 88.2C–I. The Ross Sea *slope* region (with depths of approximately 1000–3000 m) is mainly contained in SSRUs 88.1H, 88.1I and 88.1K. Areas that are shallower than about 1000 m are called *shelf* regions (comprising 88.1J, 88.1L, 88.1M for the *Ross Sea shelf*).

Section 17.2 gives a characterisation of the Ross Sea ecosystem; no characterisation is yet available for the Amundsen Sea ecosystem.

Section 17.3 presents information on the major ecosystem effects of fishing in five categories:

- Effects of fishing on bycatch species,
- Effects of fishing on prey species,
- Effects of fishing on predator species,
- Trophic and system-level effects,
- Effects of fishing on habitats.

Section 17.4 summarises information on indicators and trends for the ecosystem effects of fishing in the Ross Sea and Amundsen Sea regions in the same five categories.

17.1.2 SOUTHERN OCEAN FISHERIES

A brief history of fisheries in the Southern Ocean[1] is given by CCAMLR (Commission for the Conservation of Antarctic Marine Living Resources)[2] and this section summarises that text. Seal harvesting in the Southern Ocean began in 1790. By 1825, some populations of fur seal were hunted close to extinction, and sealers began hunting elephant seals and some species of penguins for their oil. Sealing continued on a small scale into the 20th century, but there has been no commercial sealing in Antarctica since the 1950s.

Whaling in the Southern Ocean area began in 1904 and all seven species of whales found in the region were extensively exploited. A moratorium on commercial whaling was introduced in 1987. Whale sanctuaries were established in the Indian Ocean in 1979 and Southern Ocean in 1994. Management of whales is today the responsibility of the International Whaling Commission

[1] The Southern Ocean extends from the coast of the Antarctic continent northwards to the Antarctic Polar Front and represents approximately 15% of the world's ocean area.

[2] CCAMLR, History: The southern ocean. Retrieved from https://www.ccamlr.org/en/organisation/history.

(IWC). There are indications that some species of whale are recovering, but the low abundance of some of the largest species has made total numbers difficult to estimate.

Large-scale fishing for finfish in the Southern Ocean began in the late 1960s. Overall trends in fishery catches have varied widely, reflecting intense fishing during the 1960s and 1970s prior to the establishment of CCAMLR. Such fishing led to the overexploitation of some finfish species in the mid-1970s and 1980s. This overfishing along with interest in large-scale exploitation of Antarctic krill, raised concerns about the sustainability of Southern Oceans fisheries.

In the 1980s and 1990s, fishing in the Southern Ocean focused on krill (*Euphausia superba*), Patagonian toothfish (*Dissostichus eleginoides*), mackerel icefish (*Champsocephalus gunnari*) and, to a limited extent, squid and crab. Since the 1990s there has been growing interest in fisheries targeting Antarctic toothfish (*Dissostichus mawsoni*) adjacent to the Antarctic continent.

At the Eighth Antarctic Treaty Consultative Meeting in 1975, the Parties adopted the recommendation that noted the need to promote protection, scientific study and rational use of Antarctic marine living resources. This led to a Conference on the Conservation of Antarctic Marine Living Resources (CAMLR), which resulted in the CAMLR Convention.

New Zealand was a founding member of the CAMLR Convention which entered into force in 1982. The area of jurisdiction of the CAMLR Convention is approximately south of the circumpolar Antarctic Polar Front (Antarctic Convergence) in the Southern Ocean (Figure 17.2). The position of the Antarctic Polar Front varies seasonally and geographically, but is generally located near 50°S in the Atlantic and Indian sectors of the Southern Ocean and 60°S in the Pacific sector.

17.1.3 CCAMLR'S MANAGEMENT

The aim of the CAMLR Convention is to conserve the marine life of the Southern Ocean while allowing rational use of marine resources, including commercial fishing (CAMLR Convention 1980). The CAMLR Convention was the first international fishing agreement to explicitly require that management considers the effects of fishing on dependent and associated species as well as on the target

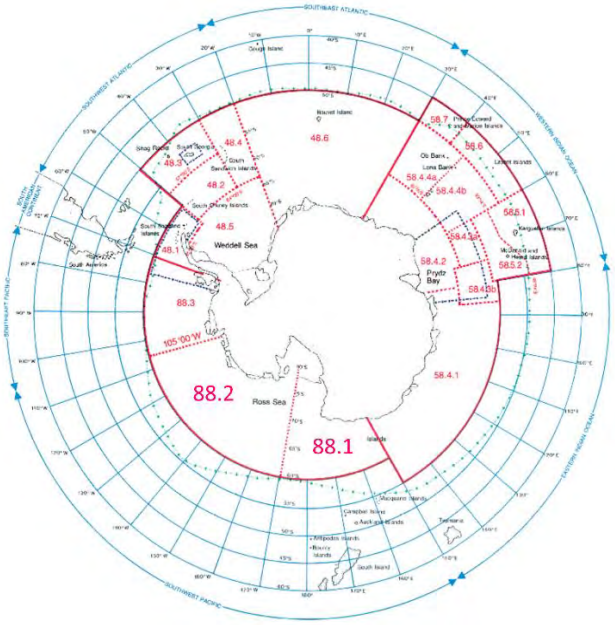


Figure 17.2: Boundary (solid pink line) of area managed according to the Convention for the Conservation of Antarctic Marine Living Resources. Dashed pink lines delineate CCAMLR statistical subareas.

species. The principles of conservation are given in Article II of the CAMLR Convention, and allow fishing in the CCAMLR Area subject to:

(a) maintenance of the size of harvested population at levels which ensure stable recruitment;
(b) maintenance of ecological relationships between harvested, dependent and related populations;
(c) prevention or minimisation of the risk of changes in the marine ecosystem that are not reversible in 20–30 years.

The regulatory framework for CCAMLR-managed fisheries recognises five types of fisheries: (1) new fishery; (2) exploratory fishery; (3) established fishery; (4) lapsed fishery; and (5) closed fishery.

Both the Ross Sea region and Amundsen Sea region toothfish fisheries are managed as '*exploratory fisheries'* by CCAMLR. Exploratory fisheries are not allowed to expand faster than the acquisition of information necessary for managing the fishery within CCAMLR's management objectives. In addition, notification and permission are required each year prior to fishing (CM 21-02). Finally, a fishery remains an exploratory fishery until sufficient information is available on appropriate catch and effort levels and the potential impacts on dependent and related species.

Decisions in CCAMLR are made by consensus among member states. At present members of the Commission include 24 States[3] and the European Union (acting as a single member). A further 11 countries[4] have acceded to the Convention. CCAMLR's Secretariat facilitates the implementation of the CAMLR Convention. Measures to manage Southern Ocean fisheries are implemented by means of a series of Conservation Measures (CMs), which are published annually[5] following decisions by the CCAMLR members at the annual Commission meetings in October. Scientific information, analyses and discussion to inform management are brought together annually by the CCAMLR Scientific Committee, which in turn is informed by several working groups, including: (1) Working Group on Ecosystem Monitoring and Management (WG-EMM); (2) Working Group on Fish Stock Assessment (WG-FSA); (3) Working Group on Statistics, Assessments and Modelling (WG-SAM); (4) Working Group on Incidental Mortality Associated with Fishing (WG-IMAF); (5) Subgroup on Acoustics, Survey and Analysis Methods (SG-ASAM).

The Scientific Committee on Antarctic Research (SCAR) has also advised CCAMLR in respect of key scientific areas for research. Work to develop greater collaboration between CCAMLR and IWC has been underway since 2013, especially with regard to managing the trophic impact of fishing for krill on baleen whales and other krill predators. Of particular relevance is the IWC-Southern Ocean Research Partnership (SORP).[6]

17.1.4 ANTARCTIC AND PATAGONIAN TOOTHFISH

Antarctic toothfish (*Dissostichus mawsoni*, Norman 1937) is endemic to the Southern Ocean, with a circumpolar distribution. The species is found in higher latitudes south of the Antarctic Convergence (Gon & Heemstra 1990). Patagonian toothfish (*Dissostichus eleginoides*, Smitt 1898), often marketed as '*Chilean sea bass*', shares many similarities with Antarctic toothfish but has a more northern distribution being rarely found in latitudes south of the Antarctic Convergence at about 65°S (Figure 17.3). A species profile, covering aspects of the biology, fisheries and stock assessment of both toothfish species was completed by Hanchet (2010) and Hanchet et al. (2015a).



Figure 17.3: Distributions of Antarctic and Patagonian toothfish in the Southern Ocean. Approximate location of Antarctic Convergence shown by the dotted line.[7]

For Antarctic toothfish in the Ross Sea region, spawning dynamics and early life history is described in Hanchet et al. (2008).

The toothfish stocks in the Amundsen Sea region are managed separately by CCAMLR from those in the Ross Sea region. Parker et al. (2014) reviewed the information from genetic studies, otolith microchemistry, stable isotopes, tagging, size and age structure, growth dynamics, and egg and larval dispersal simulations. The study concluded that it is likely that juveniles (less than 80 cm total length) from the two stocks mix in the shelf region, but that there is very limited mixing of adults between the Ross Sea and Amundsen Sea regions. The Amundsen Sea stock probably includes juveniles and adults along the continental margin of the Amundsen and Bellingshausen Seas, and a spawning region in the seamount complex of SSRU 88.2H (Amundsen Sea). Further information is needed to improve knowledge

[3] Argentina, Australia, Belgium, Brazil, Chile, China, European Union, France, Germany, India, Italy, Japan, Republic of Korea, Namibia, New Zealand, Norway, Poland, Russian Federation, South Africa, Spain, Sweden, Ukraine, United Kingdom, United States of America, Uruguay.

[4] Bulgaria, Canada, Cook Islands, Finland, Greece, Mauritius, Netherlands, Islamic Republic of Pakistan, Republic of Panama, Peru, Vanuatu.

[5] CCAMLR. Publications. Retrieved from https://www.ccamlr.org/en/publications/publications.

[6] International Whaling Commission. The Southern Ocean Research Partnership (IWC-SORP). Retrieved from https://iwc.int/sorp.

[7] NIWA. Antarctic Toothfish Fishery in the Ross Sea. Retrieved from https://www.niwa.co.nz/fisheries/research-projects/the-ross-sea-trophic-model/toothfish-fishery.

of the toothfish stock structure in the Amundsen Sea region (Delegations of New Zealand, Norway and the United Kingdom 2014).

For Antarctic toothfish in the Ross Sea and Amundsen Sea regions, spawning is thought to take place to the north of the Antarctic continental slope, during winter (Hanchet et al. 2008). The first winter longline survey of Antarctic toothfish conducted during June and July 2016 in the northern Ross Sea region confirmed toothfish spawning in this region (Stevens et al. 2016).

More information on the life history and stock structure can be found in the Fisheries Assessment Plenary (Fisheries New Zealand 2018).

The stock structure of Patagonian toothfish in the Ross Sea and Amundsen Sea regions is less well known. Patagonian toothfish in the Ross Sea region are believed to come from a stock which is widely distributed beyond Macquarie and Campbell Plateau and into the high seas.

17.1.5 ROSS SEA REGION TOOTHFISH FISHERY

A characterisation of the fishery in the Ross Sea region is given in Mormede & Parker (2018a). Fishing for toothfish began in the Ross Sea region in 1997. The Ross Sea region is the major fishing area for Antarctic toothfish in the Southern Ocean (Hanchet et al. 2008). Most of the catch in the Ross Sea region (over 99%) is Antarctic toothfish (an average of 2860 t/y since 2005), while catches of Patagonian toothfish taken mainly from the north-west of the Ross Sea region have averaged only about 6 t/y since 2005.

The toothfish fishery in the Ross Sea region saw a steady expansion of effort (number of sets) from 1998 to 2001, and an almost three-fold increase in 2004, which led to the increases in catches shown in Figure 17.4. Since 2005 effort has been more stable. All fishing for toothfish in the Ross Sea and Amundsen Sea regions uses baited longlines. In earlier years most vessels fished with the autoline system, but these have been joined by vessels fishing with Spanish lines and more recently trotlines.

The average Illegal, Unreported and Unregulated (IUU) catch of toothfish in Subarea 88.1 was estimated to be 240 tonnes in 2004, 28 tonnes in 2005 and 272 tonnes in 2008. Following the recognition of methodological issues regarding the estimation of IUU catch levels since 2011,

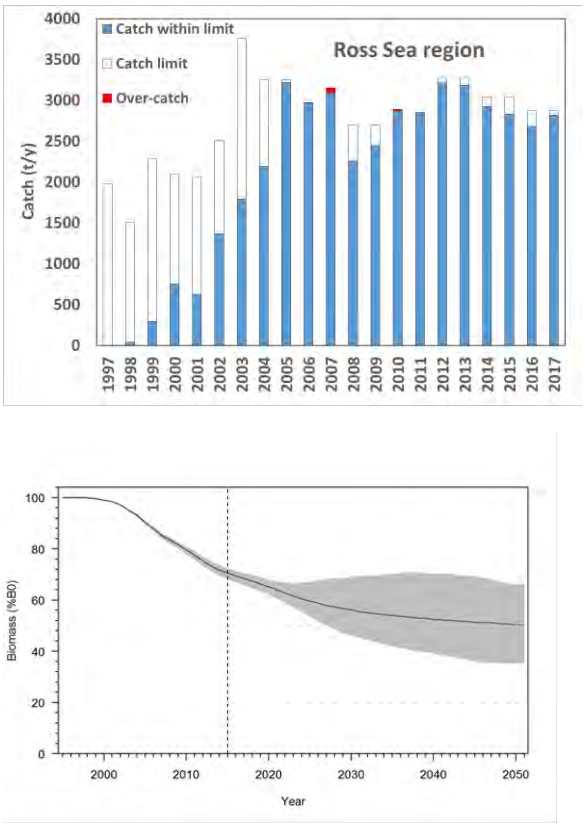


Figure 17.4: [a] Catch and catch-limit, [b] assessed spawning stock biomass (with 5th–95th percentiles in grey) for Antarctic toothfish in the Ross Sea region (CCAMLR subareas 88.1 and 88.2A, B). Antarctic fishing years are labelled as the later year of the season (e.g., the 1997–98 fishing season is labelled '1998'). [CCAMLR 2017a; Mormede et al. 2015, Mormede & Parker 2018a].

evidence of IUU presence or activity has continued to be recorded but no corresponding estimates of the IUU catch for Dissostichus spp. have been provided. One IUU-listed fishing vessel was observed in Subarea 88.1 during 2008 and an unknown vessel sighting was reported in 2012 (CCAMLR 2017a).

Annual research surveys of sub-adult (70–110 cm) toothfish have been carried out in the southern Ross Sea since 2011 to provide an estimate of any changes in recruitment (e.g., Stevens et al. 2018).

Spatial information on fishing in the 88.1/88.2 is often described using CCAMLR Small-Scale Research Units (SSRUs; Figure 17.1). Although most SSRUs have been fished over time, the proportion of effort in each SSRU has varied considerably each year and ice conditions. Two of the three slope SSRUs (88.1H and 88.1I) have been the most consistently fished SSRUs (Figure 17.5). In years with ice conditions favourable to fishing the fishery also extends into 88.1K.

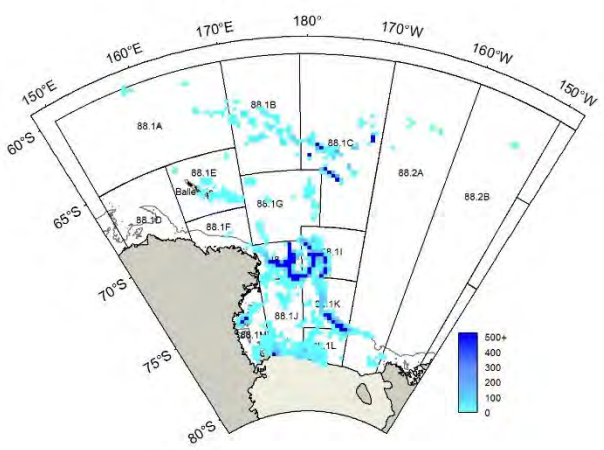


Figure 17.5: Spatial distribution of total toothfish catch in the Ross Sea region (t) from 1997 to 2015.

The length of the fishing season in the Ross Sea fishery has changed over time. In the first few years the fishery was mainly carried out from January to March, and between 2001 and 2003 extended into April and May. Since 2006, fishing starts on 1st December (ice permitting) and is usually finished by early February (Mormede & Parker 2018a).

The Ross Sea region toothfish fishery was first certified by the Marine Stewardship Council (MSC) in November 2010 and was recertified in 2015. The MSC Fisheries Standard is designed to assess whether a fishery is well managed and sustainable. There are three core principles that every fishery must meet:

1) Sustainable fish stock.
2) Minimising environmental impact.
3) Effective management.

17.1.6 AMUNDSEN SEA REGION FISHERY

The Amundsen Sea toothfish fishery is designated as an exploratory fishery by CCAMLR and a characterisation of the fishery in this region is given in Mormede & Parker (2018b). The toothfish fishery in the Amundsen Sea region has been operating since 2003 (Figure 17.6), with an annual catch of 106–624 t since 2006 (Mormede & Parker 2018b).

Up to 2014 the main fishery in this area has operated in the northern SSRU (Large et al. 2016). In contrast, up to and including 2014, the catch and effort in the southern SSRUs have been low (Figure 17.7). Parker (2014) showed high local exploitation rates and indications of localised depletion on some individual seamounts in the north.

Only five Patagonian toothfish have been caught in the Amundsen Sea region since 2004. More data is required before a robust stock assessment for the Amundsen Sea region can be developed and this work is active at the time of writing (Mormede et al. 2016).

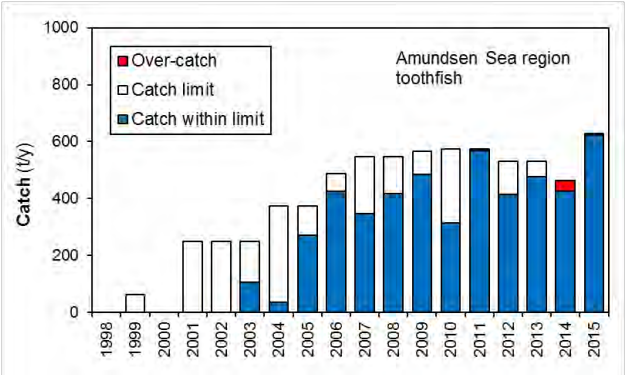


Figure 17.6: Catch and catch-limit for Antarctic toothfish in the Amundsen Sea region (CCAMLR subareas 88.2 C-I; CCAMLR 2017b).

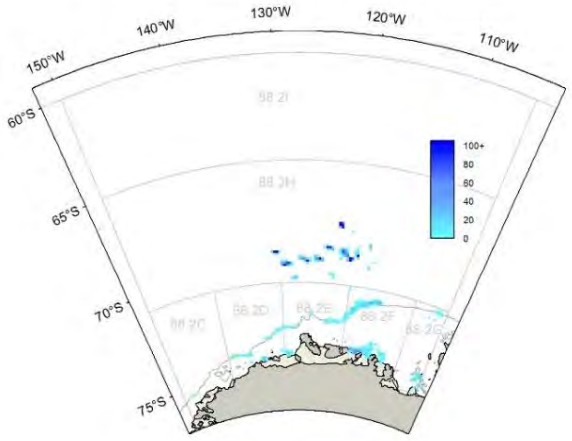


Figure 17.7: Spatial distribution of total toothfish catch in the Amundsen Sea region (t) from 1997 to 2015.

17.2 ROSS SEA ECOSYSTEM

17.2.1 OVERVIEW

Although annual primary productivity is still low on a global scale, seasonally, the shelf waters of the Ross Sea are amongst the most biologically productive areas of the Southern Ocean (Arrigo & van Dijken 2004). Irradiance, iron, and macronutrients (nitrate, silicate) variously limit algal growth through the year (Smith et al. 2003). The high latitude position of the Ross Sea means that primary production is highly seasonal, driven by the annual light/dark cycle and the freezing and thawing of the sea surface. Insolation sets the dominant limit on primary production from autumn through to spring; in summer, macronutrients are not depleted and iron appears to limit primary production (e.g., Sedwick et al. 2000, Arrigo et al.

2003). Organisms have various strategies for survival through the winter, including storage of lipids or other high-energy products, winter quiescence, vertical migration, adoption of a wide range of feeding styles, and adaptation of breeding cycles, including migrating in and out of the region (some whales, seals, birds) (Battaglia et al. 1997).

Sea ice plays a key structural role in influencing the ecology of the Ross Sea (Thomas & Dieckmann 2002, Arrigo & Thomas 2004). The mean monthly sea ice cover in the Ross Sea varies from 5% ice-free in winter to 70% ice-free in January (Arrigo & van Dijken 2004), with ice reaching a maximum thickness around November of about 2 m. The Ross Sea polynya[8] is the major structural oceanographic feature of the Ross Sea (Jacobs & Comiso 1989). The dynamics of phytoplankton in the open water of the Ross Sea polynya are very different to those in the marginal ice zone around the polynya. Although ice extent in the Ross Sea region is increasing (Comiso 2003), sea ice in the Ross Sea itself has been decreasing and getting thinner as the Ross Sea polynya has become larger and more persistent (Parkinson 2002).

The upper surface of the ice provides a habitat for a number of sea birds and mammals (Ackley et al. 2003). At the same time, the ice itself, especially the underpart, which is in contact with the water, constitutes a unique habitat for microalgae and bacteria. This provides a food source for associated microfauna and meiofauna and the cryopelagic fauna of the surface water layer immediate below the ice (Garrison 1991, Brierley & Thomas 2002, Arrigo & Thomas 2004). Present estimates suggest that the contribution of epontic[9] algae to total primary production in the Ross Sea is a few percent (Arrigo et al. 1997, Pinkerton et al. 2010a).

The flow of energy from primary production in the water column and sea ice in the Ross Sea consumers is channelled mainly through the copepods. However, the trophic connection between primary producers and copepods is usually not direct. Heterotrophic flagellates and larger heterotrophic microplankton (including dinoflagellates, tintinnids, other ciliates, and eggs and developmental stages of metazoans) graze primary production and often form a large part of the diet of many copepods (Umani et al. 1998, Caron et al. 2000).

Two species of krill are found in the Ross Sea: *Euphausia crystallorophias* and *E. superba*. *E. crystallorophias* is only found over the shelf and *E. superba* is found primarily along the continental slope. Although they form an important link between the water column, sea ice and larger predators, they are believed to be less productive and have slower turnover rates than the large epipelagic copepods (*Calanoides acutus*, *Calanus propinquus*, *Rhincalanus gigas* and *Metridia gerlachei*) (Voronina 1998, Tarling et al. 2004). Neither species of krill seems to be as abundant in the Ross Sea as *E. superba* is in the Scotia Sea, where a commercial krill fishery operates and specialist krill predators dominate the ecosystem.

In addition to krill, Antarctic silverfish (*Pleuragramma antarctica*) are a major link between mesozooplankton (mainly copepods) and the larger predators. *P. antarctica* are found in the diet of all large animals (seabirds, seals, toothed and baleen whales, toothfish, many other species of fish, squid) (DeWitt 1970, Laws 1984, Eastman 1985, Vacchi et al. 2017). Throughout their life history the distribution of Antarctic silverfish is thought to include the whole Ross Sea shelf and slope (Hubold 1985), and their juveniles dominate the Ross Sea ichthyoplankton.

More than 100 species and 18 families of fishes have been recorded from the Ross Sea shelf and slope (Chernova & Eastman 2001, Eastman & Hubold 1999, Stewart & Roberts 2001, Bradford-Grieve & Fenwick 2001). Little is known of the abundance of many of these fish species.

The fish fauna of the Ross Sea region can be divided into: (1) a coastal (shelf) fauna, (2) a continental slope fauna and (3) a northern, deeper, oceanic fauna. The shelf fish fauna is dominated (over 90% of biomass) by the four notothenioid families (Nototheniidae, Artedidraconidae, Bathydraconidae, and Channichthyidae), which are endemic to high-latitude Antarctic waters (La Mesa et al. 2004). The benthic shelf fish fauna is species-rich, but the number of species decreases with depth, particularly past the shelf break. Many species have a circum-Antarctic distribution. The Ross Sea slope fish fauna is dominated (in terms of biomass) by the macrourids *Macrourus whitsoni* and *M. caml*, skates (especially *Bathyraja eatonii*), icefish (*Chionobathyscus dewitti*) and eel cods (*Muraenolepis* sp.). To the north of the Ross Sea shelf, the fish fauna is

[8] Polynya is a stretch of open water surrounded by ice.

[9] Epontic: referring to organisms closely associated with sea ice.

dominated by the small pelagic lanternfishes (myctophidae), especially *Electrona antarctica*, *E. carlsbergii*, *Gymnoscopelus braueri* and *G. nicholsi*; Antarctic silverfish are not found north of the Ross Sea slope.

Cephalopods (squid and octopods) are likely to be important components of the Ross Sea ecosystem as they appear in the diets of many predators (Rodhouse, 2013), but their abundance and trophic roles are poorly known (Okutani 1995, Thompson et al. 2012).

Avian abundance in the Ross Sea region is dominated by penguins. About 38% of the world population of Adélie penguins (*Pygoscelis adeliae*) reside in the Ross Sea, breeding at 35 rookeries (Figure 17.8) with a total of about 1 million breeding pairs (Young 1981, Kooyman & Mullins 1990, Lyver et al. 2014). There are more than 40 000 pairs of emperor penguins (*Aptenodytes forsteri*) breeding between Cape Roget and Cape Crozier, and at Cape Colbeck (Young 1981, Harper et al. 1984, Kooyman & Mullins 1990, Wienecke 2011). There are a significant number of non-breeders and juvenile birds in addition to these breeders.

Seals are the most common marine mammals in the Ross Sea region, with more than 200 000 crabeater seals (*Lobodon carcinophaga*) alone (Ainley 1985). Weddell seals (*Leptonychotes weddellii*) are likely to be the second-most common seal in the Ross Sea, with estimates for the larger Ross Sea region of 32 000 individuals (Stirling 1969, Ainley 1985, Stewart et al. 2003), or about 45% of the entire Pacific sector population. There is debate over the degree to which Weddell seals are migratory. Some individuals may remain in residence year round in the fast ice at latitudes as high as 78°S in McMurdo Sound. Others, particularly newly weaned and sub-adult animals, might disperse north and east from the continent in the Ross Sea and may spend the winter in the pack ice north of the Ross Sea (Goetz 2015). Smaller numbers of Ross seal (*Ommatophoca rossii*) and leopard seal (*Hydrurga leptonyx*) breed in the region, but abundances are not well known (Ainley 1985, Pinkerton et al. 2010a). Southern elephant seal (*Mirounga leonina*) are also present in the Ross Sea region but the nearest breeding colony is on Macquarie Island.

The movements of minke and other baleen whales are poorly understood. In the summer, baleen whales present include minke whale (*Balaenoptera bonaerensis*), fin whale (*Balaenoptera physalus*), humpback whale (*Megaptera novaeangliae*), sei whale (*Balaenoptera borealis*) and blue whale (*Balaenoptera musculus*). They tend to congregate in a feeding zone associated with the pack ice north of the Ross Sea slope where krill are abundant. Over the Ross Sea shelf, humpback and sei whales are largely absent (Ainley 1985, Pinkerton et al. 2010b), although minke whales are relatively common in summer

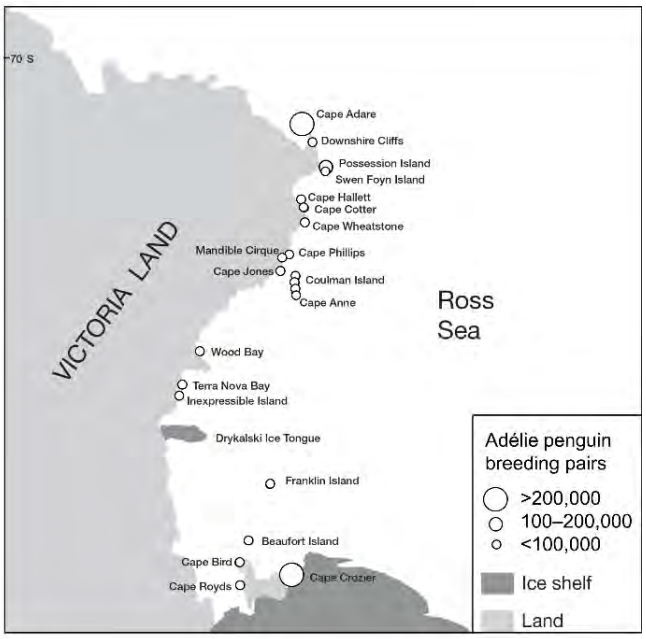


Figure 17.8: The number of breeding pairs of Adélie penguins in the Ross Sea from aerial census methods between 2001 and 2013 (Lyver, unpublished data).

Toothed whales present in the Ross Sea region include sperm whale (*Physeter macrocephalus*), killer whale (*Orcinus orca*), southern bottlenose whale (*Hyperoodon planifrons*), Arnoux's beaked whale (*Berardius arnuxii*). Information on the seasonal abundance of toothed whales in the Ross Sea is rather limited, coming primarily from infrequent surveys of their distribution and numbers (e.g., Ainley 1985). There are at least three different types of killer whale in the Ross Sea region (Pitman et al. 2001, Pitman & Ensor 2003, Pitman 2003, Eisert et al. 2015). Type-C (fish-eating) killer whale are considered to be by far the most common form in the McMurdo Sound region (extreme south-west of the Ross Sea), but the migration and feeding characteristics of this type are not well known.

The Ross Sea benthic fauna has high diversity in some taxa, but lacks crabs and lobsters and has low diversity of some major groups such as gastropods, bivalves, polychaetes and amphipods. There is a dominance of sessile animals, and benthic communities may be multi-storeyed (i.e., occurring in different layers in some areas). Gigantism is found

amongst sponges, pycnogonids, amphipods, isopods, and polychaetes.

A review of the biodiversity of the Ross Sea was provided by Bradford-Grieve & Fenwick (2001). However, in contrast, relatively little is known about the biodiversity, structure or dynamics of the ecosystem of the Amundsen Sea region.

17.2.2 TROPHIC MODELLING

Species in an ecosystem are connected in many ways, but one of the main types of connection is trophic, i.e., the feeding of one organism on another within the food web (McCann et al. 1998, Pace et al. 1999, Frank et al. 2005). Research on the structure of the food web of the Ross Sea has culminated in complex qualitative descriptions (e.g., Smith et al. 2007, Smith et al. 2012) and a quantitative mass-balance model (Pinkerton et al. 2010a, 2016).

The Ross Sea trophic model describes food web structure in a typical year during the period 1990–2000 when fishing has not reduced the toothfish population (Mormede et al. 2015). Biomass and flows were modelled in terms of organic carbon density (gC m^{-2}) as a proxy for energy flow (Figure 17.9). The Ross Sea trophic model covers an area of 637 000 km^2, which includes the Ross Sea shelf and slope and includes 41 trophic groups. The modelling framework for the trophic model is a mass-balance similar to that of Ecopath (Christensen & Walters 2004, Christensen et al. 2008), but non-trophic transfers (including the release of material from sea ice to the water column and vertical detrital flux) were included. Detailed information on the estimation of the parameters is available online from the NIWA website. Revisions and updates to the model are detailed in Pinkerton et al. (2016).

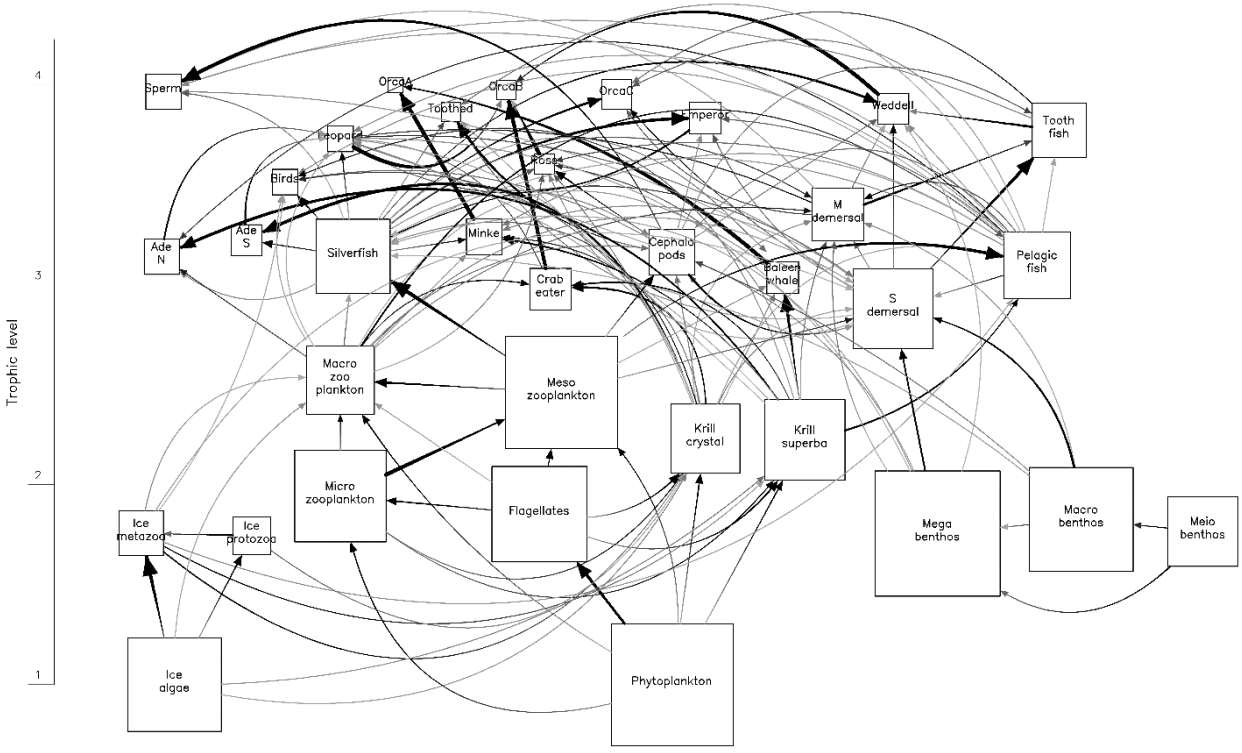


Figure 17.9: Ross Sea trophic model flow diagram, with arrows showing the direction of organic carbon flow. Bacterial and detrital groups omitted for clarity. Bigger boxes indicate more biomass. Boxes are positioned vertically according to trophic level. Thicker/darker lines show higher flows in or out of the group in proportion to total flows in or out of the group. Sperm = sperm whale; Orca-A = type A killer whale; Orca-B = type B killer whale; Orca-C = type C killer whale; Toothed = other toothed whales; Minke = minke whale; Crabeater = crabeater seal; Weddell = Weddell seal; Leopard = leopard seal; Ross= Ross seal; Ade N = Adélie penguins from northern Ross Sea breeding colonies (north of and including Wood Bay); Ade S = Adélie penguins from southern Ross Sea breeding colonies; Birds = flying birds; M demersal = medium-sized (40–100 cm total length) demersal fish ; S demersal = small demersal fish (<40 cm total length). [Pinkerton et al. 2010a, 2016].

17.2.3 HISTORICAL HUMAN EFFECTS ON THE ROSS SEA ECOSYSTEM

The Ross Sea has been identified as the one of the ocean regions least affected by human activity (Halpern et al. 2008). Major industrial sealing did not affect the Ross Sea, although early polar expeditions killed an estimated 2000 Weddell seals to supply dog food in southern McMurdo Sound (Ainley 2009). The Weddell seal population in the southern Ross Sea was thought to have recovered from the mortality caused by polar expeditions by 1950 (Stirling 1971).

Blue, fin and sei whale were taken from the continental slope of the Ross Sea in the 1920s–70s but little whaling was carried out over the Ross Sea shelf itself (Ainley 2009). The removal of an estimated 9330 blue whales from the Ross Sea region (Ainley 2009) may have represented most of the local population of this species. Subsequent industrial whaling for minke whales during the 1970s–80s was largely confined to waters north, east and west of the Ross Sea (Ainley 2009), and the minke whale population seems to have recovered after whaling ceased in the 1980s (Branch 2006). Catches of southern right whales (*Eubalaena australis*) and sperm whales in the Ross Sea region were also low and confined to waters north of the Ross Sea slope (Whitehead 2000, Ainley 2009). In the early 1980s whalers from the former Soviet Union killed more than 900 killer whales in one season (Pitman 2003), which represents a significant perturbation to a population estimated at about 3000 animals (Ainley 2009).

Before the advent of the toothfish fishery in 1997 there was no commercial fishing for finfish in the Ross Sea region.

17.3 ECOSYSTEM EFFECTS OF FISHING IN THE ANTARCTIC

17.3.1 INTRODUCTION

CCAMLR's approach to management recognises that species in an ecosystem are linked (Constable et al. 2000, Kock 2000). Target species are often important components of the ecosystem. Changing their abundance may substantially impact related and dependent species, and affect whole-system dynamics and resilience (Murawski 2000, ICES 2005). To develop management in the Ross Sea and Amundsen Sea regions consistent with CCAMLR's principles of conservation has required the management scope to extend beyond single-stock reference points (Hanchet et al. 2014).

Research and management of the ecosystem effects of fishing are more advanced in the Ross Sea region than in the Amundsen Sea region. The toothfish fishery in the Ross Sea has been operating for longer than in the Amundsen Sea, and the development of a stable stock assessment model in the former (since 2004) has allowed more focus on ecosystem effects of fishing there.

17.3.2 EFFECTS ON BYCATCH SPECIES

Seabird mortality

There are two potential impacts of the fishery on seabirds in the Ross Sea region: (1) direct mortality of flying birds from interaction with fishing gear; (2) indirect impacts on seabirds due to trophic effects (e.g., changes in availability of prey for seabirds – see Section 17.3.5). Extensive measures to mitigate the direct effects of fishing on seabirds in the Ross Sea have been in place since the initiation of the fishery (Reid et al. 2010; CMs 24-02, 25-02). These include the use of streamer lines, the use of weights or weighted lines to enable faster line sink rates, and no discharge of offal south of 60°S. Since the beginning of the fishery in 1997, only two seabirds have been caught by fishing vessels.

Mammal mortality

There has also been no reported bycatch of marine mammals on longlines in the toothfish fisheries of the Ross Sea or Amundsen Sea regions.

Fish bycatch

A detailed characterisation of the bycatch in the toothfish fishery in the Ross Sea region was carried out by Stevenson et al. (2012). Fishery bycatch in the Amundsen Sea region has not yet been characterised in detail. The main bycatch species in the Ross Sea region are macrourids or grenadiers (*Macrourus whitsoni* and *M. caml*), icefish (mainly *Chionobathyscus dewitti*), skates (mainly *Amblyraja georgiana*), eel cods (*Muraenolepis* spp.) and deepsea (morid) cods (*Antimora rostrata*). A small bycatch of rock cods and ice cods is also taken. Spatial distributions of fish bycatch are given in Figure 17.10

The highest catch rates for macrourids, skates and eel cods are on the Ross Sea continental slope, in the area of the

Iselin Bank. Icefish catch rates are higher on the Ross Sea slope and on the Amundsen Sea seamounts than elsewhere in the region. Deepsea (morid) cods have a more northern range and higher catch rates occur over seamounts in the Pacific-Antarctic Ridge. Rock cods and ice-cods tend to occur at shallower depths, especially over the Ross Sea shelf and around the Balleny Islands.

Except for skates and rays, the main bycatch species in the toothfish fishery are also the main prey items for toothfish (Fenaughty et al. 2003, Stevens et al. 2014). One of the reasons is the paucity of other large teleost or squid prey in the Ross Sea region (Bradford-Grieve & Fenwick 2001, Smith et al. 2012). For macrourids and icefish, it is likely that the predation release effect (see Section 17.3.3; Soulé et al. 1988, Prugh et al. 2009) may be stronger than the direct effect of fishing mortality on these species.

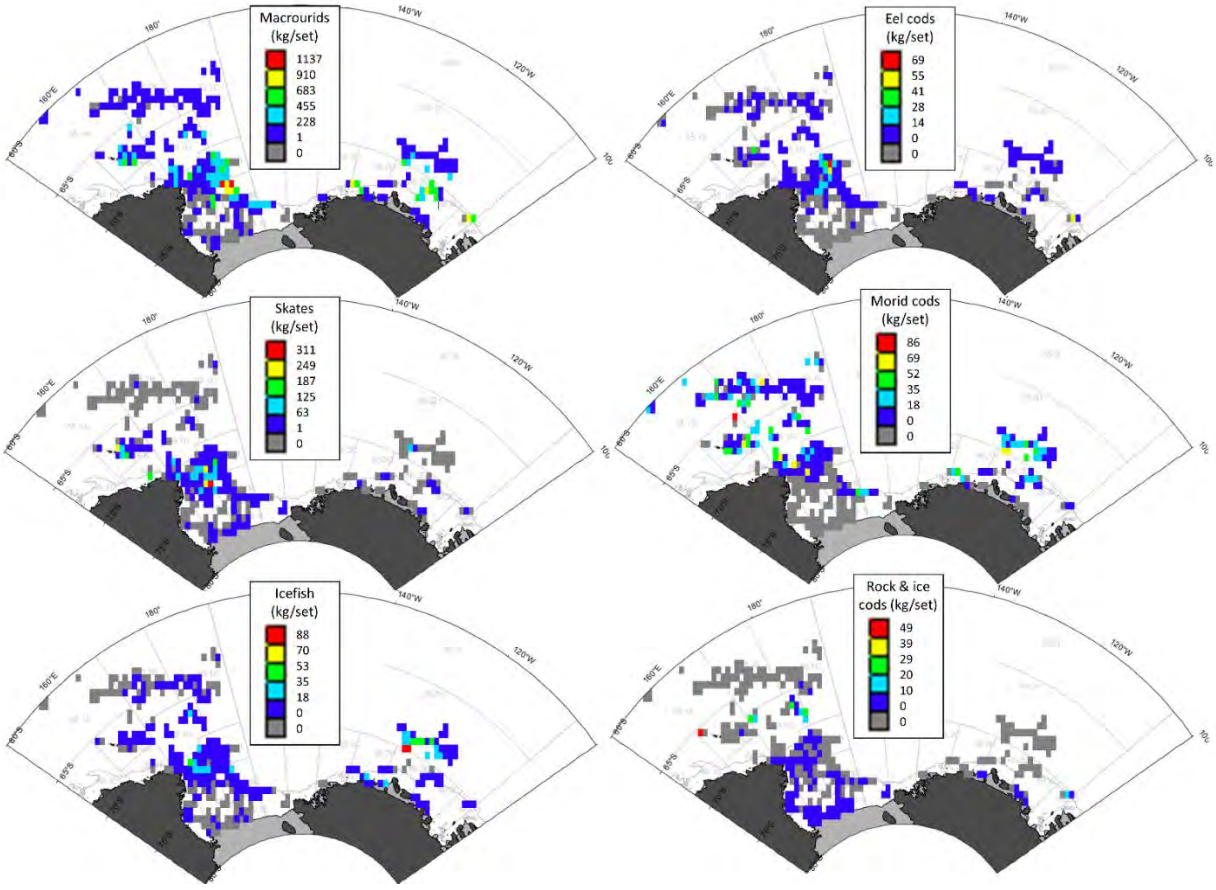


Figure 17.10: Catch rates of macrourids, skates, and icefish (left column); eel cods, morid cods, and rock cods & ice cods (right column). Depth contour at 1000 m. From Stevenson et al. (2012).

Macrourids

The main bycatch species in the Ross Sea are macrourids, which form 5–10% of the total catch per year or about 150 t/y (Large et al. 2015). Macrourid bycatch in the Ross Sea region was considered to be almost exclusively *M. whitsoni* (Regan 1913) until samples collected on the IPY-CAML voyage in 2008 led to the identification of a new species, *M. caml* (Smith et al. 2011, McMillan et al. 2012). The relative proportion of *M. caml* to *M. whitsoni* in the catch has not been assessed, but observers have been gathering information on this since 2013.

There are several approaches in place to mitigate macrourid bycatch, including Subarea and amalgamated SSRU limits on the amount of bycatch (CM 41/09, 41/10), introduced in 2002. Macrourid catch limits were initially based on analogy to fisheries in other areas of the Southern Ocean, but more recently based on the results of the IPY-CAMLR trawl survey of the Ross Sea slope. Bycatch limits for macrourids were exceeded in a number of SSRUs during the early period of the fishery but since 2007 the total macrourid catch has always been less than half of the macrourid catch limit. To help prevent localised depletion of macrourids, 'move-on' rules were introduced in the

2001/02 season (CM 33-03). These rules require a vessel to move to another location at least 5 n. miles distant if the bycatch of any one species is equal to or greater than 1 t in any one set. An additional measure in CM 33-03 makes vessels responsible for managing their individual macrourid bycatch by requiring a vessel to cease fishing in an SSRU for the remainder of the season if its macrourid catch exceeds 16% of its catch of *Dissostichus* spp.

Managing the effects of fishing on macrourids has been applied to all species of macrourid combined. Although the two species of macrourid seem to occur in the same places and at the same depths, their longevities and ages at maturity differ; females of *M. whitsoni* only become sexually mature at 79% of L_{inf} (maximum length) as opposed to about 50%L_{inf} as is common for most macrourids including *M. caml* (Pinkerton et al. 2013). This may make *M. whitsoni* more vulnerable than *M. caml* to the effects of fishing (Reynolds et al. 2005). While there remains the potential for fishing to affect the two species of macrourid differently in the Ross Sea region, perhaps necessitating species-specific management, information is not yet available to develop this.

Skates and rays (rajids)

Skates and rays (rajids) are the second highest group of bycatch species. They form less than 1% of the catch brought onboard as most are cut-off alive at the surface (Large et al. 2015). The main skate caught is the Antarctic starry skate (*Amblyraja georgiana*).

Rajids are required to be brought onboard or alongside the hauler to be checked for tags from historical tagging and for their condition to be assessed. All rajids which are caught alive and with 'a high probability for survival' are released alive at the surface; any 'dead or injured skates' are retained onboard (CM 33-03). The retained catch of rajids is very low (1 t/y) and has never exceeded the bycatch limit for rajids (Large et al. 2015). Tagging of skates and rays as part of CCAMLR fisheries is not required at present.

In the Ross Sea region, the highest catch rates of the Antarctic starry skate are in 850–1350 m, whereas Eaton's skates (*Bathyraja eatonii*) are generally caught in 750–850 m depths. Catch rates are much lower than those of starry skates (Mormede & Dunn 2010). There have been some measurements of skate survival rates in longline fisheries in the Southern Ocean (Endicott & Agnew 2004), but little data on survival of skates caught shallower than 1200 m. Skate survivorship experiments in South Georgia (Subarea 48.3) show that some skates (N=95 fish) survive the capture event, at least for 12 hrs following capture, and that survival rates are higher at shallower depths (Endicott & Agnew 2004). Although the survival rate of released skates in the Ross Sea region is unknown there have been 179 recoveries of tagged skates (Mormede & Dunn 2010). There is also a move-on rule in place to help prevent localised depletion of rajids (CM 33-03). Potential methods for monitoring skates in the Ross Sea region were reviewed by O'Driscoll et al. (2005), who concluded that a tag-recapture experiment was likely to be most successful for monitoring skates.

A preliminary stock assessment based on skate tag-recapture data and ancillary fishery data was completed by Dunn et al. (2007). They identified several problems with the data currently being collected and made the following recommendations: improve species identification, improve detection of tagged skates, increase number of skates measured and sexed, validate the estimates of age and growth, revise skate tagging protocols and undertake additional survivorship experiments. Following the CCAMLR 'Year of the Skate' in 2008–09 an updated characterisation of skate catches was carried out by Mormede & Dunn (2010). They noted that, up to and including the 2010 season, a total of 14 000 skates had been tagged and released and a total of 179 skates had been recaptured. The return rates for tagged skates in CCAMLR fisheries is typically lower than from tagging programmes elsewhere in the world and the reasons for this are unclear (McCully et al. 2013).

The medium-term research plan (Delegations of New Zealand, Norway and the United Kingdom 2014; Section 17.3.9) has identified that further analysis is needed to understand the effect of fishing on rajids in the Ross Sea region.

Icefish

Icefish are caught in low numbers with bottom-longline and trawl gears throughout the Southern Ocean (CCAMLR 2014). In the Ross Sea region the bycatch of icefish in the toothfish fishery is usually of the order of 10 t/y. Since 1998, a total of 11 species codes have been used for icefish caught in the Ross Sea fishery. Although most of the icefish catches were reported as 'unspecified icefish' (Stevenson et al. 2012), most of these are likely to be *C. dewitti* (Sutton et al. 2008).

In the Ross Sea region, *C. dewitti* becomes sexually mature aged about four years and the oldest fish aged was 12 years

old from a sample size of 296 fish (Sutton et al. 2008). Icefish are a major prey of toothfish, comprising 20–25% by weight of the prey of sub-adult and adult toothfish on the Ross Sea slope. Icefish are less common over the northern seamounts in the Ross Sea region where they comprise less than 5% by weight of the diet of toothfish.

Eel cods

Muraenolepsis ('eel cods') occur over the continental shelf and slope of cold temperate and Antarctic southern hemisphere (Nelson 2006). They are caught in low numbers with bottom longline and trawl gears throughout the Ross Sea region (Parker et al. 2012). On northern Iselin Bank where the median catch rate in the Ross Sea region is highest, catches are less than 0.01 kg/hook. Overall catch and catch rate in Subarea 88.1 has been stable throughout the fishery, with one year of high catch (2007) when there were 19 sets reporting greater than 100 kg of *Muraenolepis* (Parker et al. 2012).

Morphological identification of eel cod species continues to be difficult and previous identifications of *Muraenolepis microps* from the Ross Sea region are now considered incorrect (Parker et al. 2012). Thirteen Muraenolepis specimens captured in the Ross Sea region during the 2008 IPY/CAML voyage were identified based on morphology as *M. evseenkoi* (identification by Te Papa Tongarewa Museum of New Zealand).

Genetic methods appear to be more effective than morphology at identifying eel cod species and are increasingly used (Fitzcharles 2014). Genetic identification of more than a hundred specimens indicates that eel cods on the Ross Sea slope are exclusively *M. evseenkoi* (Fitzcharles 2014). Eel cods caught over the Pacific-Antarctic fracture zone in the north of the Ross Sea region were identified genetically as predominantly *M. evseenkoi* with a single specimen of *M. microcephalus* (Fitzcharles 2014).

The biological studies published on *Muraenolepis* species suggest a relatively fast growing, semelparous,[10] life history with a maximum age of 11 years (Parker et al. 2012). In the Ross Sea, eel cods selected by longline gear are almost exclusively female, and a localised area of high catch rates occurs on Iselin Bank on Ross Sea slope. Eel cods comprise a total of about 11% by weight of prey of sub-adult toothfish and about 14% by weight of prey of adult toothfish on the Ross Sea slope (Stevens et al. 2014).

Fishing is likely to affect eel cods in the Ross Sea region by a combination of predation release (fewer toothfish consuming eels cods) and fishing mortality (increased overall mortality), which act in opposition. The overall effects of fishing on this bycatch species depend on factors such as the distribution pattern and total biomass of eel cods, as well as their productivity. Further directed sampling to determine species composition, life-history attributes, reproductive strategy, and sex-specific distribution, and any trends in biomass is needed from the Ross Sea area and throughout the CCAMLR Convention Area (Parker et al. 2012).

Deepsea (morid) cods

Catches of deepsea (morid) cods are dominated by *Antimora rostrata*. This species has a wide spatial distribution, reaching to the New Zealand EEZ where it is called 'violet cod' or 'blue antimora'. The stock structure of this species is unknown. The species forms less than 2% of the diet of toothfish on the Ross Sea slope, but about 20% by weight of diet over the northern (seamount) region of the Ross Sea (Stevens et al. 2014).

Rock cods and ice cods

Rock cods and ice cods (Nototheniidae) formed 0.01% of the reported catch between 1997 and 2012 (Stevenson et al. 2012). Annual average reported catch in this period was about 5 t, and most of this catch was taken from SSRUs 88.1G, 88.1L, and 88.1M (Figure 17.10).

The highest catch rates for rock cods in this period were in a narrow depth band of 400–600 m (Stevenson et al. 2012).

Four different codes have been used to record rock cod and ice cod catches in the Ross Sea region and it is likely that different species dominate this group in different SSRUs. In SSRUs 88.1E and 88.1G, the highest mean catch rates are likely to mainly comprise the striped rock cod (*Lepidonotothen kempi*), as this species was the most

[10] 'Semelparous' means the adults breed once in their life then die.

abundant species caught in research trawls and observed on videos during the BioRoss and IPY-CAML biodiversity surveys (Clark et al. 2010). Catches on the Ross Sea shelf are likely to mainly comprise the deepwater notothen (*Trematomus loenbergii*), as this species was the most commonly caught species in the sub-adult toothfish survey over the southern Ross Sea shelf (Hanchet et al. 2012).

17.3.3 EFFECTS ON PREY SPECIES

Fishing can reduce predation on prey species by removing parts of the predator population. This can lead to mesopredator (or predation) release (Soulé et al. 1988, Prugh et al. 2009).

Empirical meta-analysis suggests that predation release tends to be weaker in pelagic marine and terrestrial systems than in benthic marine and freshwater systems (Shurin et al. 2002). Predation release tends to be stronger where the predator is large and mobile, has high metabolic rate, where prey species are long-lived, functional predator diversity is low, and predator intraguild predation is weak or absent (Borer et al. 2005, Heithaus et al. 2008).

Many of these factors are present in the Ross Sea. On the Ross Sea continental slope, where the majority of the regional Antarctic toothfish population feeds (Hanchet et al. 2008), toothfish are likely to be by far the major predators of macrourids, icefish and eel-cods (Pinkerton et al. 2010a, Stevens et al. 2014, Pinkerton & Bradford-Grieve 2014). There are no other piscine predators of the size of Antarctic toothfish over the Ross Sea shelf and slope (Smith et al. 2012). Some prey species of toothfish have relatively high longevities and low productivity rates. Macrourids tend to be long-lived (Bergstad 1995, Kelly et al. 1997) and, in the Ross Sea region, otolith ageing found maximum recorded ages of 27 years (*M. whitsoni*, n=227) and 62 years (*M. caml*, n=319). In contrast, *C. dewitti* and *M. evseenkoi* are faster-growing and shorter-lived species (maximum recorded age of 12 years). One mitigating factor against strong top-down changes to prey species is the relatively low consumption rate of toothfish, which is likely to be only one to two times its body mass per year because of its large size and the cold water (Pinkerton et al. 2010a).

Modelling of specific cases of predation release in marine systems are few (Prugh et al. 2009) partly because reliable information on marine predators is often scarce (Heupel et al. 2014). A number of approaches have been used to investigate ecological interactions in marine systems including full-ecosystem models (Plagányi 2007, Rose et al. 2010) and mixed-trophic impact analysis (Ulanowicz & Puccia 1990).

Mixed trophic impact analysis was applied to the Ross Sea trophic model (Pinkerton et al. 2010a, Pinkerton & Bradford-Grieve 2014) and suggested a strong trophic connection between toothfish and medium-sized demersal fish (mainly macrourids and icefish). In the Ross Sea trophic model toothfish consumed 64% of the annual production of medium-sized demersal fish. This led to the strongest, top-down impact in the whole multiple-step analysis of Pinkerton & Bradford-Grieve (2014) who concluded that at least some piscine prey of toothfish will experience a relatively strong predation-release effect as the abundance of toothfish is reduced by fishing.

Such 'whole system' approaches tend not to consider interactions over small spatial scales or affecting only parts of populations, and their ability to reliably represent the dynamics of whole ecosystems remains limited (Beckage et al. 2011, Planque 2015). Modelling predation release within a key subset of the whole marine system may be more robust and hence more useful for fisheries management (Plagányi 2007, Plagányi et al. 2014).

To explore the potential effects of the toothfish fishery on these medium-sized demersal fish, a minimum realistic model (MRM) of Antarctic toothfish, macrourids and icefish was developed (Mormede et al. 2014e). This was spatially explicit and dynamic, and based on a model of predator-prey interactions for the Ross Sea Region. The MRM included age-based population dynamics of toothfish, macrourids, and icefish, and includes natural mortality, predation mortality and fishing mortality on all three species. The MRM suggested that the predation release caused by the fishery effect on toothfish abundance was greater than the direct fishing mortality on both prey species and that icefish were expected to show a larger increase in biomass through time than macrourids (Mormede et al. 2014e). This may affect the proportions of macrourids and icefish in the diet of toothfish over time (Mormede et al. 2014e).

17.3.4 EFFECTS ON PREDATOR SPECIES

Three species are known to predate toothfish in the Ross Sea region: Weddell seals, type C killer whales, and sperm whales. Other species discussed below may also consume toothfish. In assessing the potential consequences of

fishing to its predators of toothfish, two factors are important:

1. To what extent is the predator population *ecologically dependent* on toothfish as a prey item? This includes aspects such as the proportion of toothfish in the predator's diet and whether alternative prey items are available (and at what additional ecological cost to the predator). Also relevant is whether toothfish is especially important as prey at a particular time of year, in a particular area or to a particular part of the predator population.

2. To what extent will the fishery reduce the availability of toothfish to the predators at ecologically relevant scales – i.e., taking into account temporal, spatial and population factors?

Weddell seal

There remains uncertainty over the degree to which Weddell seals are ecologically dependent on toothfish as prey (Pinkerton et al. 2008, Eisert et al. 2013). Nutritional analysis of Ross Sea prey suggests that toothfish may represent a unique high-energy food resource for Weddell seals that may not be replaceable by other prey, in particular during periods of high energy demand such as late-stage lactation and the post-breeding recovery of body weight and condition for adult females (Eisert et al. 2013).

Changes to toothfish availability near Weddell seal breeding colonies in the period between pupping and weaning could affect survival of Weddell seal pups and lactating mothers, and fertility rates in the following season, and hence have a compounding impact on Weddell seal populations in these areas (e.g., Pinkerton et al. 2008, Eisert et al. 2013).

Eisert et al. (2013) recommended that the assumed dominance of Antarctic silverfish in Weddell seal diets should be re-examined given the known biases of methods used to derive diet estimates; while large (over 30 g) silverfish occurring at high densities are likely to be a valuable nutritional resource to Weddell seals, smaller size classes of silverfish are unlikely to be adequate to meet the estimated energy requirements of adult Weddell seals.

Killer whale

Killer whales are considered to constitute a single species throughout the world (Rice 1998) but there are at least four different forms (or '*ecotypes*') of killer whale in the Antarctic (Pitman & Ensor 2003). The Ross Sea (or '*type C*') killer whale ecotype is believed to feed almost entirely on fish. There is strong circumstantial evidence that toothfish are an important prey item for type C killer whales in the Ross Sea region (Torres et al. 2013, Eisert et al. 2013, 2014).

The evidence includes:

(1) Killer whale population ecology includes high consumption rates, low abundances, low production rates, often specialised diets, and unknown potential for foraging innovation.

(2) Type C killer whales near McMurdo Sound have been commonly observed carrying toothfish in their mouths (Eisert et al. 2013, 2014).

(3) Comparison of the relative nutrient density of toothfish with silverfish and other prey shows that toothfish represent a high-energy food resource of much higher quality than other potential prey in the Ross Sea region (Eisert et al. 2014). While equivalent energy-dense non-fish prey is available in the Ross Sea (e.g., penguins or seals), observations in northern hemisphere killer whale populations suggest that switching from fish to endotherm (warm blooded) prey is unlikely (Barrett-Lennard et al. 1996, Barrett-Lennard 2011).

(4) An important recent finding is the high incidence of suckling calves observed in type C killer whale groups in McMurdo Sound (Eisert et al. 2014). Caring for young (less than six months old) calves greatly increases the energy requirement of lactating females, not only for milk production, but also because mothers assist their calves through drafting, which increases their own locomotory costs. Revised estimates of energy requirements indicate that lactating female killer whales of the fish-eating ecotype require toothfish to meet their elevated demand.

(5) Densities of other alternative potential prey (Antarctic silverfish, cryopelagic fish) seem too low to justify killer whales coming to the Ross Sea for feeding and the development of a fish-eating ecotype (Eisert et al. 2014).

However, other information on the potential feeding by killer whales on toothfish was inconclusive:

(1) It is not known to what extent toothfish forage pelagically or how deep type C killer whales can dive. Recent information (Pitman, pers. comm., Torres et al. 2013) shows that type C killer whales in the Ross Sea can routinely dive to 200–400 m, with a maximum of over 700 m. This is deep enough to reach demersal prey over much of the Ross Sea shelf, but foraging times would be short at these depths.

(2) Stable isotope analysis of tissue from killer whales and toothfish was used to test the hypothesis that toothfish were a major prey in the Ross Sea in summer but was inconclusive.

The balance of evidence suggests that toothfish are likely to form a significant part of the diet of type C killer whales in McMurdo Sound in summer, but it is not possible to say whether toothfish are an important prey item to type C killer whales in other locations on the Ross Sea shelf (e.g., Terra Nova Bay, Bay of Whales, Sulzberger Bay) or at the scale of the whole Ross Sea shelf and slope (Torres et al. 2013, Eisert et al. 2014).

Recent evidence derived from satellite tagging and photo-identification shows that type C killer whales undergo long-distance travel from the southern Ross Sea to New Zealand waters and into subtropical regions (Figure 17.11; Eisert et al. 2015).

Analysis of photo-ID data indicates that type C killer whales from the Terra Nova Bay area of the Ross Sea show a high degree of seasonal site fidelity. Individual whales returned over different years to areas of ecological significance, including New Zealand waters north and east of East Cape, the Kermadec Trench region, and the Ross Sea (Eisert et al. 2015).

Sperm whale

Sperm whales are migratory and are distributed from the tropics to the pack ice edges in both hemispheres. The subtropical convergence at about 40°S marks the southern limit of females and young males; only the larger males penetrate further south (Lockyer & Brown 1981, Knox 2007).

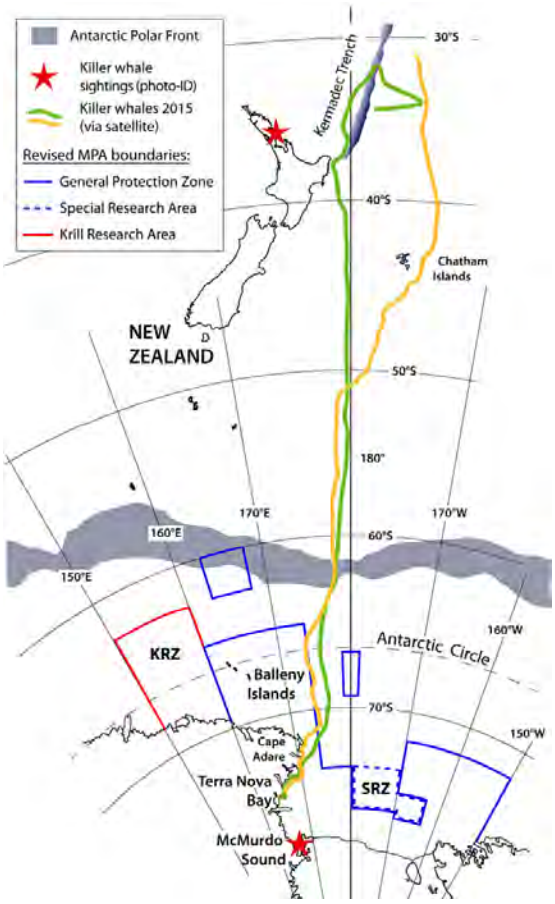


Figure 17.11: Satellite tracking of type C killer whales showing ecological connectivity for this species between the Ross Sea and New Zealand (Eisert et al. 2015).

Present and historical occurrence of sperm whales along the Ross Sea continental slope remains unclear. Kasamatsu & Joyce (1995) reported a southernmost sighting of a sperm whale at 74°S on the Ross Sea slope, and summarised data collected in sighting surveys between 1976–77 and 1987–88 during mid-December to mid-February. The IWC data from the 1990s showed sperm whales sightings in the area of 70–78°S 150–180°E (along the Ross Sea shelf edge). However, sperm whales have rarely been sighted on the Ross Sea slope from fishing vessels (Fenaughty, pers. comm.). We are not aware of any systematic surveys of sperm whales in the Ross Sea or Amundsen Sea regions in the last 30 years.

Sperm whales are deep divers, staying submerged for more than 1 h and commonly diving to about 400 m, reaching depths of 1185 m (Watkins et al. 1993, Perrin et al. 2002). Sperm whales in the Southern Ocean, and Pacific subantarctic are reported as feeding primarily on squid and secondarily on fish (Clarke 1980, Knox 2007, Evans & Hindell 2004). Knox (2007) gives the ratio of squid to fish in their diet as 9:1.

Yukhov (1971, 1972) and Abe & Iwami (1989) described Antarctic toothfish as prey items but proportions are not known. Yukhov (1971, 1972) examined large numbers of stomachs from 12–18 m long sperm whales from the Pacific Ocean sector of the Antarctic from 1965 to 1969 and found that the main prey were cephalopods but that Antarctic toothfish (97–160 cm total length) were also frequently found in the sperm whale stomachs. Although some records were associated with seamounts and ridges, many occurred over deep water (more than 4000 m) suggesting that the sperm whales were feeding pelagically (Yukhov 1972).

Other potential predators of toothfish

Other possible predators of toothfish in the Ross Sea region include Southern elephant seals, Arnoux's beaked whales, and colossal squid.

Southern elephant seals (*Mirounga leonina*) enter the Ross Sea only in the summer from breeding and feeding grounds further to the north. They are likely to be mainly feeding on small pelagic fish, squid and crustaceans (Walters et al. 2014). However, their deep diving depths (about 1500 m) and occurrence around the Ross Sea slope, as well as photographic evidence (Eisert, pers. comm.) show that they also consume toothfish to some extent.

Very little is known about the predation of Arnoux's beaked whales (*Berardius arnuxii*) on toothfish but this is unlikely to be significant. These whales are known to occur in the Ross Sea to 77°S (Eisert, pers. comm.), are capable of diving to depths where toothfish occur on the Ross Sea slope and are predominantly small fish and squid eaters (Walker et al. 2002, Ohizumi et al. 2003).

Beak-shaped bite marks on toothfish caught on longlines suggest some depredation on toothfish by colossal squid (*Mesonychoteuthis hamiltoni*). However, stable isotope analysis of tissue of this species of squid suggests that it is unlikely to feed on toothfish.

Effects of fishing on availability of toothfish as prey

There are four ways in which the fishery could alter the availability of toothfish for predators in the Ross Sea region:

(1) *Smaller stock size*: Fishing leads to fewer fish available as prey.

(2) *Local depletion by fishing within a season*: Fishing may locally reduce toothfish abundance (catch rates typically decline when an area is fished). If fishing occurs in an area where predators forage, the availability of toothfish to predators may be reduced for some time. In 2008, CCAMLR set a zero allowable catch for SSRU, 88.1M (along the Victoria Land coast), which had the effect of moving fishing effort away from the known foraging grounds of Weddell seals and type C killer whales in the south-west Ross Sea.

(3) *Reduced recruitment*: The number of sub-adult toothfish available in the southwest Ross Sea could decline if there was reduced toothfish recruitment. The stock assessment suggests that toothfish spawning biomass in 2015 was about $70\%B_0$[13] (Mormede et al. 2015). At this level, recruitment of toothfish is not estimated to be reduced. Based on the stock-recruit relationship with steepness assumed at 0.75 in the stock assessment (Mormede et al. 2014b), recruitment is predicted to be reduced to about 92% of unfished recruitment when the spawning stock biomass reaches 50% of its unfished status.

(4) *Density-dependent* or *stock-contraction effects*: As has been seen in some other species elsewhere (Swain & Sinclair 1993, Hutchings 1996, Atkinson et al. 1997, Fisher & Frank 2004), fishing may change movement patterns and distribution of toothfish through the Ross Sea region. Changes in the distribution of toothfish that affect abundance at the edges of the toothfish range may be important to their predators.

17.3.5 TROPHIC AND SYSTEM-LEVEL EFFECTS

Changes in the abundance of one species may impact other species that are neither its predators nor its prey. These are called '*second order*' trophic effect, or ecosystem-level effects, and can include trophic cascades and regime shifts. Well-documented oceanographic-induced regime shifts in marine ecosystems have historically had substantial, long-lasting and typically (but not always) negative effects on fisheries. A review of trophic and ecosystem level effects of fishing is given in Chapter 13: Trophic and ecosystem-level effects.

Trophic effects arising from fishing are more likely to be important if the target species has a key role or is of high trophic importance in the ecosystem (Fletcher et al. 2002, Fletcher 2005). An estimate of trophic importance, using mixed trophic impact analysis (Ulanowicz & Puccia 1990) was applied to the Ross Sea (Pinkerton & Bradford-Grieve 2014) based on the Ross Sea trophic model (Section 17.2.2). This concluded that Antarctic toothfish has moderate

trophic importance in the Ross Sea food web as a whole. The analysis did not support the hypothesis that changes to toothfish abundances due to fishing will cascade through the Ross Sea regional ecosystem by simple trophic effects. Pinkerton & Bradford-Grieve (2014) did not rule out cascading effects on the Ross Sea ecosystem due to changes in the abundance of toothfish, but noted that for such changes to occur, a mechanism other than simple trophic interactions would need to be involved. Instead, Pinkerton & Bradford-Grieve (2014) found that trophic importances were highest in the middle-trophic level organisms of the Ross Sea food web. Antarctic silverfish, krill, small demersal and pelagic fishes, cephalopods and mesozooplankton were identified as having key roles in maintaining ecosystem resilience.

17.3.6 EFFECTS ON HABITATS

Vulnerable Marine Ecosystems (VMEs) constitute areas that may be vulnerable to impacts from fishing activities. Taxa considered to comprise VMEs vary geographically. Essentially, VMEs are ecosystems with organisms that create biogenic structures, are fragile relative to the fishing gears in question, are rare or endemic, or have life-history traits that imply slow recovery from disturbance (Rogers et al. 2008, FAO 2008).

In 2007 CCAMLR adopted Conservation Measure (CM) 22-06 requiring Member countries to assess and manage adverse effects of bottom fishing on VMEs in the Convention Area. The *New Zealand Antarctic Bottom Fishing Impact Assessment Workshop* in 2007 identified 14 groups of taxa indicative of habitats or communities where VME organisms occur (Parker et al. 2008). A CCAMLR guide to VME taxa was produced in 2009.[11]

All fishing for toothfish in the Ross Sea and Amundsen Sea regions is by longline, which are laid on or close to the seabed and held down by weights and grapples (Fenaughty 2008). Structure-forming benthic invertebrates can be damaged by the longlines, especially during their hauling (recovery from depth) when the longlines may move laterally. Benthic invertebrates that have been brought to the surface attached to lines in the Ross Sea region include anemones (Actiniaria), stony corals (Scleractinia), gorgonians (Gorgonacea), sponges (Porifera) and ascidians (Ascidiacea) (Parker & Bowden 2009).

The potential for the longlines to significantly affect a particular group of structure-forming benthic habitat in the Ross Sea is related to the spatial scale of the area of contact between fishing gear and the sea floor as a proportion of the total area in which the habitat is present. An impact assessment method developed by Sharp et al. (2009) showed that regardless of the distribution of VME taxa (for which actual spatial distributions are unknown) the cumulative impact on VME organisms of all historical longline fishing effort in the Ross Sea region has been very low. At a very fine scale (i.e., spatial cells measuring 0.05° latitude by 0.167° longitude) fewer than 5% of cells within fishable depths have been fished. Average impacts within fished cells are less than 0.1% total mortality of vulnerable taxa; estimated impact in the single most heavily impacted cell is less than 5% (Sharp 2010). These low impacts reflect both the spatially restricted area within which the fishery operates and the very narrow spatial footprint of individual longlines.

A spatially explicit production model was developed and used to simulate likely population level effects (including recovery) arising from benthic impacts from longline fishing effort in the Ross Sea region (Dunn et al. 2010). Simulations included different productivity assumptions, impact, and spatial scale, with and without management by areal closures. The results of the simulations suggested that management action of areal closures in the Ross Sea region would improve the outcome for VMEs, but given the already low level of impact, that the improvement was very small.

Research has not found significant correlation between the occurrence of VMEs and toothfish abundance within areas fished for toothfish in the Ross Sea region (Parker & Mormede 2009, Parker et al. 2010, Parker & Smith 2010). Dunn et al. (2010) recommended further work on simulating effects of fishing on VMEs, including investigating how changes in the distribution of future fishing may result in alternative impacts or how different

[11] CCAMLR. VME Taxa Classification Guide 2009. Retrieved from https://www.ccamlr.org/en/system/files/VME-guide.pdf.

assumptions of the underlying distributions of benthic organisms may influence the results.

17.3.7 SYNERGISTIC EFFECTS OF FISHING AND CLIMATE CHANGE

There is increasing understanding of the potential impacts of climate change on fisheries (Valdes et al. 2009, Rice & Garcia 2011, IPCC 2014). Fishing can also act synergistically with climate variation/change and lead to ecosystem-level change (e.g., Winder & Schindler 2004, Brierley & Kingsford 2009, Kirby et al. 2009, Perry et al. 2010; see also Chapter 13).

If change to the level of toothfish recruitment in the Ross Sea or Amundsen Sea regions did occur (for example due to effects of fishing and climate change), would the current monitoring and management framework be able to detect this, and after how long?

Changes in age structure caused by changes in recruitment strength would most likely be detected from the fisheries catch data themselves, but would not be apparent until the relevant cohort was of sufficient age to be fully selected by the fishery. Even then, the signal may be confounded with changing effort patterns. Without specific monitoring of sub-adult toothfish in the Ross Sea any substantial change in recruitment would not likely be detected until after some years after it occurs.

This delay in detecting any effect of the fishery on recruitment was one reason for the start of the sub-adult survey for Antarctic toothfish over the southern Ross Sea shelf in 2012 (Hanchet et al. 2012). There have been six surveys to date (Hanchet et al. 2012, Parker et al. 2013b, Mormede et al. 2014d, Hanchet et al. 2015b, Dunn et al. 2016, Large et al. 2017). These surveys use a consistent, stratified design for sampling sub-adult toothfish in order to better estimate recruitment variability and provide an early-warning of changes in toothfish recruitment. It is likely that this survey, if continued on the same basis as at present, would detect changes to recruitment about five years after it occurred. In contrast, in the absence of a fishery-independent survey the relevant cohort would not be available to the commercial fishery for approximately 10 years, and it is possible that any recruitment signal in the fishery-dependent data would be confounded by the effects of variable or uncontrolled commercial fishery selectivity. The survey hence reduces risk of changes in recruitment of toothfish in the Ross Sea region being detected too late for management to respond.

17.3.8 REVERSIBILITY OF ECOSYSTEM EFFECTS OF FISHING

Principle (b) of Article II of the CAMLR Convention requires the *maintenance of ecological relationships* in the ecosystem. Principle (c) of Article II of the CAMLR Convention, states also that changes due to fishing should be *reversible in 20–30 years*.

It has been suggested that trophic interactions can affect the ability of fish populations (and by extension, related or dependent species in the ecosystem) to regain their former characteristics following exploitation (Hutchings 2000, Steele & Schumacher 2000). Predictions as to the reversibility of ecosystem effects of fishing would be limited by three key factors. First, there is presently no information with which to estimate density dependent effects of changes to toothfish (Abrams 2014). Second, the reversibility of different types of ecosystem effects of fishing will vary, so any theoretical investigations of reversibility will need to be carried out for each effect of fishing separately. Third, it is not known whether trophic and ecosystem-level effects, genetic or behavioural factors may come into play should fishing for toothfish cease. At present, there is very limited scientific ability to predict the dynamics of ecosystems (Planque 2015).

Keith & Hutchings (2012) concluded that 'emergent and demographic Allee [density-dependent] effects, coupled with altered interspecific interactions, render questionable the presumption that the recovery of heavily depleted populations can be reliably forecasted by population dynamical behaviour during the decline.' However, in this context, 'heavily depleted' means depleted to much lower levels than the CCAMLR target of $50\%B_0$ so issues of reversibility are likely to be relevant only in the case of *significant overdepletion* of toothfish or arising from *substantial ecosystem effects* in other parts of the system. The focus in CCAMLR and in the Ross Sea and Amundsen Sea regions has hence been on preventing significant overdepletion of target species and on developing indicators for changes in dependent or related species (e.g., CEMP 2004, Delegations of New Zealand, Norway and the United Kingdom 2014). At present, evidence does not suggest that significant overdepletion of target species or substantial ecosystem effects are occurring in the Ross Sea and Amundsen regions.

17.3.9 RESEARCH PRIORITIES

The medium-term (5–10 year) research priorities with regard to the Antarctic toothfish fishery in the Ross Sea and Amundsen Sea regions was updated in 2014 (Delegations of New Zealand, Norway and the United Kingdom 2014). The first two sections of the medium-term research plan (MTRP, Table 17.1) prioritised research to assess, monitor and maintain the reproductive potential of the toothfish population. The third section dealt with issues related to the ecosystem effects of fishing, including reversibility of any effects of fishing.

Table 17.1: Medium-term research plan (MTRP) priorities with regard to the ecosystem effects of the fishery for toothfish in the Ross Sea and Amundsen Sea regions (Delegations of New Zealand, Norway and the United Kingdom 2014). The other two parts of the MTR plan are not shown. These are to (1) reduce uncertainty in toothfish model parameters; and (2) reduce management uncertainty.

Section	Key research priorities	
Maintenance of ecosystem structure and function	(i)	To determine the temporal and spatial extent of the overlap in the distribution of toothfish and its key predators (in particular killer whales and Weddell seals).
	(ii)	To investigate the abundance, foraging ecology, habitat use, functional importance and resilience of key toothfish predators (in particular killer whales and Weddell seals).
	(iii)	To develop methods of monitoring changes in relative abundance of key prey/bycatch species (in particular macrourids and icefish) on the Ross Sea slope and hence assess the potential impact of the toothfish fishery on these species.
	(iv)	To monitor diet of toothfish in key areas, especially on the Ross Sea slope.
	(v)	To simulate the effect of the fishery on populations of toothfish, its predators, and its prey (using Minimum Realistic Models or similar).
	(vi)	To develop quantitative and testable hypotheses as to the 'second-order' effects (such as trophic cascades, regime shift) and ensure data collection is adequate to monitor for any risks deemed reasonable.
	(vii)	To assess the impact of the toothfish fishery on Patagonian toothfish.
	(viii)	To estimate survivorship of released skates.
	(ix)	To develop semi-quantitative and spatially explicit risk assessments for macrourids and Antarctic skates (*A. georgiana*), especially in the slope fishery of the Ross Sea.
	(x)	To develop methods to assess whether the potential impacts of the toothfish fishery on the ecosystem are likely to be reversible in two to three decades.

The research priorities for the ecosystem effects of fishing were:

1. Further analysis is needed to understand the effect of fishing on rajids in the Ross Sea region.

2. To improve our understanding of the effect of fish on the prey assemblage of toothfish, especially in the most heavily-fished area of the Ross Sea slope, further information on the two species of macrourid separately is needed. In particular, information is needed on the relative abundances of *M. whitsoni*, *M. caml*, the relative catch of the two species across the Ross Sea and Amundsen Sea regions and the relative amount of the two species consumed by toothfish. Some of this research is underway. For example, the *RV Tangaroa* voyage to the Ross Sea in February 2015 included three days of a depth-stratified demersal trawl survey of the Iselin Bank (SSRU 88.1I) and the results are being analysed. New Zealand observers have been identifying some of the macrourid bycatch in the Ross Sea region to species level since 2012 (i.e., separating *M. caml* from *M. whitsoni*), and macrourid prey found in the stomachs of toothfish during diet analysis will be identified to species level.

3. The minimum realistic model of interactions between toothfish and key prey species (especially macrourids and icefish) should be further developed. This modelling enables the potential impacts of the fishery on key prey species to be evaluated in order to generate hypotheses of future change, and to design monitoring tools for ecosystem effects.

4. Ongoing monitoring of toothfish diet is recommended, as is the monitoring of the icefish and macrourid populations (especially in SSRUs 88.1H and 88.1K) through the development of age frequencies (length measurements and ageing) (Pinkerton & Bradford-Grieve 2014, Mormede et al. 2014e).

5. Our ability to determine to what extent Weddell seal, type C killer whale and sperm whale populations in the Ross Sea and Amundsen Sea regions are ecologically dependent

on toothfish requires further information on their diet and improved information on their seasonal and spatial abundances.

17.3.10 MARINE PROTECTED AREAS

The CAMLR Convention provides the overarching basis for marine resource conservation in the Southern Ocean. It includes a role for marine protected areas (MPAs). The CCAMLR position on MPAs is given online.[12] A decision was made at the World Summit on Sustainable Development (WSSD) in 2002 in Johannesburg, South Africa, to achieve a representative network of MPAs by 2012. CCAMLR responded to the WSSD target by aiming to establish a representative network of MPAs in the CAMLR Convention Area by 2012.

Globally, spatial fishing closures have been proposed as one way that fisheries management can manage, avoid or mitigate the risk of ecosystem effects of fishing.[13] Although there are different types, in general, an MPA is a kind of spatial fisheries management that provides protection for all or part of the natural resources it contains. MPAs do not necessarily exclude fishing, research or other human activities. MPAs in which no fishing is allowed are often referred to as *no-take areas*. Other uses may still be permitted.

The Ross Sea MPA was approved by CCAMLR at its Commission meeting in October 2016. Conservation Measure 91-05 (2016) (CM 91-05)[14] details the specificities of the MPA. The boundaries of the MPA can be found in the CM (Figure 17.12). It has an area of 1.55 million km^2. It came into force in December 2017 and the period of designation is 35 years. The full chronology and scientific basis for the design and designation of the Ross Sea region MPA by CCAMLR is summarised in Delegations of New Zealand and the United States (2014).

The MPA limits activities inside its boundaries in order to meet conservation, habitat protection, ecosystem monitoring, and fisheries management objectives (Table 17.2). The MPA is divided into three zones:

- The General Protection Zone, which corresponds to 72% of the MPA, is a '*no-take*' zone, which prohibits commercial fishing;
- The Special Research Zone (SRZ), which permits some commercial fishing as a part of scientific research;
- The Krill Research Zone (KRZ), which permits some harvesting of krill as a part of scientific research.

A management plan has been agreed and provides further details about the features or areas within the MPA associated with the specific objectives, as well as the management measures and administrative arrangements for achieving them (Annex 91-05/B of CM 91-05).

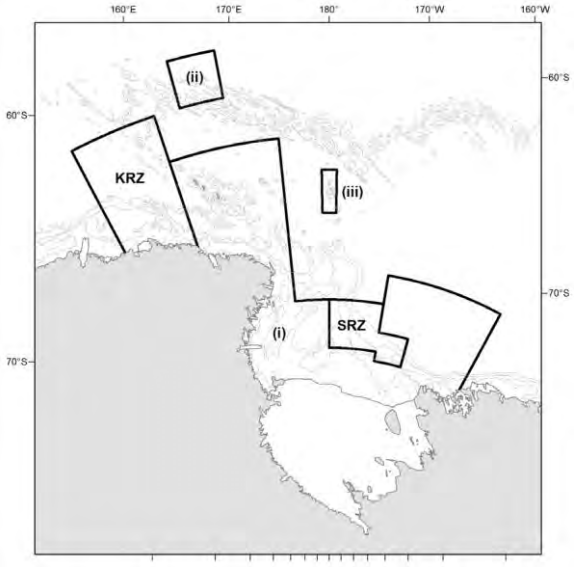


Figure 17.12: Map of the Marine Protected Area in the Ross Sea region. The black lines indicate the boundaries of the General Protection Zone (composed of areas (i), (ii), and (iii)), the Special Research Zone (SRZ), and the Krill Research Zone (KRZ). Depth contours are at 500 m, 1500 m and 2500 m.

A Scientific Research and Monitoring Plan was developed for the October 2017 CCAMLR Commission meeting (Dunn et al. 2017). Priority elements for the plan can be found in Annex 91-05/C of Conservation Measure 91-05.

[12] CCAMLR. Marine Protected Areas. Retrieved from https://www.ccamlr.org/en/science/marine-protected-areas-mpas.

[13] Scientific Consensus Statement on Marine Reserves and Marine Protected Areas, https://www.nceas.ucsb.edu/Consensus.

[14] CCAMLR. Conservation Measure 91-05 (2016). Retrieved from https://www.ccamlr.org/en/measure-91-05-2016.

The Conservation Measure defining the Ross Sea is due for review at least every 10 years to evaluate whether the specific objectives of the MPA are still relevant or being achieved and to evaluate the delivery of the research and monitoring plan.

There are no proposals to establish MPAs in the Amundsen Sea region.

Table 17.2: Objectives of the Marine Protected Area in the Ross Sea region. [Continued on next page]

1	To conserve ecological structure and function throughout the Ross Sea Region at all levels of biological organisation, by protecting habitats that are important to native mammals, birds, fishes and invertebrates.
2	To provide a reference area in which fishing is limited, to better gauge the ecosystem effects of climate change and fishing, and to provide other opportunities for better understanding the Antarctic marine ecosystem.
3	To promote research and other scientific activities (including monitoring) focused on marine living resources.
4	To protect a representative portion of benthic and pelagic marine environments.
5	To protect large-scale ecosystem processes responsible for the productivity and functional integrity of the ecosystem.
6	To protect core distributions of trophically dominant pelagic prey species.
7	To protect core foraging areas for land-based predators or those that may experience direct trophic competition from fisheries.
8	To protect coastal locations of particular ecological importance.
9	To protect areas of importance in the lifecycle of Antarctic toothfish.
10	To protect known rare or vulnerable benthic habitats.
11	To promote research and scientific understanding of krill, including in the Krill Research Zone in the north-western Ross Sea region.

17.4 INDICATORS AND TRENDS

17.4.1 EFFECTS ON BYCATCH SPECIES

Rajids (skates and rays) are the bycatch group deemed at most risk from a direct effect of fishing in the Ross Sea region (Delegations of New Zealand, Norway and the United Kingdom 2014). No information or indicators as to the ecological effects of fishing on rajids in the Ross Sea and Amundsen Sea regions are available. Before the 2008 fishing season skates were cut-off in the water with hook attached. Starting in the 2008 fishing season, skates that were not already tagged (i.e., recaptured tagged fish) and which were deemed to be in reasonable condition were required to be cut-off from longlines (CM 33-03). This led to a fall in the number of rajids landed onboard and an increase in numbers released (Figure 17.13). There is no requirement to tag skates in the Ross Sea or Amundsen Sea regions at present.

Macrourid bycatch in the Ross Sea and Amundsen Sea regions increased to a maximum in 2005–06 as the fisheries expanded and then has decreased (Figure 17.14).

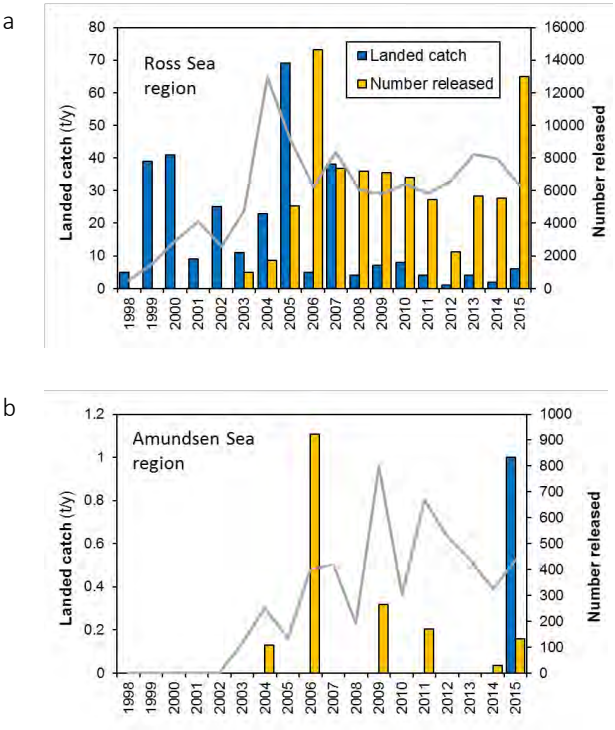


Figure 17.13: Catch of rajids (skates and rays). [a] Ross Sea region; [b] Amundsen Sea region. Weight of landed rajids (blue) and numbers cut off alive (orange). The relative effort (number of sets) is shown as the grey line.

Predation release of macrourids and icefish is expected to be larger than fishing mortality, and may lead to increased abundance over time (Pinkerton & Bradford-Grieve 2014, Mormede et al. 2014e). Analysis of the rates of bycatch for macrourids (*M. whitsoni* and *M. caml*), icefish (principally *Chionobathyscus dewitti*), eel cods (*Muraenolepis* spp.) and deepsea cods (*Antimora rostrata*) has been carried out (), using standardisation to control for area and vessel reporting.

It is likely that changes to CCAMLR management rules aimed at reducing bycatch of macrourids together with more targeted fishing practice has led to decreases in the catch of macrourids in the Ross Sea slope region. These changes in fishing locations and practices are also likely to

have affected catch rates for bycatch species so that changes in catch rates in Figure 17.15 probably do not reflect changes in population sizes.

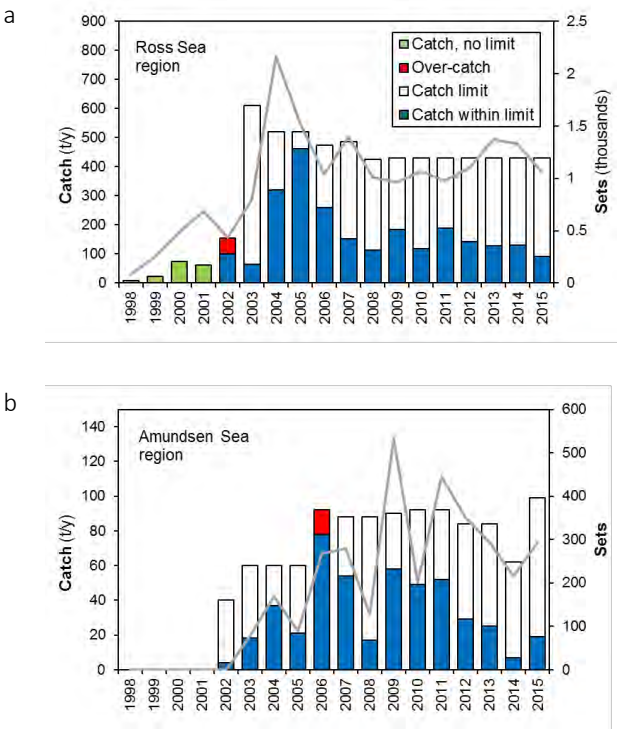
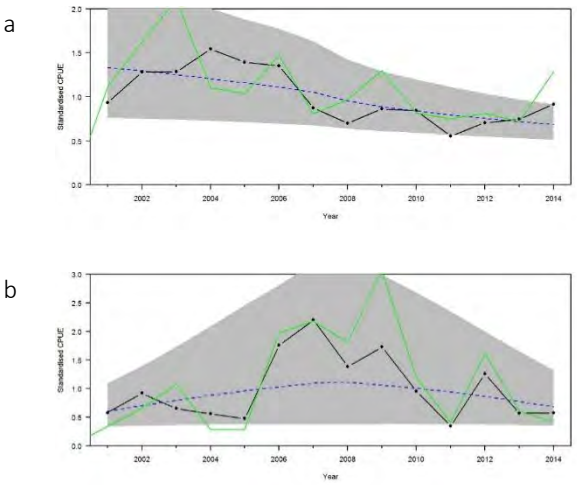


Figure 17.14: Macrourid bycatch (bars) and total fishing effort in terms of number of sets (grey line). [a] Ross Sea region; [b] Amundsen Sea region. White bars shows where the catch limit exceeds the catch, and red bars indicate that catch exceeded the catch limit in that year.



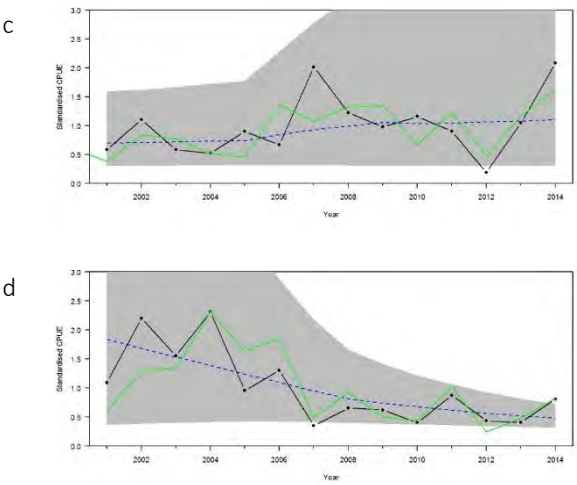


Figure 17.15: Raw (green), standardised (black), trends (blue) and 95% confidence intervals (grey) for catch-per-unit-effort (CPUE) in groups of bycatch species on the Ross Sea continental slope (small-scale research units (SSRUs) 88.1 H and I). [a] Macrourids; [b] icefish; [c] eel cods; [d] deepsea cod.

Alternative methods to look for changes in the population abundance of macrourids over time are being explored, including acoustics (O'Driscoll et al. 2012, Ladroit et al. 2014). Also, 'catch-curve' analysis to explore changes in the total mortality rate of macrourids are being investigated.

17.4.2 EFFECTS ON PREY SPECIES

Both mixed trophic impact analysis and the minimum realistic model of trophic interactions between toothfish, macrourids, and icefish in the Ross Sea region suggest that the toothfish fishery is likely to cause predation release in prey species, especially on the Ross Sea slope (Pinkerton & Bradford-Grieve 2014, Mormede et al. 2014e). The differential strength of the predation release on macrourids and icefish would be likely to lead to a change in the diet of toothfish over time in favour of more icefish being consumed (Mormede et al. 2014e).

Stevens et al. (2014) found no significant temporal change in the diet of toothfish between 2003 and 2010 based on examination of stomach contents of toothfish on the Ross Sea slope. Pinkerton et al. (2014) found a small, but significant reduction in the trophic level of toothfish between 2006 and 2014 in a direction consistent with more icefish and fewer macrourids being consumed.

Monitoring for changes in the diet of toothfish, with a focus on the Ross Sea slope, is a research priority (Delegations of New Zealand, Norway and the United Kingdom 2014) and is continuing through periodic collection of toothfish

stomachs and analysis of toothfish tissue samples by stable isotope analysis to test for changes in trophic level over time.

17.4.3 EFFECTS ON PREDATOR SPECIES

At present, no indicators are available to monitor changes to the ecological state of known predators of toothfish (type C killer whales, Weddell seals and sperm whales) in the Ross Sea or Amundsen Sea regions, and this is a recognised priority for future research (Delegations of New Zealand, Norway and the United Kingdom 2014). The fact that type C killer whales, and potentially sperm whales, move between the Ross Sea and the EEZ, gives New Zealand a key role in the management of risks to these species.

Information is available on the extent to which fishing is likely to have reduced the *availability of toothfish to predators* of toothfish. Two factors are important when considering indicators for changes to the availability of toothfish as relevant to toothfish predators.

First, different predators forage over different spatial scales so that spatial patterns in changes to toothfish abundance over time are important. For example, the foraging ranges of lactating Weddell seals are constrained by the seals having to return to shore to feed the pups. Foraging range of type C killer whales in the Ross Sea is not known, but it appears that the McMurdo Sound is important (Eisert et al. 2015). Sperm whales are unlikely to come south of the Ross Sea slope.

Second, the size of toothfish consumed by predators is important as different size classes of toothfish will be affected differentially over time by fishing. Weddell seals consume toothfish of total length (TL) 60–110 cm (median TL about 80 cm; Kim et al. 2011, Ainley & Siniff 2009). Although information on the size of toothfish taken by killer whales is scarce, type C killer whales appear to predate on larger toothfish than Weddell seals. In the McMurdo Sound region at least, a type C killer whale was observed with an approximately 150 cm TL toothfish (Eisert et al. 2015). This size of toothfish coincides with the modal size classes (130–159 cm TL) of toothfish caught in McMurdo Sound by scientists (Ainley et al. 2013). For sperm whales, because of their ability to access the entire water column, it is likely that all sizes of toothfish present in the Ross Sea slope region are available as prey.

Simulations of changes to the abundance of toothfish by geographic area were generated by the *Spatial Population Model* (SPM; Mormede et al. 2014c). This model estimates the distribution of age-classes of toothfish in the Ross Sea region.

Over the Ross Sea continental shelf (where Weddell seals and type C killer whales overlap in distribution with toothfish) the spatial population model suggests that the biomass of sexually mature toothfish (greater than about 110–130 cm TL; Parker & Marriott 2012) was about 74%B_0 in 2013 and will decrease to about 57%B_0 in 2048 (Pinkerton et al. 2016). In SSRUs 88.1H and 88.1I on the Ross Sea slope (where sperm whales may occur and feed on toothfish) the spatial model suggests that total toothfish biomass (all lengths) in 2013 was 77% of that before fishing, and that this will decrease to 60% of the pre-exploitation biomass by 2048.

Changes to the length-frequency distribution of toothfish taken over the Ross Sea shelf by the fishery are summarised in Large et al. (2015) and can be found in Figure 17.16.

For the southern part of the Ross Sea shelf, the sub-adult survey catches a lower proportion of toothfish over 150 cm TL than the commercial fishery over the whole shelf (Figure 17.17; Hanchet et al. 2015b) but again, changes to the proportion of large toothfish in the sub-adult survey over this period are not obvious. Furthermore, the standardised catch rates from a research longline survey of pre-recruit toothfish (70–110 cm TL) in the southern Ross Sea in 2012 were similar to those made by the same vessel fishing in the area earlier in the fishery, between 1999 and 2003 (Hanchet et al. 2012).

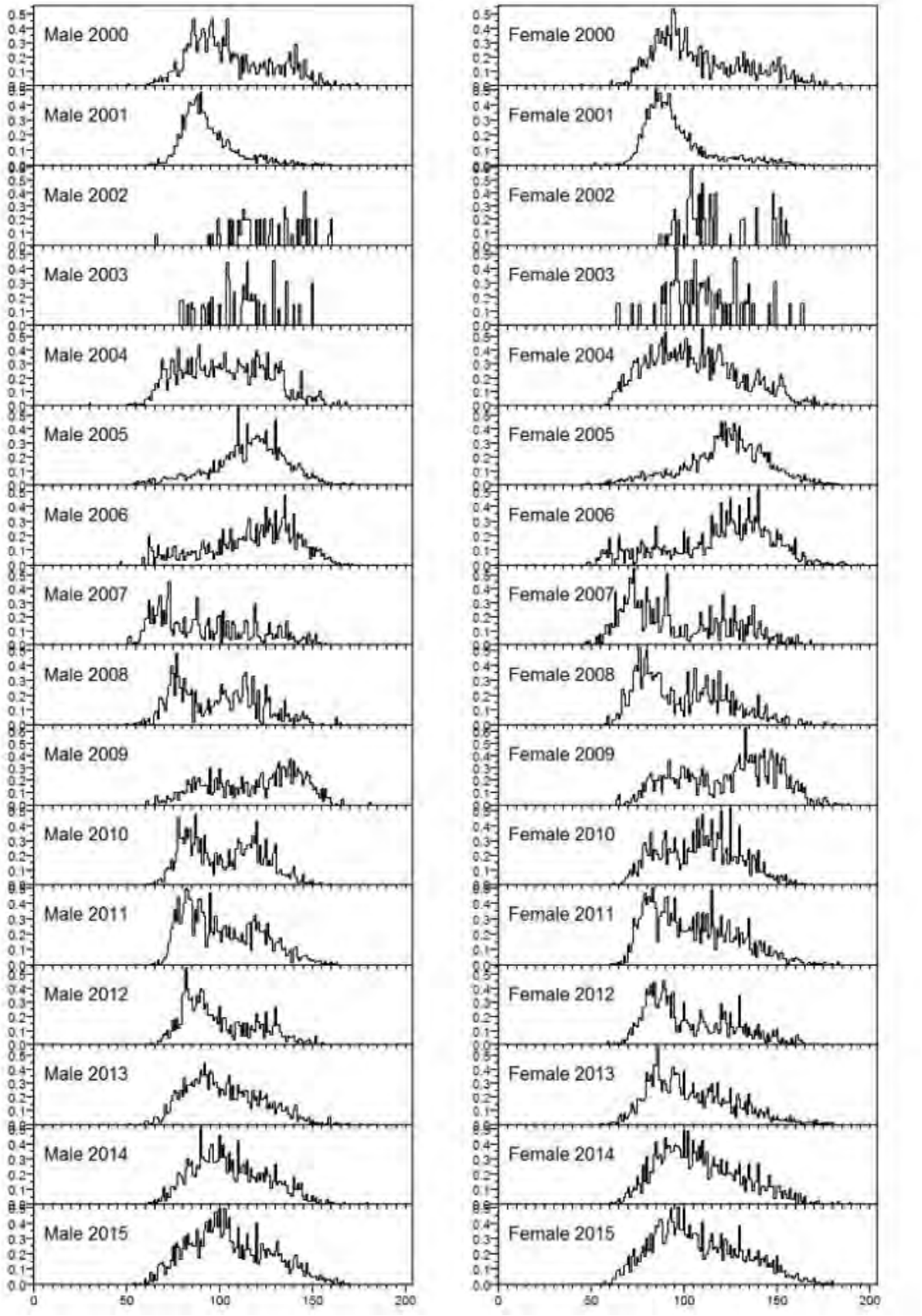


Figure 17.16: Estimated proportion of fish at length by sex for all vessels in the shelf region of the Ross Sea, for the years 2000–15 (Large et al. 2015).

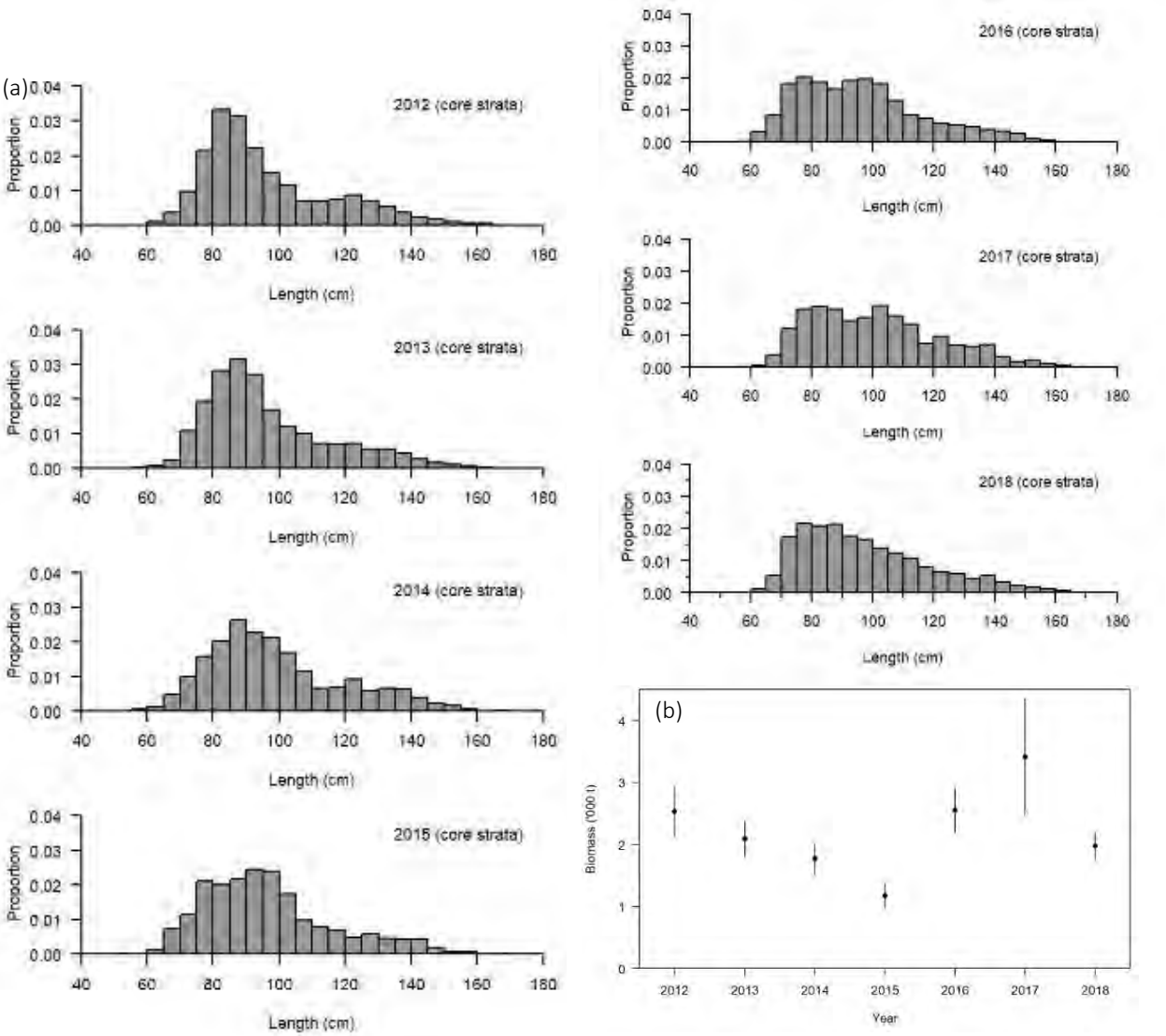


Figure 17.17: [a] Toothfish length frequency distributions in the Ross Sea shelf core strata (A–C) for the 2012–18 sub-adult surveys (Stevens et al. 2018). [b] Estimated biomass index for Antarctic toothfish in the core strata of the Ross Sea shelf survey 2012–2018. Error bars indicate the 95% confidence intervals (Stevens et al. 2018).

In the vicinity of McMurdo Sound, scientific droplining (through ice holes) had suggested large decreases in toothfish abundance since the 1970s (Ainley et al. 2013) but Parker et al. (2015) obtained catch rates of toothfish similar to those prior to the advent of the toothfish fishery (Figure 17.18). Results from Parker et al. (2015) suggest that either large old fish have returned to McMurdo Sound following a temporary environmentally driven absence, or that they remained locally present but were not detected in the areas sampled.

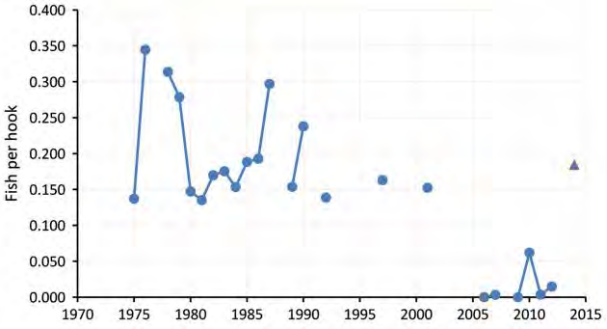


Figure 17.18: Catch rates (fish per hook) for toothfish sampled in McMurdo Sound, Antarctica, 1975–2014. Circles indicate pre-2013 data recalculated from Ainley et al. (2013) and triangle indicates the 2014 value from Parker et al. (2015). [Parker et al. (2015), figure 3].

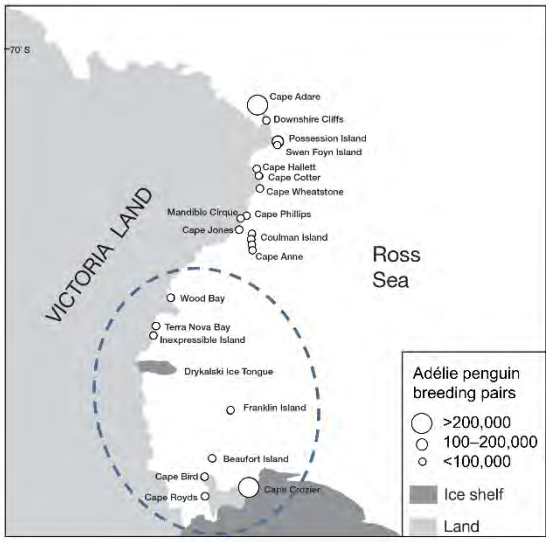
17.4.4 TROPHIC AND SYSTEM-LEVEL EFFECTS

The Ross Sea is home to about a third of the world population of Adélie penguins. Between 2001 and 2013 the number of breeding pairs of Adélie penguins at colonies in the southwestern Ross Sea more than doubled (Figure 17.19a,b) from about 235 000 to more than half a million (Lyver et al. 2014).

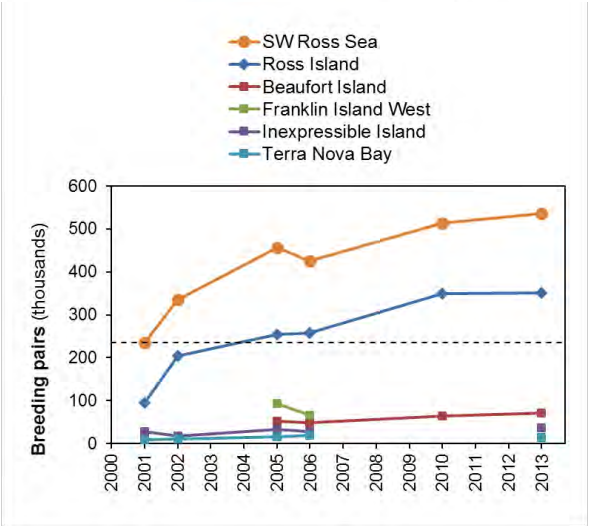
It is not known what has caused this increase but it is likely that changing ice patterns (Stammerjohn et al. 2008) play a primary role. Some researchers (Ainley et al. 2013, Lyver et al. 2014) previously suggested that reduced toothfish abundance in association with the Antarctic toothfish fishery had reduced predation on Antarctic silverfish and that the observed magnitude of the population response led to increases in the abundance of this species, which is known to be an important prey for Adélie penguins, especially during chick rearing. Other small fish are also taken by Adélie penguins. However, a predation release model of this effect acting via silverfish was not consistent with the magnitude of any plausible fishery-associated predation release. The mass of silverfish released from predation due to the effects of fishing was estimated to be equivalent to less than 2% of the biomass of silverfish estimated to be consumed annually by Adélie penguins (Pinkerton et al. 2016). Even if toothfish consumed only silverfish, the predicted predation release effect would still not be sufficient to explain the observed increase in the number of Adélie penguins in the southern Ross Sea (Pinkerton et al. 2016, Figure 17.19c).

The reasons for the increase in Adélie penguin numbers in the Ross Sea region are still not known. The fact that similar colony growth rates were seen for several Adélie penguin colonies in the south-west Ross Sea suggests that large-scale factors were responsible (Whitehead et al. 2015). The paucity of census data for the northern Ross Sea metapopulation makes it difficult to discern trends there (Lyver et al. 2014).

a



b



c

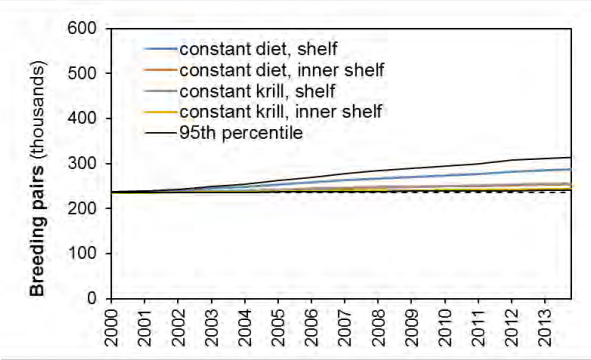


Figure 17.19: [a] Location and sizes of breeding colonies of Adélie penguins in the Ross Sea region. Those forming the 'southwest (SW) metapopulation' are enclosed in the dashed ellipse and forage over the Ross Sea shelf between chick hatching and fledging. [b] Changes to the total number of Adélie penguins breeding in the SW colonies (orange line) driven largely by increases in numbers breeding on Ross Island (blue line, Capes Crozier, Bird and Royds). [c] Greatest modelled effect of number of Adélie penguins that could be supported from additional silverfish released from predation by the toothfish fishery. [Pinkerton et al. 2016].

17.4.5 EFFECTS ON HABITATS

There are no indicators available to assess effects on the benthic habitat of fishing for toothfish in the Ross Sea or Amundsen Sea regions.

The status of selected habitat-forming benthic invertebrates likely to be physically impacted by fishing (vulnerable marine ecosystems, VMEs) was simulated (Dunn et al. 2010) under various scenarios of future fishing and assuming no correlation between distributions of VMEs and fishing. Predicted changes to the status of selected VMEs were small at the scale at the Ross Sea region, even with no specific management of VME impacts (Figure 17.20).

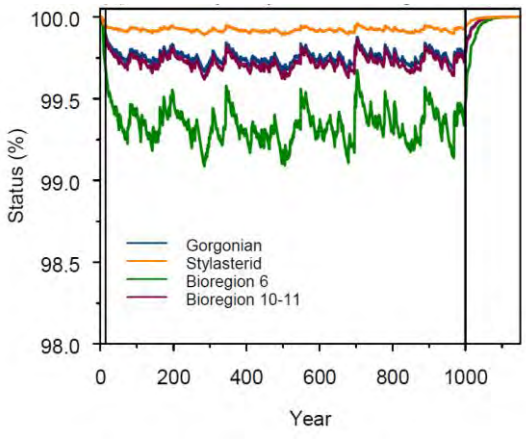


Figure 17.20: Simulated changes in the status of selected vulnerable marine ecosystems (VMEs) over time. Here, a status of 100% indicates that the habitat has the same extent and biomass as before fishing began and 0% indicates habitat removal at the scale of the Ross Sea region. Results are based on the medium-scale benthic habitat model of Dunn et al. (2010). The runs are for VMEs characterised as Gorgonian, Stylasterid, or for all VMEs in three indicative areas of the Ross Sea that have been identified as having different benthic biological conditions (*benthic bioregions*; Sharp et al. 2010). The benthic model assumes historical fishing pattern intensity up to 1000 years.

17.5 REFERENCES

Abe, T; Iwami, T (1989) Notes on fishes from the stomachs of whales in the Antarctic II: on *Dissostichus* and *Ceratias*. *Proceedings NIPR Symposium on Polar Biology 2*: 78–82.

Abrams P A (2014) How precautionary is the policy governing the Ross Sea Antarctic toothfish (*Dissostichus mawsoni*) fishery? *Antarctic Science 26*: 3–13.

Ackley, S F; Bengtson, J L; Boveng, P; Castellini, M; Daly, K L; Jacobs, S; Kooyman, G L; Laake, J; Quentin, L; Ross, G; Siniff, D B; Stewart, D B; Stirling, I ; Torres, J ; Yochem, P K (2003) A top-down, multidisciplinary study of the structure and function of the pack-ice ecosystem in the eastern Ross Sea, Antarctica. *Polar Record 39 (210)*: 219–230.

Ainley, D G (1985) Biomass of birds and mammals in the Ross Sea. *In*: Siegfried, W R; Condy, P R; Laws, R M (Eds.), Antarctic Nutrient Cycles and Food webs. pp 498–515. Springer-Verlag, Berlin.

Ainley, D G (2009) A history of the exploitation of the Ross Sea, Antarctica. *Polar Record 46 (238)*: 233–243.

Ainley, D G; Nur, N ; Eastman, J T; Ballard, G ; Parkinson, C L; Evans, C W; DeVries, A L (2013) Decadal trends in abundance, size and condition of Antarctic toothfish in McMurdo Sound, Antarctica, 1972–2011. *Fish and Fisheries 14 (3)*: 343–363.

Ainley, D G; Siniff, D B (2009) The importance of Antarctic toothfish as prey of Weddell Seals in the Ross Sea: a review. *Antarctic Science 21*: 317–327.

Arrigo, K R; Thomas, D (2004) Large scale importance of sea ice biology in the Southern Ocean. *Antarctic Science 16 (4)*: 471–486.

Arrigo, K R; van Dijken, G L (2004) Annual changes in sea ice, chlorophyll a, and primary production in the Ross Sea, Antarctica. *Deep Sea Research II 51 (1–3)*: 117–138.

Arrigo, K R; Worthen, D L; Lizotte, M P; Dixon, P; Dieckmann, G (1997) Primary production in Antarctic sea ice. *Science 276 (5311)*: 394–397.

Arrigo, K R; Worthen, D L; Robinson, D H (2003) A coupled ocean-ecosystem model of the Ross Sea: 2 Iron regulation of phytoplankton taxonomic variability and primary production. *Journal of Geophysical Research 108 (C7)*: 3231. doi:10.1029/2001JC000856.

Atkinson, D B; Rose, G A; Murphy, E F; Bishop, C A (1997) Distribution changes and abundance of northern cod (*Gadus morhua*), 1981–1993. *Canadian Journal of Fisheries and Aquatic Sciences 54 (Suppl. 1)*: 132–138.

Barrett-Lennard, L (2011) Killer Whale Evolution. *Journal of the American Cetacean Society 40 (1)*: 48–53.

Barrett-Lennard, L G; Ford, J K B; Heise, K A (1996) The mixed blessing of echolocation: Differences in sonar use by fish-eating and mammal-eating killer whales. *Animal Behaviour 51*: 553–565.

Battaglia, B; Valencia, J; Walton, D W H (Eds.) (1997) Antarctic communities: Species, structure and survival. Cambridge University Press, Cambridge. 464 p.

Beckage, B; Gross, L J; Kauffman, S (2011) The limits to prediction in ecological systems. *Ecosphere 2 (11)*: 125.

Bergstad, O A (1995) Age determination of deepwater fishes; experiences, status and challenges for the future. *In*: Hopper, A G (Ed.), Deep-water Fisheries of the North Atlantic Oceanic Slope. pp 267–283. Kluwer Academic Publishers, Netherlands.

Borer, E T; Seabloom, E W; Shurin, J B; Anderson, K E; Blanchette, C A; Broitman, B; Cooper, S D; Halpern, B S (2005) What

determines the strength of a trophic cascade? *Ecology 86 (2)*: 528–537.

Bradford-Grieve, J; Fenwick, G (2001) A review of the current knowledge describing biodiversity in the Ross Sea region. Report completed by NIWA for the Ministry of Fisheries project ZBD2000/01. 93 p.

Branch, T A (2006) Abundance estimates for Antarctic minke whales from three completed circumpolar sets of surveys, 1978/79 to 2003/04. International Whaling Commission Paper SC/58/IA18. 28 pp.

Brierley, A S; Kingsford, M J (2009) Impacts of climate change on marine organisms and ecosystems. *Current Biology 19*: R602–R614.

Brierley, A S; Thomas, D N (2002) Ecology of Southern Ocean pack ice. *Advances in Marine Biology 43*: 171–277.

CAMLR Convention (1980) Convention for the Conservation of Antarctic Marine Living Resources, Convention text. Retrieved from www.ccamlr.org/en/organisation/camlr-convention-text.

Caron, D A; Dennett, M R; Lonsdale, D J; Moran, D M; Shalaponok, L (2000) Microzooplankton herbivory in the Ross Sea, Antarctica. *Deepsea Research II 47 (15–16)*: 3249–3272.

CCAMLR (Secretariat) (2014) Catches in the Convention Area 2012/13 and 2013/14. CCAMLR document SC-CAMLR-XXXIII/BG/01, Hobart, Australia. 9 p.

CCAMLR (2017a) Fishery Report 2017: Exploratory fishery for *Dissostichus* spp. in Subarea 88.1. CCAMLR document, Hobart, Australia.

CCAMLR (2017b) Fishery Report 2017: Exploratory fishery for *Dissostichus* spp. in Subarea 88.2. CCAMLR document, Hobart, Australia.

CEMP (2004) CCAMLR Ecosystem Monitoring Program: Standard Methods. 268 p. Retrieved from http://archive.ccamlr.org/pu/E/e_pubs/std-meth04.pdf.

Chernova, N V; Eastman, J T (2001) Two new species of snailfish genus Paraliparis (Pisces:Liparidae) from the Ross Sea, Antarctica. *Journal of Fish Biology 59*: 92–104.

Christensen, V; Walters, C J (2004) Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling 172*: 109–139.

Christensen, V; Walters, C J; Pauly, D; Forrest, R (2008) Ecopath with Ecosim Version 6: user guide. Fisheries Centre, University of British Columbia, Vancouver, Canada.

Clark, M R; Dunn, M R; McMillan, P J; Pinkerton, M H; Stewart, A; Hanchet, S M (2010) Latitudinal variation of demersal fish assemblages in the western Ross Sea. *Antarctic Science 22 (6)*: 782–792.

Clarke, M R (1980) Cephalopoda in the diet of sperm whales of the Southern Hemisphere and their bearing on sperm whale biology. *Discovery Reports 37*: 1–324.

Comiso, J C (2003) Large-scale characteristics and variability of the global sea ice cover. *In*: Thomas, D N; Dieckmann, G S (Eds.), Sea Ice: An Introduction to its Physics, Chemistry, Biology and Geology pp 112–142. Blackwell Science, Oxford, UK.

Constable, A J; de la Mare, W K; Agnew, D J; Everson, I; Miller, D (2000) Managing fisheries to conserve the Antarctic marine ecosystem: practical implementation of the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR). *ICES Journal of Marine Science 57*: 778–791.

Delegations of New Zealand and the USA (2014) Chronology of previously submitted scientific documents, and updated maps and analyses supporting MPA planning in the Ross Sea region. CCAMLR document SC-CAMLR-XXXIII/BG/23, Hobart, Australia.

Delegations of New Zealand, Norway and the United Kingdom (2014) Medium-term research plan for the Ross Sea toothfish fishery. CCAMLR document WG-FSA-14/60, Hobart, Australia. 27 p.

DeWitt, H H (1970) The character of the midwater fish fauna of the Ross Sea, Antarctica. *In*: Holdgate, M W (Ed.), Antarctic Ecology. Volume 1. pp 305–314. Academic Press, New York.

Dunn, A; Hanchet, S M; Ballara, S L; Francis, M P (2007) Preliminary investigations of an assessment model for skates and rays in the Ross Sea. CCAMLR document WG-SAM-07/4, Hobart, Australia.

Dunn, A; Jones, C; Mormede, S; Parker, S (2016) Results of the fifth Ross Sea shelf survey to monitor abundance of sub-adult Antarctic toothfish in the southern Ross Sea, February 2016, and notification for continuation in 2017. CCAMLR document WG-SAM-16/14, Hobart, Australia.

Dunn, A; Parker S J; Mormede S (2010) Development of methods for evaluating the management of benthic impacts from longline fishing using spatially explicit production models. CCAMLR document WG-SAM-10/19, Hobart, Australia.

Dunn, A; Vacchi, M; Watters, G (2017) The Ross Sea region Marine Protected Area Research and Monitoring Plan. CCAMLR document SC-CAMLR-XXXVI/20, Hobart, Australia.

Eastman, J T (1985) *Pleuragramma antarcticum* (Pisces, Nototheniidae) as food for other fishes in McMurdo Sound, Antarctica. *Polar Biology 4*: 155–160.

Eastman, J T; Hubold, G (1999) The fish fauna of the Ross Sea, Antarctica. *Antarctic Science 11 (3)*: 293–304.

Eisert, R; Lauriano, G; Panigada, S; Ovsyanikova, E N; Visser, I N; Ensor, P H; Currey, R J C; Sharp, B R; Pinkerton, M H (2015) Activity, seasonal site fidelity, and movements of Type-C killer whales between the Ross Sea, Antarctica and New Zealand. CCAMLR document WG-EMM-15/52, Hobart, Australia.

Eisert, R; Pinkerton, M H; Newsome, S D; Oftedal, O T (2013) A critical re-examination of the evidence for a possible dependence of Weddell seals (*Leptonychotes weddellii*) on Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea, Antarctica. CCAMLR document WG-EMM-13/28, Hobart, Australia.

Eisert, R; Pinkerton, M H; Torres, L; Currey, R J C; Ensor, P H; Ovsyanikova, E N; Visser, I N; Oftedal, O T (2014) Update on the Top Predator Alliance project, 2013-14 season: Killer whales. CCAMLR document WG-EMM-14/52, Hobart, Australia.

Endicott, M; Agnew, D J (2004) The survivorship of rays discarded from the South Georgia longline fishery (Short note). *CCAMLR Science 11*: 155–164.

Evans, K; Hindell, M A (2004) The diet of sperm whales (*Physeter macrocephalus*), in southern Australian waters. *ICES Journal of Marine Science 61 (8)*: 1313–1329.

FAO (2008) United Nations Food and Agriculture Organization Code of Conduct for Responsible Fisheries. *Fisheries: Technical Guidelines 4 (supp.2, add.1)*. Rome, FAO. 78 p.

Fenaughty, J (2008) The autoline system – an updated descriptive review of the method with recommendations to clarify CCAMLR Conservation Measures regulating longline fisheries within the Convention Area. CCAMLR document WG-FSA-08/60, Hobart, Australia.

Fenaughty, J M; Stevens, D W; Hanchet, S M (2003) Diet of the Antarctic toothfish (*Dissostichus mawsoni*) from the Ross Sea, Antarctica (CCAMLR statistical subarea 88.1). *CCAMLR Science 10*: 113–123.

Fisher, J A D; Frank, K T (2004) Abundance-distribution relationships and conservation of exploited marine fishes. *Marine Ecology Progress Series 279*: 201–213.

Fisheries New Zealand (2018) Fisheries Assessment Plenary, May 2018: stock assessments and stock status. Compiled by the Fisheries Science Groups, Fisheries New Zealand, Wellington, New Zealand. 1660 p.

Fitzcharles, E M (2014) Genetic diversity of Antarctic fish. PhD Thesis, University of St Andrews, Scotland. 199 p.

Fletcher, W J (2005) The application of qualitative risk assessment methodology to prioritize issues for fisheries management. *ICES Journal of Marine Science 62 (8)*: 1576–1587.

Fletcher, W J; Chesson, J; Fisher, M; Sainsbury, K J; Hundloe, T; Smith, A D M; Whitworth, B (2002) National ESD reporting framework for Australian Fisheries: The how to guide for wild capture fisheries. FRDC Report 2000/145, Canberra, Australia.

Frank, K T; Petrie, B; Choi, J S; Leggett, W C (2005) Trophic cascades in a formerly cod-dominated ecosystem. *Science 308*: 1621–1623.

Garrison, D L (1991) Antarctic sea ice biota. *American Zoologist 31*: 17–33.

Goetz, K (2015) Movement, Habitat, and Foraging Behavior of Weddell Seals (*Leptonychotes weddellii*) in the Western Ross Sea, Antarctica. PhD Thesis, University of California Santa Cruz. Retrieved from http://escholarship.org/uc/item/0jx2107r.

Gon, O; Heemstra, P C (Eds.) (1990) Fishes of the Southern Ocean. JBL Smith Institute of Ichthyology, Grahamstown, South Africa. 462 p.

Halpern, B S; Walbridge, S; Selkoe, K A; Kappel, C B; Micheli, F; D'Agrosa, C; Bruno, J F; Casey, K S; Ebert, C; Fox, H E; Fujita, R; Heinemann, D; Lenihan, H S; Madin, E M P; Perry, M T; Selig, E R; Spalding, M; Steneck, R; Watson R (2008) A global map of human impact on marine ecosystems. *Science 319*: 948–951.

Hanchet, S M (2010) Updated species profile for Antarctic toothfish (*Dissostichus mawsoni*). CCAMLR WG-FSA-10/24.

Hanchet, S; Dunn, A; Parker, S; Horn, P; Stevens, D; Mormede, S (2015a) The Antarctic toothfish (*Dissostichus mawsoni*): biology, ecology, and life history in the Ross Sea region. *Hydrobiologia 761 (1)*: 397–414.

Hanchet, S M; Mormede, S; Parker, S J; Dunn, A (2012) Results of a CCAMLR-sponsored research survey to monitor abundance of pre-recruit Antarctic toothfish in the southern Ross Sea, February 2012. CCAMLR document WG-FSA-12/41, Hobart, Australia.

Hanchet, S M; Rickard, G J; Fenaughty, J M; Dunn, A; Williams, M J (2008) A hypothetical lifecycle for Antarctic toothfish *Dissostichus mawsoni* in Antarctic waters of CCAMLR Statistical Area 88. *CCAMLR Science 15*: 35–54.

Hanchet, S; Sainsbury, K; Butterworth, D S; Darby, C; Bizikov, V; Rune Godo, O; Ichii, T; Kock, K-H; Lopez, Abellan, L; Vacchi, M (2014) CCAMLR's precautionary approach to management of Ross Sea toothfish fishery. *Antarctic Science 27 (4)*: 333–340.

Hanchet, S M; Sharp, B R; Mormede, S; Parker, S J; Vacchi, M (2015b) Results of the fourth CCAMLR sponsored research survey to monitor abundance of sub-adult Antarctic toothfish in the southern Ross Sea, February 2015 and further development of the time series. CCAMLR document WG-SAM-15, Hobart, Australia.

Harper, P C; Knox, G A; Spurr, E B; Taylor, R H; Wilson, G J; Young, E C (1984) The status and conservation of birds in the Ross Sea sector of Antarctica. *In*: Croxall J P; Evans P G H; Schreiber R W (Eds.), Status and Conservation of the World's Seabirds. International Council for Bird Preservation Technical Publication No. 2.

Heithaus, M R; Frid, A; Wirsing, A J; Worm, B (2008) Predicting the consequences of declines in marine top predators. *Trends in Ecology and Evolution 23 (4)*: 202–210.

Heupel, M R; Knip, D M; Simpfendorfer, C A; Dulvy, N K (2014) Sizing up the ecological role of sharks as predators. *Marine Ecology Progress Series 495*: 291–298.

Hubold, G (1985) The early life-history of the high-Antarctic silverfish *Pleuragramma antarcticum*. *In*: Siegfried, W R; Condy, P R; Laws, R M (Eds.), Antarctic Nutrient Cycles and Food Webs. pp 445–451. Springer-Verlag, Berlin, Germany.

Hutchings, J A (1996) Spatial and temporal variation in the density of northern cod and a review of hypotheses for the stock's collapse. *Canadian Journal of Fisheries and Aquatic Sciences 53*: 943–962.

Hutchings, J A (2000) Collapse and recovery of marine fishes. *Nature 406*: 882–885.

ICES (2005) Ecosystem effects of fishing: Impacts, metrics, and management strategies. *ICES Cooperative Research Report No. 272*. Rice, J C (Ed.). 177 p.

International Whaling Commission (2001) Report of the sub-committee on the comprehensive assessment of whale stocks-other stocks. *Journal of Cetacean Research and Management (Special Issue) 3*: 209–228.

IPCC (2014) Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Pachauri, R K; Meyer, L A (Eds.). IPCC, Geneva, Switzerland. 151 p.

Jacobs, S S; Comiso, J C (1989) Satellite passive microwave sea ice observations and oceanic processes in the Ross Sea, Antarctica. *Journal of Geophysical Research 94*: 195–211.

Kasamatsu, F; Joyce, G G (1995) Current status of Odontocetes in the Antarctic. *Antarctic Science 7 (4)*: 365–379.

Keith, D M; Hutchings, J A (2012) Population dynamics of marine fishes at low abundance. *Canadian Journal of Fisheries and Aquatic Sciences 69*: 1150–1163.

Kelly, C J; Connolly, P L; Bracken, J J (1997) Age estimation, growth, maturity and distribution of the roundnose grenadier from the Rockall Trough. *Journal of Fish Biology 50*: 1–17.

Kim, S Z; Ainley, D G; Pennycook, J; Eastman, J T (2011) Antarctic toothfish heads found along tide cracks of the McMurdo Ice Shelf. *Antarctic Science 23 (5)*: 469–470. doi:10.1017/S095410201100040X.

Kirby, R R; Beaugrand, G; Lindley, J A (2009) Synergistic effects of climate and fishing in a marine ecosystem. *Ecosystems 12*: 548–561.

Knox, G A (2007) Biology of the Southern Ocean. 2nd edition. Boca Raton, CRC Press. 621 p.

Kock, K-H (Ed.) (2000) Understanding CCAMLR's Approach to Management. Retrieved from www.ccamlr.org/en/document/publications/understanding-ccamlr%E2%80%99s-approach-management.

Kooyman, G L; Mullins, J L (1990) Ross Sea emperor penguin breeding populations estimated by aerial photography. *In*: Kerry, K R; Hempel, G (Eds.), Antarctic ecosystems: ecological change and conservation. Springer-Verlag, Berlin-Heidelberg.

La Mesa, M; Eastman, J T; Vacchi, M (2004) The role of notothenioid fish in the food web of the Ross Sea shelf waters: a review. *Polar Biology 27*: 321–338.

Ladroit, Y; O'Driscoll, R L; Mormede, S (2014) Using acoustic echo counting to estimate grenadier abundance in the Ross Sea (SSRU88.1I). CCAMLR document WG-FSA-14/62, Hobart, Australia.

Large, K; Hanchet, S M; Mormede, S (2015) A characterisation of the toothfish fishery in Subareas 88.1 and 88.2 from 1997–98 to 2014–15. CCAMLR document WG-FSA-15/36, Hobart, Australia.

Large, K; Parker, S; Hanchet, S (2016) A characterisation of the toothfish fishery and tagging programme in the Amundsen Sea region (SSRUs 88.2C–H) through 2015–16. CCAMLR document WG-FSA-16/45, Hobart, Australia.

Large, K; Robinson, L; Parker, S (2017) Results of the sixth Ross Sea shelf survey to monitor abundance of sub-adult Antarctic toothfish in the southern Ross Sea, January 2017. CCAMLR document WG-SAM-17/01, Buenos Aires, Argentina.

Laws, R M (1984) (Ed.) Antarctic Ecology. Volume 2. Academic Press, London. 850 p.

Lockyer, C H; Brown, S G (1981) The migration of whales. *In*: Aidley, D J (Ed.), Animal Migration. pp 105–137. Cambridge University Press, Cambridge.

Lyver, P O; Barron, M; Barton, K J; Ainley, D G; Pollard, A; Gordon, S; McNeill, S; Ballard, G; Wilson, P R (2014) Trends in the breeding population of Adélie penguins in the Ross Sea, 1981–2012: A coincidence of climate and resource extraction effects. *PLoS ONE 9 (3)*: e91188. doi:10.1371/journal.pone.0091188.

McCann, K; Hastings, A; Huxel, G R (1998) Weak trophic interactions and the balance of nature. *Nature 395*: 794–798.

McCully, S R; Goldsmith, D; Burt, G; Scott, R; Ellis, J R (2013) An overview of tagging skates (Rajiformes) and CCAMLR skate tagging data. CCAMLR document WG-SAM-13/25r1, Hobart, Australia.

McMillan, P J; Iwamoto, T; Stewart, A; Smith, P J (2012) A new species of grenadier, genus Macrourus (Teleostei, Gadiformes, Macrouridae) from the southern hemisphere and a revision of the genus. *Zootaxa 3165*: 1–24.

Mormede, S (2017) Assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea region to 2016/17. CCAMLR document WG-FSA-17/37, Hobart, Australia.

Mormede, S; Dunn, A (2010) Characterisation of skate catches in the Ross Sea region. CCAMLR document WG-FSA-10/25, Hobart, Australia.

Mormede, S; Dunn, A; Hanchet, S M (2014b) A stock assessment model of Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea region incorporating multi-year mark-recapture data. *CCAMLR Science 21*: 39–62.

Mormede, S; Dunn, A; Hanchet, S M (2015) Assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea region for the years 1997–98 to 2014–15. CCAMLR document WG-FSA-15/38, Hobart, Australia.

Mormede, S; Dunn, A; Hanchet, S; Parker, S (2014c) Spatially explicit population dynamics models for Antarctic toothfish in the Ross Sea region. *CCAMLR Science 21*: 19–37.

Mormede, S; Large, K; Hanchet, S (2016) Progress towards an assessment of Antarctic toothfish (*Dissostichus mawsoni*) in Subarea 88.2

SSRUs 882C–H for the years 2002–03 to 2015–16 using a two-area model. CCAMLR document WG-FSA-16/44, Hobart, Australia.

Mormede, S; Parker, S J (2018a) Summary of the toothfish fishery and tagging programme in the Ross Sea region (Subarea 88.1 and SSRUs 88.2A-B) through 2017-18. CCAMLR document WG-FSA-18/46, Hobart, Australia.

Mormede, S; Parker, S J (2018b) Summary of the toothfish fishery and tagging programme in the Amundsen Sea region (SSRUs 882C-H.) to 2017/18. CCAMLR document WG-FSA-18/36, Hobart, Australia.

Mormede, S; Parker, S J; Hanchet, S M; Dunn, A; Gregory, S (2014d) Results of the third CCAMLR sponsored research survey to monitor abundance of sub-adult Antarctic toothfish in the southern Ross Sea, February 2014 and development of the time series. CCAMLR document WG-FSA-14/51, Hobart, Australia.

Mormede, S; Pinkerton, M; Dunn, A; Hanchet, S; Parker, S (2014e) Development of a spatially-explicit minimum realistic model for Antarctic toothfish (*Dissostichus mawsoni*) and its main prey (Macrouridae and Channichthyidae) in the Ross Sea. CCAMLR document WG-EMM-14/51, Hobart, Australia.

Murawski, S A (2000) Definitions of overfishing from an ecosystem perspective. *ICES Journal of Marine Science 57*: 649–658.

Nelson, J S (2006) Fishes of the world. 4th edition. John Wiley & Sons, Hoboken. 601 p.

O'Driscoll, R L; Hanchet, S M; Miller, B S (2012) Can acoustic methods be used to monitor grenadier (Macrouridae) abundance in the Ross Sea region? *Journal of Ichthyology 52 (10)*: 700–708.

O'Driscoll, R L; Hanchet, S M; Wood, B A (2005) Approaches to monitoring and assessing the abundance of rattails (*Macrourus* spp) and skates in the Ross Sea. CCAMLR document WG-FSA-05/22, Hobart, Australia.

Ohizumi, H; Isoda, T; Kishiro, T; Kato, H (2003) Feeding habits of Baird's beaked whale *Berardius bairdii*, in the western North Pacific and Sea of Okhotsk off Japan. *Fisheries Science 69*: 11–20.

Okutani, T (1995) Cuttlefish and squids of the world in color. Publication for the 30th anniversary of the foundation of the National Cooperative Association of Squid Processors. 185 p.

Pace, M L; Cole, J J; Carpenter, S R; Kitchell, J F (1999) Trophic cascades revealed in diverse ecosystems. *Trends in Ecology and Evolution 14 (12)*: 483–488.

Parker, S J (2014) Analysis of seamount-specific catch and tagging data in the Amundsen Sea, SSRU 88.2H. CCAMLR document WG-SAM-14/27, Hobart, Australia.

Parker, S J; Bowden, D A (2009) Identifying taxonomic groups as vulnerable to bottom longline fishing gear in the Ross Sea region. CCAMLR document WS-VME-09/8, Hobart, Australia.

Parker, S J; Cole, R G; Hanchet, S M (2010) Further analysis of spatial patterns of benthic invertebrate habitats from fishery bycatch in the Ross Sea region. CCAMLR document WG-FSA-10/30, Hobart, Australia.

Parker, S J; Hanchet, S M; Horn, P L (2014) Stock structure of Antarctic toothfish in Statistical Area 88 and implications for assessment and management. CCAMLR document WG-SAM-14/26, Hobart, Australia.

Parker, S; Hanchet, S M; Mormede, S; Dunn, A; Sarralde, R (2013b) Results of a CCAMLR sponsored research survey to monitor abundance of sub-adult Antarctic toothfish in the southern Ross Sea, February 2013. CCAMLR document WG-SAM-13/32, Hobart, Australia.

Parker, S; McMillan, P; Marriott, P (2012) Characterisation of Muraenolepis species bycatch in the CCAMLR Convention Area. CCAMLR document WG-FSA-12/50, Hobart, Australia.

Parker, S J; Marriott, P (2012) Indexing maturation of Antarctic toothfish *Dissostichus mawsoni* in the Ross Sea region. *CCAMLR Science 17*: 53–73.

Parker, S J; Mormede, S (2009) Is the bycatch of vulnerable invertebrate taxa associated with high catch rates of fish in the Ross Sea longline fisheries? CCAMLR document WS-VME-09/7, Hobart, Australia.

Parker, S J; Mormede S; Devries A L; Hanchet S M; Eisert R (2015) Have Antarctic toothfish returned to McMurdo Sound? *Antarctic Science 28 (1)*: 29–34. doi:10.1017/S0954102015000450.

Parker, S J; Smith, M H (2010) Is toothfish catch correlated with the catch of vulnerable benthic invertebrate taxa? CCAMLR document WG-EMM 10/27, Hobart, Australia. 11 p.

Parker, S; Tracey, D; Mackay, E; Mills, S; Marriott, P; Anderson, O; Schnabel, K; Bowden, D; Kelly, M (2008) Classification guide for potentially vulnerable invertebrate taxa in the Ross Sea longline fishery. CCAMLR document WG-FSA-08/19, Hobart, Australia.

Parkinson, C L (2002) Trends in the length of the Southern Ocean sea ice season. *Annals of Glaciology 34*: 435–440.

Perrin, W F; Wursig, B; Thewissen, J G M (Eds.) (2002) Encyclopedia of Marine Mammals. Academic Press.

Perry, R I; Cury, P; Brander, K; Jennings, S; Mollmann, C; Planque, B (2010) Sensitivity of marine systems to climate and fishing: concepts, issues and management responses. *Journal of Marine Systems 79*: 427–435.

Pinkerton, M H; Boyd, P; Bradford-Grieve, J (2011) Ecosystem modelling, validation, and network characteristics of the Ross Sea, Antarctica: Past, Present and Future. 8th Symposium on Polar Studies, Palma de Mallorca, Spain.

Pinkerton, M H; Bradford-Grieve, J M (2014) Characterizing foodweb structure to identify potential ecosystem effects of fishing in the Ross Sea, Antarctica. *ICES Journal of Marine Science 71 (7)*: 1542–1553.

Pinkerton, M H; Bradford-Grieve, J M; Hanchet, S M (2010a) A balanced model of the food web of the Ross Sea, Antarctica. *CCAMLR Science 17*: 1–31.

Pinkerton, M H; Bradford-Grieve, J; Sagar, P M (2010b) Whales: Trophic modelling of the Ross Sea. Supporting information for Pinkerton et al. (2010a). Retrieved from https://www.niwa.co.nz/sites/niwa.co.nz/files/import/attachments/04_Whales_CCAMLR_Apr10.pdf.

Pinkerton, M H; Bury, S J; Brown, J C S; Forman, J; Kilimnik, A (2014) Stable isotope analysis of tissue samples to investigate trophic linkages of Antarctic toothfish (*Dissostichus mawsoni*) in the Ross and Amundsen Sea regions. CCAMLR document WG-EMM-14/50, Hobart, Australia.

Pinkerton, M H; Dunn, A; Hanchet, S M (2008) Trophic overlap of Weddell seals (*Leptonychotes weddelli*) and Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea, Antarctica. CCAMLR document WG-EMM-08/43, Hobart, Australia.

Pinkerton, M H; Lyver, P O'B; Stevens, D W; Forman, J; Eisert, R; Mormede, S (2016) Increases in Adélie penguins in the Ross Sea: could the fishery for Antarctic toothfish be responsible? *Ecological Modelling 337*: 262–271.

Pinkerton, M H; McMillan, P; Forman, J; Marriott, P; Horn, P; Bury, S; Brown, J (2013) Distribution, morphology and ecology of *Macrourus whitsoni* and *M caml* (Gadiformes, Macrouridae) in the Ross Sea region. *CCAMLR Science 20*: 37–61.

Pinkerton, M H; Sharp, B (2012) Preliminary plan for research and monitoring in the Ross Sea region, in association with spatial marine protection. CCAMLR document WG-EMM-12/57, Hobart, Australia.

Pitman, R L (2003) Good whale hunting. *Natural History*: 24–28.

Pitman, R L; Balance, L T; Mesnick, S L; Chivers, S (2001) Killer whale predation on sperm whales: observations and implications. *Marine Mammal Science 17 (3)*: 494–507.

Pitman, R L; Ensor, P (2003) Three different forms of killer whales in Antarctic waters. *Journal of Cetacean Research and Management 5 (2)*: 131–139.

Plagányi, E E (2007) Models for an ecosystem approach to fisheries. *FAO (Food and Agriculture Organization of the United Nations) Fisheries Technical Paper 477*.

Plagányi, E E; Punt, A E; Hillary, R, et al. (2014) Multispecies fisheries management and conservation, tactical applications using models of intermediate complexity. *Fish and Fisheries 15 (1)*: 1–22.

Planque, B (2015) Projecting the future state of marine ecosystems, 'la grande illusion'? *ICES Journal of Marine Science 73 (2)*: 204–208. doi:10.1093/icesjms/fsv155.

Prugh, L R; Stoner, C J; Epps, C W; Bean, W T; Ripple, W J; LaLiberte, A S; Brashares, J S (2009) The rise of the mesopredator. *BioScience 59 (9)*: 779–791.

Reid, E; Sullivan, B; Clark, J (2010) Mitigation of seabird captures during hauling in CCAMLR longline fisheries. *CCAMLR Science 17*: 155–162.

Reynolds, J D; Dulvy, N K; Goodwin, N B; Hutchings, J A (2005) Biology of extinction risk in marine fishes. *Proceedings of the Royal Society B 272*: 2337–2344.

Rice, D W (1998) Marine mammals of the world: systematics and distribution. *Society for Marine Mammalogy, Special Publication Number 4*. Wartzok, D (Ed.). Lawrence, KS, USA.

Rice, J C; Garcia, S M (2011) Fisheries, food security, climate change, and biodiversity: characteristics of the sector and perspectives on emerging issues. *ICES Journal of Marine Science 68 (6)*: 1343–1353.

Rodhouse, P G K (2013) Role of squid in the Southern Ocean pelagic ecosystem and the possible consequences of climate change. *Deep Sea Research II 95*: 129–138.

Rogers, A D; Clark, M R; Hall-Spencer, J M; Gjerde, K M (2008) The Science behind the Guidelines: A Scientific Guide to the FAO Draft International Guidelines (December 2007) for the Management of Deepsea Fisheries in the High Seas and Examples of How the Guidelines may be Practically Implemented. IUCN, Switzerland. 39 p.

Rose, K A; Allen, J I; Artioli, Y, et al. (2010) End-to-end models for the analysis of marine ecosystems, challenges, issues, and next steps. *Marine and Coastal Fisheries 2 (1)*: 115–130.

Sedwick, P N; DeTullio, G R; Mackey, D J (2000) Iron and manganese in the Ross Sea, Antarctica: Seasonal iron limitation in Antarctic shelf waters. *Journal of Geophysical Research 105 (C5)*: 11321–11336.

Sharp, B R (2010) Updated impact assessment framework to estimate the cumulative footprint and impact on VME taxa of bottom longline fisheries in the CCAMLR Area. CCAMR document WG-FSA-10/31, Hobart, Australia.

Sharp, B R; Parker, S J; Pinkerton, M H; Breen, B B; Cummings, V; Dunn, A; Grant, S M; Hanchet, S M; Keys, H J R; Lockhart, S J; Lyver, P O'B; O'Driscoll, R L; Williams, M J M; Wilson, P R (2010) Bioregionalisation and spatial ecosystem processes in the Ross Sea region. CCAMLR document WG-EMM-10/30, Hobart, Australia.

Sharp, B R; Parker, S J; Smith, N (2009) An impact assessment framework for bottom fishing methods in the CAMLR convention area. *CCAMLR Science 16*: 195–210.

Sharp, B R; Watters, G M (2011) Marine Protected Area planning by New Zealand and the United States in the Ross Sea region. CCAMLR document WS-MPA-11/25, Hobart, Australia.

Shurin, J B; Borer, E T; Seabloom, E W; Anderson, K; Blanchette, C A; Broitman, B R; Cooper, S D; Halpern, B (2002) A cross-ecosystem comparison of the strength of trophic cascades. *Ecology Letters 5*: 785–791.

Smith P J; Steinke D; McMillan P J; Stewart A L; McVeagh S M; Diaz De Astarloa J M; Welsford D; Ward R D (2011) DNA barcoding highlights a cryptic species of grenadier (genus *Macrourus*) in the Southern Ocean. *Journal of Fish Biology 78*: 355–365.

Smith, Jr W O; Ainley, D G; Cattaneo-Vietti, R; Hofmann, E E (2012) The Ross Sea continental shelf: regional biogeochemical cycles, trophic interactions, and potential future changes. *In*: Rogers, A D; Johnston, N M; Murphy, E J; Clarke, A (Eds.), Antarctic Ecosystems: An Extreme Environment in a Changing World. John Wiley and Sons Ltd, Chichester, UK.

Smith, W O; Ainley, D G; Cattaneo-Vietti, R (2007) Marine ecosystems: the Ross Sea. *Philosophical Transactions of the Royal Society of London B Biological Sciences 362 (1477)*: 149–166.

Smith, W O; Dinniman, M S; Klinck, J M; Hofmann, E (2003) Biogeochemical climatologies in the Ross Sea, Antarctica: seasonal patterns of nutrients and biomass. *Deep Sea Research II 50 (22–26)*: 3083–3101.

Soulé, M E; Bolger, D T; Alberts, A C; Wright, J; Sorice, M; Hill, S (1988) Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. *Conservation Biology 2*: 75–91.

Stammerjohn, S E; Martinson, D T; Smith, R C; Yuan, X; Rind, D (2008) Trends in Antarctic annual sea ice retreat and advance and their relation to El Nino–Southern Oscillation and Southern Annular Mode variability. *Trends in Geophysical Research 113*: C03S90.

Steele, J H; Schumacher, M (2000) Ecosystem structure before fishing. *Fisheries Research 44*: 201–205.

Stevens, D; Di Blasi, D; Parker, S (2016) Summary of the results of the first winter longline survey to the northern Ross Sea region to investigate toothfish reproductive life history. CCAMLR document WG-FSA-16/XX, Hobart, Australia.

Stevens, D W; Dunn, M R; Pinkerton, M H; Forman, J S (2014) Diet of Antarctic toothfish (*Dissostichus mawsoni*) from the continental slope and oceanic features of the Ross Sea region, Antarctica. *Antarctic Science 26 (5)*: 502–512.

Stevens, D; Fu, X; Mormede, S; Parker, S (2018) Results of the seventh Ross Sea shelf survey to monitor abundance of Antarctic toothfish in the southern Ross Sea, January 2018, and notification for research in 2018/19. CCAMLR document WG-SAM-18/10, Hobart, Australia.

Stevenson, M L; Hanchet, S M; Mormede, S; Dunn, A (2012) A characterisation of the toothfish fishery in Subareas 88.1 and 88.2 from 1997/98 to 2011/12. CCAMLR document WG-FSA-12/42, Hobart, Australia.

Stewart, A L; Roberts, C D (2001) Fishes collected during the Ross sea exploratory fishery (88.1) in 1999-00 and 2000-01, and registered in the national fish collection at the Museum of New Zealand Te Papa Tongarewa. CCAMLR document WG-FSA-01/45, Hobart, Australia.

Stewart, B S; Yochem, P K; Gelatt, T S; Siniff, D B (2003) The pack ice niche of Weddell seals in the western Ross Sea. *In*: Huiskies, A; Gieskes, W W; Rozema, J; Schornu, R M L; van der Vies, S M; Wolff, W J (Eds.), Antarctic Biology in a Global Context. pp 224–229. Backhuys Publishers, Leiden.

Stirling, I (1969) Ecology of the Weddell seal in McMurdo Sound, Antarctica. *Ecology 50*: 533–544.

Stirling, I (1971) Population aspects of Weddell seal harvesting at McMurdo Sound, Antarctica. *Polar Record 15 (98)*: 652–667.

Sutton, C P; Manning, M J; Stevens, D W; Marriott, P M (2008) Biological parameters for icefish (*Chionobathyscus dewitti*) in the Ross Sea, Antarctica. *CCAMLR Science 15*: 139–166.

Swain, D P; Sinclair, A F (1993) Fish distribution and catchability: What is the appropriate measure of distribution? *Canadian Journal of Fisheries and Aquatic Sciences 51*: 1046–1054.

Tarling, G A; Shreeve, R S; Ward, P; Atkinson, A; Hirst, A G (2004) Lifecycle phenotypic composition and mortality of *Calanoides acutus* (Copepoda: Calonoida) in the Scotia Sea: a modelling approach. *Marine Ecology Progress Series 272*: 165–181.

Thomas, D N; Dieckmann, G S (2002) Antarctic sea ice – a habitat for extremophiles. *Science 295*: 641–644.

Thompson, D R; Pinkerton, M H; Stevens, D W; Cherel, Y; Bury, S J (2012) The Ross Sea cephalopod community: insights from stable isotope analysis. CCAMLR document WG-EMM-12/55, Hobart, Australia.

Torres, L; Pinkerton, M H; Pitman, R; Durban, J; Eisert R (2013) To what extent do type C killer whales (*Orcinus orca*) feed on Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea, Antarctica? CCAMLR document WG-EMM-13/29, Hobart, Australia.

Ulanowicz, R E; Puccia, C J (1990) Mixed trophic impacts in ecosystems. *Coenoses 5*: 7–16.

Umani, S F; Monti, M; Nuccio, C (1998) Microzooplankton biomass distribution in Terra Nova Bay, Ross Sea (Antarctica). *Journal of Marine Systems 17 (1–4)*: 289–303.

Vacchi, M; Pisano, E; Ghigliotti, L (Eds.) (2017) The Antarctic silverfish, a keystone species in a changing ecosystem. Advances in Polar Ecology Series, Springer. 314 p.

Valdes, L; Peterson, W; Church, J; Brander, K; Marcos, M (2009) Our changing oceans: conclusions of the first international symposium on the effects of climate change on the world's oceans. *ICES Journal of Marine Science 66*: 1435–1438.

Voronina, N M (1998) Comparative abundance and distribution of major filter-feeders in the Antarctic pelagic zone. *Journal of Marine Systems 17 (1–4*): 375–390.

Walker, W A; Meade, J G; Brownell, Jr. R L (2002) Diets of Baird's Beaked Whales, *Berardius bairdii*, in the Southern Sea of Okhotsk and off the Pacific Coast of Honshu, Japan. *Marine Mammal Science 18*: 902–918.

Walters, A; Lea, M-A; van den Hoff, J; Field, I C; Virtue, P; Sokolov, S; Pinkerton, M H; Hindell, M A (2014) Spatially explicit estimates of prey consumption reveal a new krill predator in the Southern Ocean. *PLoS ONE 9 (1)*: e86452.

Watkins, W A; Daher, M A; Fristrup, K M; Howald, T J (1993) Sperm whales tagged with transponders and tracked underwater by sonar. *Marine Mammal Science 9*: 55–67.

Whitehead, A L; Lyver, P O'B; Ballard, G; Barton, K; Karl, B J; Dugger, K M; Jennings, S; Lescroël, A; Wilson, P R; Ainley, D G (2015) Factors driving Adélie penguin chick size, mass and condition at colonies of different sizes in the Southern Ross Sea. *Marine Ecology Progress Series 523*: 199–213.

Whitehead, H (2000) Density-dependent habitat selection and the modeling of sperm whale (*Physeter macrocephalus*) exploitation. *Canadian Journal of Fisheries and Aquatic Sciences 57*: 223–230.

Wienecke, B (2011) Review of historical population information of emperor penguins. *Polar Biology 34*: 153. doi:10.1007/s00300-010-0882-0.

Winder, M; Schindler, D E (2004) Climate change uncouples trophic interactions in an aquatic ecosystem. *Ecology 85 (8)*: 2100–2106.

Young, E C (1981) The ornithology of the Ross Sea. *Journal of the Royal Society of New Zealand 11 (4)*: 287–315.

Yukhov, V L (1971) The range of *Dissostichus mawsoni* Norman and some features of its biology. *Journal of Ichthyology 11*: 8–18.

Yukhov, V L (1972) The range of fish of the genus *Dissostichus* (Family: Nototheniidae) in the Antarctic waters of the Indian Ocean. *Journal of Ichthyology 12 (2)*: 346–347.

THEME 5: MARINE BIODIVERSITY

AEBAR 2018: *Snapshot* of Chapter 18 - Biodiversity

DECLINING MARINE BIODIVERSITY



Habitat loss, debris, climate change

1. THE ISSUE:

Ecosystems, biodiversity and the productivity of the marine environment are under threat *WORLDWIDE* from the cumulative effects of human pressures on the ocean. Poorly understood taxonomy combines to making it difficult to track the rate of decline and the contribution of biodiversity to a healthy functioning marine ecosystem is not easily quantified.

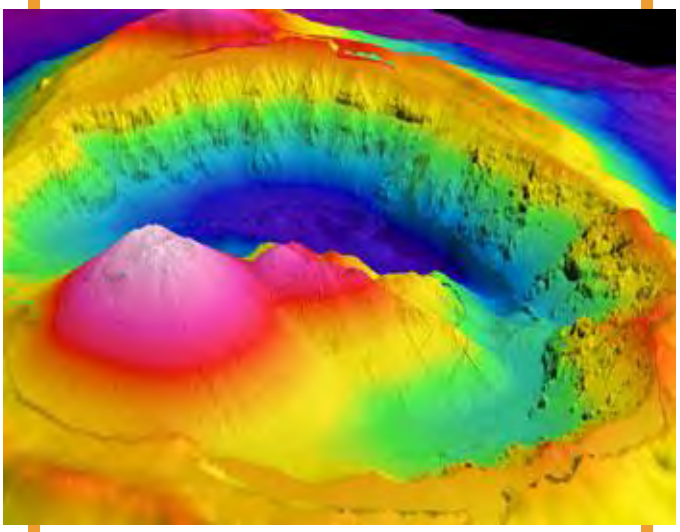
Since the launch of New Zealand's Biodiversity Strategy 2000, MPI has run a Marine Biodiversity Research Programme with 67 projects to date that address some of the knowledge gaps in New Zealand. (Note: the effects of fishing are covered in other chapters.)

2. New Zealand's marine biodiversity is rich with ~ 18,000 known species and a further 13,000 species yet to be described. Our marine biodiversity forms 8% of global marine biodiversity and approximately 40% of species identified so far are endemic to New Zealand waters.

New Zealand's Exclusive Economic Zone, Territorial Sea and Legal Continental Shelf forms an area of 5.8 million km^2, one of the five largest in the world.

The number of species discovered continues to climb annually. Our state of knowledge about marine biodiversity is assessed as being sixth across 18 nations. ID guides are published for a wide range of biota and many species have been genetically bar coded.

3. Info Needs: Seabed Mapping



Methods to quantify biota across large tracts of ocean bed have gradually improved through the development of new technology. However, "Mapping the farm" to identify habitats and biodiversity hotspots is a long-term goal that is far from complete in New Zealand.

More holistic fisheries management approaches require greater understanding of biodiversity and habitat distribution.

4. There are six biodiversity work-streams

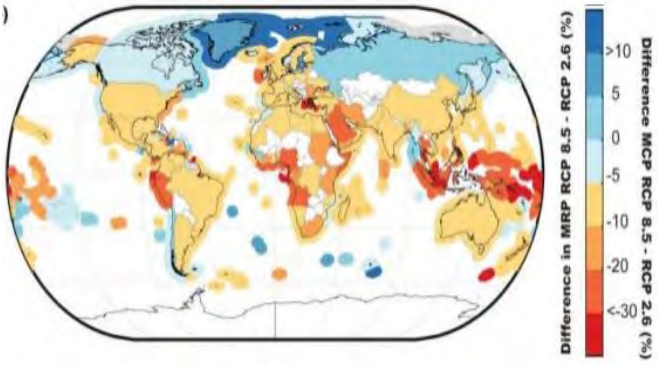
1. Ecological inputs for more holisitic fisheries management
2. Indicators and monitoring change
3. External threats to fisheries and marine ecosystems
4. Developing the blue-green economy
5. Social license for fisheries
6. International biodiversity obligations

5. DIRECT THREATS TO BIODIVERSITY and marine ecosystems from human activities include commercial fishing, recreational fishing, seabed mining, nutrient and sediment run-off, biosecurity risks, noise pollution and marine debris including plastics, noise. Climate change and ocean acidification is also a serious threat to biodiversity (see Chapter 12 also). Some useful indicators of change have been identified, but tracking ecosystem health as a whole is more elusive.

6. The marine environment is changing as a result of climate change and CO2 emissions

1. measurable changes in seawater such as temperature, acidification, oxygen
2. marine heatwave extremes
3. changes in primary productivity
4. shifts in distribution of some species
5. poor shell growth in juvenile shellfish
6. behavioural changes of some animals
7. changes in food supply from surface waters
8. dissolution of deepwater corals at shallower depths

7. Projected impacts of climate change



Some fish will be winners while others will be losers. : Lam et al (2016) IPCC

8. Strategic uptake of biodiversity research

The Biodiversity Research Programme enables Fisheries New Zealand and MPI to meet longer term commitments to New Zealand's international and national obligations regarding biodiversity, the environmental footprint of fishing, spatial management and cumulative effects of environmental change.

Several research projects have been operationalised. Examples include land-sea interactions; spatial marine protection; Tier 1 Statistic development for the marine environment; innovative modelling approaches; Vulnerable Marine Ecosystem move-on rules in SPRFMO; Marine Environmental Classification; eco-certification; substrate-disturbance modelling and sedimentation; bioindicators.

9. New drivers are influencing our biodiversity research

- Public pressure for more holistic long-term fisheries management is increasing.
- New tools for ecosystem based management are emerging from the Sustainable Seas science challenge
- Climate change effects in New Zealand fisheries and the marine environment are more evident.
- New gear design and new survey technologies are evolving rapidly.
- Finer-scale fisheries management is being called for inshore areas
- The NZ Biodiversity Strategy 2000 is being re-written for release in 2020.
- The CBD's Aichi Targets will be re-written for release in 2020

18. BIODIVERSITY

Status of chapter	This chapter has been fully updated for AEBAR 2018.
Scope of chapter	This chapter outlines the current status and trends in marine biodiversity research, links between biodiversity and ecosystem function, and innovative methods and indicators for evaluating marine biodiversity and ecosystem health. An overview of research within the Fisheries New Zealand Marine Biodiversity Programme from 2000–2018 is described within the context of international and national policy obligations, whole-of-government research initiatives (e.g., Ocean Survey 20/20, International Polar Year, Census of Marine Life) and aligned international and national research programmes.
Area	New Zealand Territorial Seas, EEZ and extended Continental shelf; South-west Pacific Region associated with South Pacific Regional Fisheries Management Organisation (SPRFMO); Southern Ocean and Ross Sea region
Focal issues	• Development of robust methods to 1) estimate biodiversity patterns and trends, 2) use as suitable indicators for national and international reporting, and 3) meet obligations within constraints of limited data. • Improving understanding of links between biodiversity and ecosystem function in near-shore and offshore marine ecosystems, and effects of cumulative stressors on ecosystem health. • New approaches and indicators to evaluate efficacy of current spatial measures and management actions to protect marine biodiversity. • Increasing recognition of connections between land-based stressors (e.g. sediment and nutrients) and health of near-shore biodiversity and ecosystems (see Chapter 15) • Mapping and documenting the identity, abundance and distribution patterns of New Zealand's marine biodiversity in this extremely large area of responsibility (about 5.8 million km^2) is far from complete. Identifying areas of high biodiversity remains a challenge, particularly for environmental impact evaluation or assessing response to climate change scenarios.
Emerging issues	• Poor understanding of risks of climate change on marine biodiversity and ecosystems. • Increases in marine debris and pollution, particularly in the coastal zone. • Ecosystem approaches to marine resource management are urgently needed, to allow development of the blue economy within environmental constraints and facilitate integration of socioeconomic factors into fisheries management. • Exploration of new technologies for estimating marine biodiversity (e.g. eDNA). • Poor understanding of the role of marine microbial biodiversity in large-scale biogeochemical and ecosystem processes including productivity.
Fisheries New Zealand Research (current)	Current research includes: ZBD2018-01 *5 year continuous plankton survey (Phase 3)*; ZBD2018-02 *Climate change, fish distribution meta-analysis*; ZBD2018-03 *Climate variability, trends and fish population parameters*; ZBD2018-05 *Ecosystem function and regime shifts in the Sub-Antarctic*; ZBD2016-03 *Multiple Stressors on Coastal Ecosystems-in situ*; ZBD2016-11 *Quantifying benthic biodiversity across natural gradients*; ZBD2016-04 *Organic Carbon Recycling* in Deepwater; ZBD2014-10 *BPA biodiversity*; ZBD2014-03 *Sub-lethal effects of environment change on fish populations*; ZBD2014-04 *Isoscapes for trophic studies*; ZBD2014-06 *Macroalgae mapping and potential as national scale indicators*; ZBD2014-07 *Southern coralline algae shellfish habitat*; ZBD2014-09 *Climate change risks and opportunities*; ZBD2013-02 *VME connectivity*.

Government and other research	Synergies with Aquatic and Environment Working Group (AEWG). Fisheries New Zealand "Fisheries Change Programme", Natural Resource Sector, environmental reporting (MfE), NZ Biodiversity Action Plan (DOC). Research programmes and database initiatives at research institutes and universities, e.g. NIWA Strategic Science Investment Funding – Coasts & Oceans and Fisheries centres; World Register of Marine Species (WoRMS), OBIS, NZ Organisms Register. MBIE National Science Challenge 'Sustainable Seas'; MBIE funded project CARIM (Coastal Acidification: Rates, Impact and Management); MBIE funded project Ross Sea Research And Monitoring Programme: is the world's largest MPA effective?
Related chapters/issues	Cumulative effects, land-based effects, protected areas, benthic impacts, ecosystem approaches to fisheries and marine resource management.

18.1 INTRODUCTION

This chapter summarises the development and progress of the Fisheries New Zealand Biodiversity Research Programme 2000–18, and reviews the work commissioned in the context of national and global concerns about maintenance of biodiversity and the marine ecosystem in a healthy functioning state, as identified by the New Zealand Biodiversity Strategy (NZBS; Anon 2000).

The recognition of increasing societal expectation to use fisheries management measures that will achieve biodiversity conservation was signalled in the policy document Fisheries 2030 (Ministry of Fisheries 2009) in its long-term commitment to '*ecosystem based fisheries management*' and to ensuring that *'biodiversity and the function of ecological systems, including trophic linkages, are conserved*'. While New Zealand's environmental performance with regard to fishing is perceived to be relatively high on an international scale, Fisheries New Zealand is not complacent about the ongoing requirement to monitor and provide evidence that measures to achieve biodiversity conservation needs are being met. This includes the need to better understand and mitigate the effects of fishing in the areas impacted by fishing, and to evaluate the effectiveness of other management approaches such as marine reserves and benthic protection. The effects of fishing on the aquatic environment and risks to biodiversity and the aquatic environment are supported through Fisheries New Zealand's Deepwater Fisheries Plan, as well as the Aquatic Environment and Biodiversity Research Programmes.

The Ministry is also one of several New Zealand government agencies with a strong interest and a statutory management mandate in the Ross Sea region of Antarctica through the Antarctic Marine Living Resources Act 1981. Fisheries New Zealand's Antarctic science contributes strongly to New Zealand's whole-of-government involvement in contributions to the Commission for the Convention on Antarctic Marine Living Resources (CCAMLR) and the Antarctic Treaty.

There are a range of societal values beyond commercial, customary and recreational take from the sea that are recognised as part of 'strengthening our society' in New Zealand. These include aesthetic and cultural values as well as other economic values such as tourism and marine recreation other than fishing (Le Heron et al. 2016). To link socioeconomic values of biodiversity to science supporting fisheries management will require a multi-disciplinary approach only just beginning in New Zealand (Lundquist et al. 2016).

18.1.1 NATIONAL BIODIVERSITY STRATEGY

In June 2000, the 'New Zealand Biodiversity Strategy – Our Chance to Turn the Tide' (NZBS) was launched as part of New Zealand's commitment to the international Convention on Biological Diversity 1993 (Anon 2000). To meet long-term goals of the NZBS, (i.e., to halt the decline of biodiversity in New Zealand and protect and enhance the environment), a comprehensive plan with stated objectives and actions, was developed to address biodiversity issues in terrestrial, freshwater and marine systems.

In the marine environment, biodiversity decline is characterised primarily in relation to megafaunal taxa (marine mammals, seabirds, see Chapters 4-8) and other protected species, which may experience changes in threat status and risk of extinctions. Biodiversity is also influenced by environmental degradation due to habitats being

diminished or removed, and by the disruption of ecosystem structure and function, as well as the disruption of ecological processes (e.g., biological cycling of water, nutrients and energy), species invasion and hybridisations that affect the diversity of marine species and their life history strategies. Measuring the decline of marine biodiversity is complicated by the 'shifting baseline syndrome', a common obstacle to useful biodiversity assessment and monitoring (Soga & Gaston 2018). Furthermore the size range of organisms sampled is often limited to macroscopic or larger. Changes (declines) in the diversity of smaller-sized organisms below the sampling threshold that may be critical to marine ecosystem health and well-being are therefore likely to be missed (Azam & Malfatti 2007).

The task of implementing New Zealand's Biodiversity Strategy is led by the Department of Conservation (DOC), with significant input from the Ministry for Environment (MfE), and the former Ministry of Fisheries (now Fisheries NZ, part of MPI). In 2016 DOC completed a cross-government process to refresh the Biodiversity Strategy to better meet the Aichi Agreement, and released New Zealand's Biodiversity Action Plan 2016–2020 (Department of Conservation 2016).

The key new goal in the Action of direct relevance to Fisheries New Zealand is:

Goal B: National Target 5. Biodiversity is integrated into New Zealand's fisheries management system.

18.2 THE FISHERIES NEW ZEALAND BIODIVERSITY RESEARCH PROGRAMME

The Ministry of Fisheries responded to the NZBS in 2000 with the establishment of the Marine Biodiversity Research Programme (MBRP) to address aspects of biodiversity in NZBS Theme 3 that would complement research under the Aquatic Environment Programme. The focus of the MBRP was initially to map and describe marine biodiversity across New Zealand and in the Ross Sea, particularly to conduct benthic habitat surveys and describe the taxonomy of new species. DOC and Ministry of Fisheries/MPI research on protected species and marine spatial protection was largely dealt with outside the MBRP. In more recent years, the MBRP focus has incorporated the effects of climate change on marine biodiversity.

The core purpose of the Marine Biodiversity Programme is

"To ensure that biodiversity, marine habitat diversity, and ecosystem services that underpin the sustainability of wild caught fisheries productivity and ecosystem resilience are understood and maintained."

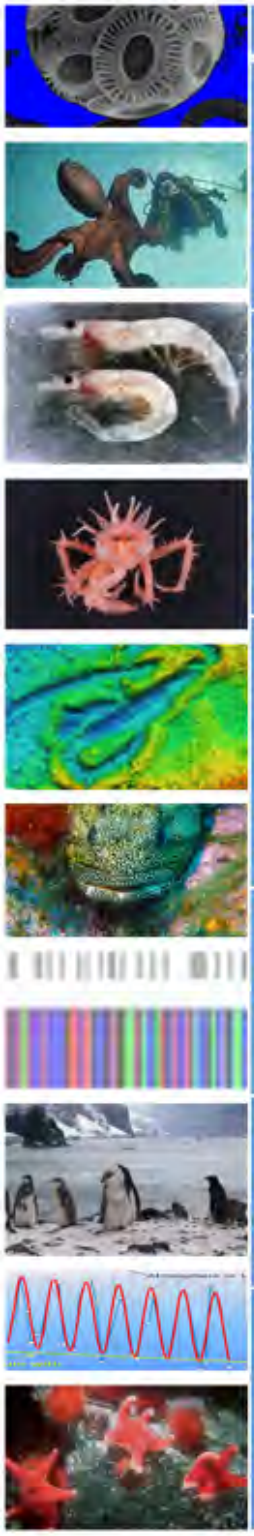
The Biodiversity Research Programme is guided by a multi-stakeholder biodiversity research advisory group (BRAG), chaired by Fisheries New Zealand. The research commissioned for the period 2001–05 reflected goals set by the NZBS and the BRAG, while remaining compatible with the Ministry of Fisheries Statements of Intent. During the first three years of this period, the Ministry of Fisheries also commissioned marine biosecurity research under NZBS, but this was transferred to Biosecurity New Zealand (MAFBNZ) in 2004. From 2006 to 2010, the programme evolved further with the development of a five-year work programme to address shortcomings identified in a review of the NZBS by Green & Clarkson (2006). Biodiversity Workstreams were revised in 2017 to reflect the changes in strategic direction across MPI and to ensure that information needs for MPI were not only met but also connected well to wider government directions and emerging issues. An overview of the current Biodiversity Programme Workstreams is given in Figure 18.1.

18.3 MARINE BIODIVERSITY TRENDS IN NEW ZEALAND

18.3.1 NEW ZEALAND'S MARINE BIODIVERSITY

New Zealand's ocean territory is vast, covering approximately 58 million km^2 including the territorial sea and the recent continental shelf extension. It is very large relative to the area of land, and includes approximately 15–18 000 kilometres of coastline extending from the subtropical north to the cool subantarctic waters in the south. New Zealand also has a rich marine biodiversity that is globally significant with up to 38% of all marine species

(46% for Animalia) estimated as endemic (Gordon et al. 2010, Lundquist et al. 2014) and comprising up to 8% of global marine biodiversity. These estimates do not include estimated undiscovered species, which are likely to increase the proportion of endemics.



Workstream	Purpose
Workstream 1. Ecological information needs for a whole of systems approach to domestic fisheries management	Identify key biodiversity and ecosystem components at risk from fisheries activity that will inform mitigation, spatial planning in New Zealand and help to meet development goals for Ecosystem Based Fisheries Management (EBFM)
Workstream 2. Develop tools and methods to assess and track the footprint of fisheries related activities on biodiversity and ecosystem functioning	Work synergistically across government in producing tools such as ID guides, ecological maps or atlases of seabed habitats and resilience, ecosystem and biodiversity indicators and report cards. Genetic tools and emerging technology is of relevance to this workstream.
Workstream 3. Identify and monitor threats and opportunities for adaptation or mitigation associated with environmental change.	Work across government, CRs, regional councils and universities to gain a national and regional record of long term changes in our marine environment through the development of robust indicators and monitoring programmes; investigate the effects of environmental change (climate change, regime shifts) and ocean acidification on ecosystems and fisheries
Workstream 4. Develop the blue-green economy within environmental constraints.	Work across government to assess the cumulative effects of multiple human stressors on the marine environment and identify creative solutions that allow for sustainable development of the blue-green economy without degrading habitats critical to marine ecosystem health.
Workstream 5. Evaluate and safeguard natural capital for future generations.	Developing methods and understanding to enable fisheries management to take socio-economic factors into account under EBFM. Includes social license to operate, socio-econmic modelling, and matauranga Maori.
Workstream 6. Progressing Ecosystem Based Fisheries Management under international obligations	Purpose. Work across government to identify information and research needs on marine biodiversity to meet international obligations to CCAMLR, the Antarctic Treaty, UNCLOS, SPRFMO, CBD, UNGA, IPBES (the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services), and FAO.

Figure 18.1 Summary of Fisheries New Zealand's Biodiversity Research Programme 2017-2023 Work streams.

The most recent summary of knowledge about marine biodiversity in New Zealand is provided by Gordon (2009, 2010, 2012) and Gordon et al. (2010), which estimates a tally of 17 987 living species in the EEZ, including 4320 known undescribed species in collections. More recent updates using all records available within OBIS, NIWA and Te Papa collections assessed the spatial distribution of biodiversity records and suggested metrics for reporting on the status of marine biodiversity (Lundquist et al. 2014). Species diversity for the most intensively studied animal phyla (Cnidaria, Mollusca, Brachiopoda, Bryozoa, Kinorhyncha, Echinodermata, Chordata) is more or less equivalent to that in the ERMS (European Register of Marine Species) region, an area 5.5 times larger than the New Zealand EEZ (Gordon et al. 2010), suggesting that the New Zealand region biodiversity is proportionately richer than the ERMS region. New Zealand has approximately 50% lower species richness than Australia (in terms of species/area), noting that Australia's EEZ also includes substantial tropical and subtropical regions.

These approximately 18 000 known marine species and associated ecosystems around New Zealand deliver a wide range of environmental goods and services that sustain considerable fishing, aquaculture and tourism industries, as well as drive major biogeochemical and ecological processes. An estimate of undiscovered marine biodiversity suggests another 13 000 species, with several factors suggesting that this estimate of total marine species is conservative. Such factors include the region's size, the depth range, geomorphological and hydrological complexity, limited water column sampling and limited benthic sampling, especially below 1500 m, and rates of new species descriptions, currently about 50 per year. Inflating estimates of undiscovered marine biodiversity is the potentially very large numbers of parasitic and commensal protists (especially microsporidia) and parasitic animals such as myxozoans and nematodes, as well as free-living nematodes. Expectations of massive oceanic microbial diversity (e.g., Sogin et al. 2006) have yet to eventuate; an expert assessment recently determined that between one-third and two-thirds of marine eukaryotic species may be undescribed and previous estimates of there being more than one million such species appear highly unlikely (Appeltans et al. 2012).

18.3.2 MECHANSIMS FOR INCREASING MARINE BIODIVERSITY RECORDS

Scientific research has provided information about the predicted distribution and abundance of marine biodiversity in some areas of New Zealand's coasts and oceans. Advances in the marine protection of the Ross Sea Region have been made and available information has been used to assess habitat types at greatest risk from disturbance, particularly fishing (Clark & Rowden 2009, Clark & Tittensor 2010, Hewitt et al. 2011a, 2011b, Floerl & Hewitt 2012). Many ecosystems within New Zealand waters remain poorly sampled however, and the efficacy of current spatial protection measures for biodiversity in New Zealand is unknown. Further, the proportion of different marine habitat types that should be or can be protected to maintain a healthy aquatic environment is also unknown (Lundquist et al. 2015).

A number of initiatives have been supported by Fisheries New Zealand and its predecessors to meet the goals of the NZBS. Commitments include the creation of NABIS (the National Aquatic Biodiversity Information System; http://www.nabis.govt.nz), for making data on marine biodiversity more accessible. DOC surveys and monitors aspects of marine biodiversity, particularly in marine reserves and in relation to protected and threatened marine species. Periodic marine 'BioBlitzes' around New Zealand yield surprising numbers of new species, even in presumed well-studied areas, have the advantage of engaging children, parents and teachers in discovery, and involve research scientists who go on to describe the new taxa (e.g., Harper et al. 2009). More of these Citizen Science projects can be found under the MBIE funded Curious Minds programme (https://www.curiousminds.nz). The museums of Auckland, Canterbury, Otago and the Museum of New Zealand (Te Papa) also conduct biodiversity sampling expeditions and national collections of specimens have been set up within museums and at NIWA.

Marine biodiversity research is largely supported through public good funding and is conducted in both universities and CRIs. Both have contributed to New Zealand's high profile for marine biodiversity on the international scientific network through participation in global initiatives such as the Census of Marine Life (CoML) (http://www.coml.org), as well as to local programmes that have improved understanding of the role of biodiversity in the marine ecosystem.

In 2010, New Zealand contributed to the CoML, an unprecedented collaboration among researchers from more than 80 nations to assess and explain the diversity, distribution, and abundance of life in the oceans. NIWA scientists were part of the team that led CenSeam (http://www.coml.org/global-census-marine-life-seamounts-censeam), the seamount component of CoML, and New Zealand scientists played significant roles in a number of other programmes. The New Zealand International Polar Year-Census of Antarctic Marine Life (IPY-CAML) voyage to the Ross Sea in 2008 was also a major contribution to CoML (O'Driscoll 2009).

The Census facilitated activities that led to better assessments of global marine biodiversity, resulting in an increase in the total number of known marine species by about 20 000, from 230 000 in 2000 to about 250 000 in 2010. The digital archive (the Ocean Biogeographic Information System OBIS, http://www.iobis.org) has now grown to 31 million observations, and the Census helped to create the first comprehensive list of the known marine species, and facilitated an initiative which resulted in a web-based database of over 80 000 species, the Encyclopedia of Life (http:/www.eol.org).

A summary of the overall state of knowledge about marine biodiversity after the Census by Costello et al. (2010) places New Zealand sixth out of 18 national regions based on the collective knowledge assembled by the Census National and Regional Implementation Committees (NRIC) and comparing the Spearman rank correlation coefficients between known diversity (total species richness, alien species, and endemics) and available resources, such as numbers of taxonomic guides and experts.

18.3.3 THREATS TO BIODIVERSITY

Understanding about New Zealand's coastal marine environment and its land-sea interactions has progressed since the launch of the NZBS, although knowledge about the state of the marine environment and marine biodiversity at a national scale remains limited (Lundquist et al. 2014). Current knowledge about New Zealand's and the Ross Sea's marine biodiversity suggests that it may generally be in better shape than that of many other countries (Costello et al. 2010, Gordon et al. 2010). New Zealand was placed 12th out of 18 regions in terms of overall threat levels to biodiversity, overfishing and alien species invasion. Habitat loss and ocean acidification were identified as the biggest threats to marine biodiversity and marine habitats in New Zealand (Costello et al. 2010, MacDiarmid et al. 2012).

Key marine environment and biodiversity related stressors are outlined in Environment Aotearoa 2015 (Ministry for the Environment & Statistics New Zealand 2015). New Zealand's marine biodiversity is affected by many uses of the marine environment, particularly fishing, aquaculture, shipping, petroleum and mineral extraction, renewable energy, tourism and recreation (Royal Society of New Zealand 2012). Impacts from changing land use, including agricultural, urban run-off and coastal development can also affect marine biodiversity (Morrison et al. 2009). There are ongoing concerns about the decline of some key species (Ministry for the Environment 2016), localised impacts on habitats and conditions (Thrush & Dayton 2002, Cryer et al. 2002, Clark et al. 2010a, 2010b, Gordon et al. 2010) and emerging threats to the marine environment (MacDiarmid et al. 2012).

The past 750 years of human activity has impacted on marine environments. For example, depletion of fur seals and sea lions occurred from the earliest days of human settlement, not just with European arrival (Smith 2005, 2011). There was also a pulse of sedimentation coinciding with the initial clearance of 40% of New Zealand forests within 200 years of Polynesian settlement (McWethy et al. 2010). Impacts have occurred in remote areas as well as near population centres, and to depths in excess of 1000 m (Carroll et al. 2015, MacDiarmid et al. 2014, 2015, 2016a, 2016b, Maxwell & MacDiarmid 2016, Pinkerton et al. 2015, Fisheries New Zealand 2018).

While New Zealand has reasonable archaeological, historical and contemporary data on the decline in abundance of some individual marine species, current trends in the status of New Zealand's marine biodiversity are difficult to determine for several reasons. These include a lack of both pre-disturbance baseline and recent information, and a lack of a nationally coordinated approach to assessing and monitoring marine biodiversity (Lundquist et al. 2014).

Threat status and susceptibility to capture as fisheries bycatch is discussed in prior chapters for marine mammals, seabirds and other protected species (see Chapter 4-8). Progress has been made on evaluating threats and risks to the marine environment and components within it (e.g., Currey et al. 2012, MacDiarmid et al. 2011, 2012, 2014,

Ministry for Primary Industries 2013; Ministry for Primary Industries & Department of Conservation 2013) and some of these have been followed up with a Spatially Explicit Risk Assessment (Richard et al. 2017). Marine mammals, seabirds and reptiles are regularly assessed using the New Zealand Threat Classification System (NZTCS). However, most fish and invertebrates have not been assessed (reviewed in Lundquist et al. 2014). For example, only a small fraction (less than 5%) of New Zealand's marine invertebrate fauna have been evaluated for their threat status and many taxa remain 'data deficient' or unlisted (Freeman et al. 2010, Freeman et al. 2013).

MacDiarmid et al. (2012) undertook an expert assessment of the impact of 65 potentially hazardous human activities on 62 identifiable marine habitats in New Zealand's Territorial Sea and 200 nautical mile Exclusive Economic Zone (EEZ). Experts concluded that many of the biggest threats stemmed from human activities outside the marine environment itself. The two biggest threats identified by participants were ocean acidification and ocean warming. Seven other threats deriving from global climate change all ranked in the top 20 threats indicating the importance of global climate change to New Zealand's marine ecosystems.

Climate change can have an adverse impact on the spatial patterns of marine biodiversity and ecosystem function through changes in species distributions, species mix and habitat availability, particularly at critical stages of species life histories (Lundquist et al. 2011, Poloczanska et al. 2013). Understanding the dynamics of climate change and predicting the impacts on food webs and fisheries productivity has improved and is a substantial research topic in many parts of the world (e.g., Brown et al. 2010; Blasiak et al. 2017; Phillips & Perez-Ramirez 2017), including New Zealand (e.g., CARIM; http://www.carim.nz).

Understanding about the resilience of biodiversity and its rate of change in response to the cumulative effects of multiple stressors across large spatial scales (e.g., ocean acidification, temperature increase and oxygen depletion) remain ongoing topics of investigation. A study of global patterns of climate change impacts predicted dramatic species turnovers of over 60% of present biodiversity by 2050. This was based on model projections of changes in the distributional ranges of a sample of 1066 exploited marine fish and invertebrates using a newly developed dynamic bioclimate envelope model (Cheung et al. 2009). The study showed that climate change and other stressors may lead to numerous local extinctions in the sub-polar regions, the tropics and semi-enclosed seas, as well as ecological disturbances that potentially disrupt ecosystem services. (Cheung et al. 2009).

The loss of marine biodiversity and loss of functionality associated with climate change and ocean acidification are of increasing concern worldwide (e.g., Guinotte et al. 2006, Ramirez-Llodra et al. 2011), as well as in New Zealand (e.g., Royal Society of New Zealand 2012). In late June 2011, two science-based reports heightened concerns about the critical state of the world's oceans in response to climate change. One focussed on the potential impacts of ocean acidification on fisheries and higher trophic level ecology and took a modelling approach to scaling from physiology to ecology (Le Quesne & Pinnegar 2012) and the other assessed the critical state of the world's oceans in relation to climate change and other stressors (Rogers & Laffoley 2011). Implications of global climate change have been further summarized by the IPCC (IPCC 2018).

Global-scale threats associated with the potential effects of ocean acidification on microbial diversity and their roles in biogeochemical processes have yet to be quantified but could have EEZ-wide implications (Bostock et al. 2012). The growing arrival of non-indigenous (sometimes invasive) marine species is also a threat to local biodiversity (e.g., Cheung et al. 2009, Coutts & Dodgshun 2007, Cranfield et al. 2003, Gould & Ahyong 2008, Russell et al. 2008, Williams et al. 2008).

18.3.4 NATIONAL BIODIVERSITY LEGISLATION, REGULATIONS AND POLICIES

New Zealanders increasingly value environmental, economic and social aspects of marine biodiversity and the ecosystem services that a healthy marine environment provides. They also recognise the need to sustainably manage the use of coastal and marine environments and maintain biological diversity as reflected by policy statements by the New Zealand government (Ministry for the Environment 2011; Department of Conservation 2010). A broad range of legislation, regulations and policies are in place to manage and regulate uses of the marine environment, to protect marine biodiversity, to improve management of the coastal and marine environment and to meet worldwide consumer demands for improved sustainability. Trying to manage this in a more holistic way

through EBFM has already been identified by Fisheries New Zealand as a medium-term goal in the Future of Our Fisheries review.[1]

New Zealand's Business Growth Agenda acknowledges the lack of progress in halting the decline in biodiversity in New Zealand, and indicates that development of the marine economy requires a careful approach to the environment. The BGA was updated in 2017, and the Natural Resources Chapter outlines priorities for both progressing biodiversity protection, and sustainable growth of marine resource industries:

> '*develop our aquaculture, fisheries and other marine industries to enable sustainable growth while maintaining marine biodiversity and sustainability.* '

The most recent introduction of new legislation is the Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012. However, there is currently no integrated national oceans policy, and ocean management is fragmented across agencies and statutes (Lundquist et al. 2016; Davies et al. 2018a). Initiatives and research funded through the Sustainable Seas National Science Challenge are developing a framework for cumulative effects management across land and sea resource management sectors, industries and communities, an internal component of an EBM approach. A further Sustainable Seas project has examined whether current legislation, policy and institutional practice are sufficient to enact EBM, and what further approaches could be acted on to further enhance EBM.

Marine spatial planning and marine reserves have allowed for stakeholder driven processes to inform marine management and allocation of space for marine protection (Davies et al. 2018b) and other uses of the marine environment (e.g. aquaculture). The Hauraki Gulf Forum initiated the Sea Change – Tai Timu Tai Pari Hauraki Gulf Marine Spatial Plan in 2013, taking a holistic and integrated approach to marine management and restoring the marine productivity in the area (http://www.seachange.org.nz), with a recent announcement from government to support implementation of the plan (November 2018).

18.3.5 INTERNATIONAL BIODIVERSITY OBLIGATIONS

At its 65th session, The United Nations General Assembly declared the period 2011–2020 to be '*the United Nations Decade on Biodiversity, with a view to contributing to the implementation of the Strategic Plan for Biodiversity for the period 2011–2020*' (Resolution 65/161). The decade serves to support and promote implementation of the objectives of the Strategic Plan for Biodiversity and the Aichi-Nagoya Biodiversity Targets. The principal instruments for implementation are to be National Biodiversity Strategies and Action Plans or equivalent instruments (NBSAPs).

There are five strategic goals and 20 ambitious yet achievable targets, collectively known as the Aichi Targets. The five Strategic Goals are:

- Goal A – Address the underlying causes of biodiversity loss by mainstreaming biodiversity (NBSAPs) across government and society.
- Goal B – Reduce the direct pressures on biodiversity and promote sustainable use.
- Goal C – Improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity.
- Goal D – Enhance the benefits to all from biodiversity and ecosystem services.
- Goal E – Enhance implementation through participatory planning, knowledge management and capacity building.

Targets 6–11 specifically refer to fisheries and marine ecosystems (Box 18.1). New Zealand has responded and updated its Biodiversity Action Plan (Department of Conservation 2016).

[1] Ministry for Primary Industries (MPI). Future of our Fisheries. https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/strengthening-fisheries-management/future-of-our-fisheries.

Box 18.1 Aichi Targets relevant to marine biodiversity

Target 6: By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem-based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits.
Target 7: By 2020 areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.
Target 8: By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.
Target 9: By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.
Target 10: By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning.
Target 11: By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscape and seascapes.

The full text of the First Global Integrated Marine Assessment, conducted by some of the world's foremost experts on ocean issues for policymakers, was released online in 2016 (http://www.worldoceanassessment.org). The Intergovernmental Panel on Climate Change released its 6th Report in 2018, providing further evidence of current and projected climate change trajectories, and likely impacts on humans and ecosystems (IPCC 2018).

A new United Nations platform, the IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services), was established in 2012, and provides a mechanism to assess the state of the planet's biodiversity, its ecosystems and the essential services they provide to society (http://www.ipbes.net). This new international platform is similar in function to the IPCC in terms of bringing together international expertise, and will review information on the provisioning of biodiversity for ecosystem services, stimulate science and innovation on this research topic, and interact with national and international management agencies to integrate IPBES results into policy and management. IPBES is currently completing its first 5 year work programme, and to date has released two technical assessments and four regional assessments (for which New Zealand is included in the Asia-Pacific regional assessment). A global assessment of biodiversity and ecosystem services will be presented to governments for approval at next IPBES plenary.

Most of New Zealand's commercial fisheries are wild-caught, and continuity of their productivity is dependent on the retention of a healthy functioning marine ecosystem. The 'license to operate' is mandated by the Fisheries Act 1996 that requires strict compliance with sustainable and environmentally responsible use of fish stocks. Compliance is also required with other legislation such as the Marine Mammals Protection Act 1978 and a range of international obligations such as the United Nations Convention on the Law of the Sea (UNCLOS). Under the Quota Management System, considerable monitoring of fishing activity and the environmental footprint of commercial operators is required.

18.3.6 CURRENT CHALLENGES AND AGENDAS

Since the launch of the Biodiversity Strategy, there have been substantial changes in government policy and changes to the science agenda in New Zealand. Improving New Zealand's economic performance while continuing to strengthen society and protect the environment has been a key component for the marine economy as the general public have become increasingly aware of how human behaviour and actions affect the environment.

The latest initiative from the government is the '*Conservation and Environment Science Roadmap that identifies the areas of scientific knowledge needed by government over the next 20 years to support decision-making for conservation and environmental policy and management to achieve the most desirable future for New*

Zealand' (Ministry for the Environment & Department of Conservation 2017).

The economy of the sea is a significant part of the overall economy in New Zealand and has potential for growth, particularly in aquaculture, oil and gas, minerals (MBIE Business Growth Agenda 2017). It is important that the aquatic environment and biodiversity are not adversely affected by new or increasing activities, be they in the seafood sector or other natural resource industries (Fisheries Act 1996; Exclusive Economic Zone and Continental Shelf (Environment Effects) Act 2012).

The large-scale threats to the marine environment and biodiversity include climate change and ocean acidification, increasing exploitation of resources (living or non-living) and the cumulative effect of multiple uses of the marine environment (e.g., renewable energy, commercial fisheries, recreational fisheries, aquaculture, hydrocarbon and mineral extraction) (Kingsford et al. 2009, Lundquist et al. 2011). These threats are increasingly being recognised in policy and government circles (e.g., Office of the Prime Minister's Science Advisory Committee 2013; Statistics New Zealand 2013; Royal Society of New Zealand 2012; Capson & Guinotte 2014).

Despite this recognition, progress on tackling marine-related climate change effects and investment in long-term monitoring of the marine environment and ready access to data remains slow (although see Box 18.2 for a number of data-collecting initiatives that are contributing to long-term historical datasets, noting that these are primarily of environmental drivers rather than measures of biodiversity itself). Long-term monitoring and environmental reporting has however been recognised as a major gap by the government and in September 2015 the Environmental Reporting Act (2015) was passed into law. The new framework for environmental reporting divides the environment into five environmental domains. Under each domain three main types of information are reported on: pressures, states and impacts. 'Our marine environment 2016' was the first report released under the new legislation (Ministry for the Environment 2016).

18.3.7 GLOBAL MONITORING AND INDICATORS FOR MARINE BIODIVERSITY

There are numerous schemes within and between nations to monitor the marine environment, including physical, chemical and biological components (Box 18.2). A challenge for Fisheries New Zealand, other government agencies and for New Zealand is how to assimilate any or all of the above monitoring approaches and international datasets to assess the nature and extent of biodiversity change, and to assess the effectiveness of management measures to protect or enhance biodiversity or halt its decline.

18.4 OVERALL PROGRESS IN FISHERIES NEW ZEALAND MARINE BIODIVERSITY RESEARCH

18.4.1 HISTORICAL BIODIVERSITY RESEARCH

To date, 74 research projects have been commissioned. Early studies focused primarily on reviews, identification guides, habitat and community characterisations, and revised taxonomy for certain groups of organisms. Later studies have included large collaborative ship-based surveys that have contributed to improved seabed classification in New Zealand waters and the exploration of new habitats in the region and in Antarctic waters.

Progress in the BRAG research programme has increased in conceptual complexity from simply cataloguing biodiversity to an increasingly complex understanding of environmental drivers and the functionality of biodiversity; and ultimately towards the development of standards and the protection of biodiversity. A full list of projects can be obtained from Appendix 19.9 at the back of the full document.

Greatest progress has been made in the shallower inshore parts of the marine environment, not least because of cost and ease of access. However, by leveraging from existing offshore projects, significant progress has also been made to depths of 1500 m. Biodiversity research based in Antarctica lags behind EEZ-based research, simply because of the difficulty in securing additional funding to access and work in such a remote marine environment.

Over time, the complexity and scale of studies has increased, with projects on the functional ecology of marine ecosystems ranging from localised experimental manipulation to broad-scale observations across hundreds of square kilometres. Such studies have also prompted the development of improved measures of biodiversity and indicators. A study on changes in shelf ecosystems over the past 1000 years has yielded insights into the effects of long-

Box 18.2: Examples of international marine global monitoring datasets.

Name	Description	Website
Global Ocean Observing System (GOOS)	Permanent global system for observations, modelling and analysis of marine and ocean variables.	http://www.goosocean.org/
Global Climate Observing System (GCOS)	United Nations-ratified programme which regularly assesses the status of global climate observations and produces guidance for its improvement.	https://public.wmo.int/en/programmes/global-climate-observing-system
Southern Ocean Observing System (SOOS)	International initiative of the Scientific Committee on Antarctic Research (SCAR) and the Scientific Committee on Oceanic Research (SCOR), officially launched in 2011 and hosted by the Institute for Marine and Antarctic Studies (IMAS) at the University of Tasmania, Australia.	Southern Ocean Observing System, http://www.soos.aq http://www.scar.org/soos
ARGO	International deep water monitoring system of free floating buoys that are part of the integrated global observation strategy. New Zealand makes a significant contribution to ARGO floats in the Pacific Ocean.	http://www.argo.net/ https://www.niwa.co.nz/coasts-and-oceans/research-projects/argo-floats
Continuous Plankton Recorder (CPR) Surveys	CPR surveys collect data about the spatio-temporal patterns of zooplankton and then use the sensitivity of plankton to environmental change as an early warning indicator of ocean health. CPR originated in the North Atlantic and the North Sea in 1931. New Zealand BRAG funded projects contribute to the SCAR Southern Ocean CPR data collection established in 1991 by the Australian Antarctic Division to map the Southern Ocean.	https://www.cprsurvey.org/
Australia's Integrated Marine Observing System (IMOS)	IMOS, established in 2007, is designed to be a fully integrated national array of observing equipment to monitor the open oceans and coastal marine environment around Australasia, covering physical, chemical and biological variables. All IMOS data is freely and openly available.	http://imos.org.au
Oceans 2025	This 5 year programme, now completed, was an initiative of the Natural Environment Research Council (NERC) funded Marine Research Centres to address environmental issues that require sustained long-term observations.	https://nerc.ukri.org/research/funded/programmes/oceans2025/
Global Ocean Acidification Observing Network (GOA-ON)	An existing global ocean carbon observatory network of repeat hydrographic surveys, time-series stations, floats and glider observations, and volunteer observing ships, with participation from scientists from over 30 countries.	http://www.goa-on.org/
NZ Ocean Acidification monitoring Network (NZOA-ON)	Initiated in 2014 by NIWA and the University of Otago, this programme now has 11 sampling sites around NZ, and is part of the Ocean Acidification Alliance.	https://www.niwa.co.nz/coasts-and-oceans/research-projects/new-zealand-ocean-acidification-observing-network-nzoa-on https://www.oaalliance.org/current-members

term climate change, land-use effects and fishing, on marine ecosystems while more recently, some studies have begun to address the effects of ocean acidification on marine biodiversity. One study reviewed genetic variation

in the New Zealand marine environment and conducted field observations on several species to examine genetic variation across latitudinal gradients. Aspects of the original seven Workstreams have also been addressed through a range of biodiversity projects in the Ross Sea region including the International Polar Year Census of Antarctic Marine Life project (IPY-CAML). A key to study findings is consideration of biodiversity within the context of the carrying capacity of the system and the natural assemblages of biota supported by that system in the absence of human disturbance. The new Workstreams review focusses on ecosystem-based management to target socio-economic values aligned with research funded by the Sustainable Seas National Science Challenge.

18.4.2 ALIGNED MARINE BIODIVERSITY RESEARCH

In New Zealand, a number of marine research projects have contributed to the state of knowledge and management of marine biodiversity. The 'Marine Futures' programme (2012–14) investigated decision-making frameworks for ocean management and developed new tools for enabling participation of all stakeholders (public, iwi, industry, government), to facilitate economic growth, improve marine stewardship and ensure that cumulative stresses placed on the environment do not degrade the ecosystem beyond its ecological adaptive capacity (MBIE project code C01X1227). The 'Ross Sea Climate & Ecosystem' Programme (concluded in 2016) modelled likely future changes in the physical environment of the region and potential consequences of these changes on the ecosystem in terms of functional links between the environment and the marine food web (MBIE project code C01X1226). 'Management of offshore mining' (concluded in 2016) developed a clear framework that will guide appropriate and robust environmental impact assessments and the development of integrated environmental management plans for the marine-mining sector, other resource users and resource management agencies (MBIE project code C01X1228).

The MBIE-funded Climate Change Impacts and Implications project had a coastal and an ocean case study, and released a synthesis report upon its conclusion in 2016 (Law et al. 2016). The MBIE-funded CARIM project 'Coastal Acidification: Rate, Impacts & Management' that the Ministry has contributed to is also nearing completion (http://www.carim.nz/).

One of the largest marine research developments in 2014 was the launch of the National Science Challenge 'Sustainable Seas'. The Challenge aims to enhance the utilisation of our [NZ] marine resources within environmental and biological constraints (https://sustainableseaschallenge.co.nz/). Achieving this aim will require a new way of managing the many uses of our marine resources that combines the aspirations and experience of Māori, communities, and industry with the evidence of scientific research to transform New Zealand into a world leader in sustainable marine ecosystem-based management.

International partnerships are also being leveraged to support BRAG research priorities. (Table 18.1).

TABLE 18.1: CURRENT NZ/EU PARTNERSHIPS

BAYESIANMETAFLATS – Spatial organisation of species distributions: hierarchical and scale-dependent patterns and processes in coastal seascapes (completed)	https://cordis.europa.eu/result/rcn/185059_en.html
Chess – Biogeography of Deep-Water Chemosynthetic Ecosystems	http://www.coml.org/projects/biography-deep-water-chemosynthetic-ecosystems-chess.html
INDEEP – International Network for Scientific Investigation of Deepsea Ecosystems	http://www.indeep-project.org/
PHARMASEA – Increasing Value and Flow in the Marine Biodiscovery Pipeline	http://www.pharma-sea.eu/
MAREFRAME – Co-creating Ecosystem-based Fisheries Management Solutions	http://mareframe-fp7.org/
BENTHIS – Studies the impacts of fishing on benthic ecosystems and will provide the science base to assess the impact of current fishing practices	http://www.benthis.eu/en/benthis.htm

18.4.3 PROGRESS ON CHARACTERISATION AND CLASSIFICATION OF BIODIVERSITY

The characterisation and classification of biodiversity requires an assessment of the abundance and distribution of marine life. Building on earlier research to map fish and squid species (Anderson et al. 1998, Bagley et al. 2000) and the biodiversity of the New Zealand ecoregion (Arnold 2004), numerous literature reviews, taxonomic studies and habitat mapping surveys have been undertaken, including 'The New Zealand Inventory of Biodiversity' (Gordon 2009, Gordon 2010, Gordon 2012). Field identification guides have also been published by the Ministry of Fisheries on deep sea invertebrates (projects ENV2005-20 and ZBD2010-39, Tracey et al. 2005, 2007, 2011a), macroalgae (Nelson 2013), bryozoans (project IPA2009/14, Smith & Gordon 2011) and on fish species (IDG2006-01, McMillan et al. (2011a, 2011b, 2011c), which further contribute to the accurate monitoring and identification of biodiversity in New Zealand waters.

Several hundred new species of marine organisms have been discovered, and the known range of species extended, through exploratory surveys such as the NORFANZ project ZBD2002-16 (Clark & Roberts 2008); MSI's Seamount Programme, mainly commissioned through public-good science, supplemented by Ministry projects ZBD2000-04 (e.g., Rowden et al. 2002, 2003), ZBD2001-10 (Rowden et al. 2004), ZBD2004-01 (Rowden & Clark. 2010), and Ministry projects ENV2005-15, ENV2005-16 (Clark et al. 2010a, Rowden et al. 2008) and the Ocean Survey 20/20 programme (Clark et al. 2009); inshore surveys of bryozoans at Tasman Bay ZBD2000-03 (Grange et al. 2003); Farewell Spit ZBD2002-18 (Battley et al. 2005); Fiordland ZBD2003-04 (Wing 2005); coralline algae ZBD2001-05, ZBD2004-07 (Harvey et al. 2005, Farr et al. 2009, Opresko et al. 2014) and other deep sea invertebrates (Tracey et al. 2011a, Williams et al. 2014); soft sediment environments ZBD2003-08 (Neill et al. 2012); rhodolith community study ZBD2009-03 (Nelson et al. 2012, 2014); offshore surveys of the Chatham Rise and Challenger Plateau funded through whole-of-government Ocean Survey 20/20 Programme, ZBD2006-04 (Nodder 2008) and ZBD2007-01 (Nodder et al. 2011), ZBD2012-03 (Hewitt et al. 2011a, 2011b, Bowden 2011, Bowden & Hewitt 2012, Bowden et al. 2011a, 2011b, 2011c, 2014, Compton et al. 2012). Other national efforts included the Biosecurity New Zealand mapping projects (Beaumont et al. 2008, 2010).

Research in the Ross Sea Region (BioRoss projects) have also generated records of new species including MPI projects ZBD2000-02 (Page et al. 2001), ZBD2001-03 (Norkko et al. 2002), ZBD2002-02 (Sewell et al. 2006, Sewell 2005, 2006), ZBD2003-02 (Cummings et al. 2003, 2006b), ZBD2003-03 (Rowden et al. 2012a, 2013a), ZBD2005-03 (MacDiarmid & Stewart 2012), ZBD2006-03 (Cummings et al. 2003, 2006a;), ZBD2008-23 (Nelson et al. 2010) and IPY2007-01 (Bowden et al. 2011a, Clark et al. 2010b, Eakin et al. 2009, Hanchet 2009, 2010, Hanchet et al. 2008a, 2008b, 2008c, 2008d, 2013, Koubbi et al. 2011, Lörz et al. 2009, Mitchell 2008, O'Driscoll 2009, O'Driscoll et al. 2011, 2012, O'Loughlin et al. 2011).

The development of the Marine Environment Classification or 'MEC' (Snelder et al. 2006) was an important step in the delineation of areas with similar environmental attributes in the offshore environment. However, significant environmental drivers of variability in marine biodiversity, such as substrate type for seafloor organisms, were absent from the classification. In 2005, DOC and Ministry of Fisheries jointly commissioned a project to optimise the MEC using fish distribution data. This project (ZBD2005-02) demonstrated a substantial improvement in the MEC classification for offshore habitats (Leathwick et al. 2006a, 2006b, 2006c). In addition, the Ministry implemented spatial management tools (Benthic Protection Areas; https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/protected-areas/benthic-protection-areas), on the basis of the Marine Environment Classification (Snelder et al. 2006) to address broader statutory responsibilities on the environmental effects of fishing on biodiversity.

In 2006, three projects to map coastal biodiversity were completed in the Coromandel scallop, Foveaux Strait oyster and southern blue whiting fisheries around the Subantarctic Islands as part of fishery plan development for these fisheries (ZBD2005-04, ZBD2005-15 and ZBD2005-16). These projects found that the biological distribution of organisms and their habitats were not well predicted by the MEC. Ministry project (BEN2006-01) aimed to further optimise the MEC by producing a methodology for a Benthic Optimised MEC (Leathwick et al. 2010). Ecological studies to improve habitat classification and vulnerability indices have also been completed through AEWG projects on seamounts (ENV2005-15, ENV2005-16) (e.g., Clark et al. 2010c, 2011), and to supplement other studies funded by

the Ministry and MSI (e.g., ZBD2004-01, ZBD2001-10, ZBD2000-04 and CO1X0508).

Distribution maps providing indicative abundance and characterisation of biodiversity are now emerging and have been produced through projects using predictive modelling tools, e.g., Compton et al. 2012, ZBD2010-40; the fish optimised MEC in project ZBD2005-02 (Leathwick et al. 2006a, 2006b, 2006c); the benthic optimised MEC (Leathwick et al. 2010); macroalgal diversity associated with soft sediment habitats ZBD2008-05 (Rowden et al. 2012b); deep sea benthic biodiversity in trench, canyon and abyssal habitats below 1500 m depth ZBD2008-27 (Lörz et al. 2012); and rhodolith distribution and associated biodiversity ZBD2009-03 (Nelson 2009, Neill et al. 2014).

Current and recently started projects are as follows:

ZBD2014-10 BPA Biodiversity

Status: near completion.

The project builds on initial work reported by Clark et al. (2014) that summarised NIWA data on benthic invertebrate samples from Benthic Protection Areas and Seamount Closure Areas. Inventory of benthic samples and biodiversity data was updated, and additional undescribed samples were processed and identified for select taxonomic groups. A sampling programme was designed following identification of priority gaps in sample coverage, and a spatial management planning exercise was undertaken to assess the effectiveness of the current BPAs to protect biodiversity.

ZBD2016-11 Quantifying benthic biodiversity across natural gradients

Quantitative data on the distribution and abundance of benthic species are sparse in New Zealand waters, resulting in high levels of uncertainty in status and trends in benthic biodiversity. This project supports opportunities both for limited field validation of existing models (e.g., those developed from the Chatham-Challenger OS20/20 surveys), and development of new abundance-based models. These new models will have improved predictive ability that will better inform management options, and the data and models generated by the project will also inform a benthic risk assessment being developed under a separate Ministry project (BEN2014-01).

Other research relevant or specifically linked to BRAG research Workstreams are listed in Table 18.2

Table 18.2: Other aligned research

Fisheries New Zealand	HAB2007-01 Biogenic habitats as areas of particular significance for fisheries management (complete) ZBD2006-02 NABIS (ongoing) Useful data related to defining potential VMEs are collected by Fisheries New Zealand scientific fisheries observers working on NZ authorised fishing vessels that operate on the high seas in the South Pacific.
CRI Marine platform or MBIE funding	NIWA Strategic Science Investment Funding (SSIF) Coasts & Oceans Centre programmes: Programme 2 – Habitat characterisation Programme 3 – Documenting biota Programme 4 – Ecosystem structure and function Programme 5 – Managing ecosystem change Programme 6 – Marine biosecurity C01X0907 Coastal Conservation Management (fish habitat classification) CO1X0906 Vulnerable deep sea communities (mapping and sampling a range of deep sea habitats (seamounts, slope, canyons, seeps, vents)) (NIWA)
Central government	MEC development and application to MPAs, Regional surveys; refined habitat suitability modelling for protected coral species in the New Zealand EEZ has been undertaken along with the development of a pilot ecological risk assessment for protected corals. DOC – currently developing a biological habitat classification system for the NZ coastal environment, along with methodologies for habitat mapping as part of a monitoring 'toolbox'. DOC Ecological integrity programme - development and testing of indicators of ecological integrity for New Zealand's marine protected areas, e.g., functional traits (de Juan et al. 2015). DOC - internet-based estuaries resource hub and review of citizen science and monitoring programmes. DOC - Marine ecosystem services review and matrix analysis DOC - Marine Cultural Health Indicators with Te Rūnanga o Toa Rangatira (TRoTR) and Greater Wellington Regional Council (GWRC) DOC - Ecologically Significant Marine Areas

18.4.4 PROGRESS ON ECOSYSTEM-SCALE RESEARCH

Near-shore ecosystems are complex and changes in diversity and community composition may be driven by multiple variables. Interactions between variables are likely to be non-linear, with disturbance thresholds and the potential for multiple stable states. As a consequence, it is often difficult to distinguish 'natural' from 'anthropogenic' impacts affecting ecosystem dynamics, and research is needed to help disentangle this complexity.

Marine ecosystems influence, and are influenced by, a wide array of oceanic, climatic and ecological processes across a broad range of spatial and temporal scales. Marine communities are generally dynamic, can occur over large areas and have strong links to other communities through processes such as migration and long-distance physical transport (e.g., of larvae, nutrients and biomass). Patterns observed on a small scale can interact with larger and longer-scale processes that in turn result in large-scale patterns. Marine food webs are usually complex and dynamic over time (Link 1999). To distinguish useful descriptors of long-term ecosystem change from short-term fluctuations requires innovative approaches that integrate broad-scale correlative studies from smaller-scale manipulative experiments (Hewitt et al. 1998, 2007).

Recent theoretical and technical advances show great promise toward the goal of understanding the role of biodiversity in ecosystems. Technologies for remote sensing and deep water surveying, combined with powerful integrative and interpretive tools such as GIS, climate modelling, qualitative ecosystem modelling, and trophic ecosystem modelling, will contribute to the development of an ecosystem-based approach to management (Thrush et al. 1997, 2000), with potential benefits for marine conservation and management. Ecosystem modelling of species distribution (and habitats) with respect to known and projected environmental parameters will improve predictability for both broad and fine-scale biodiversity distribution. This has already resulted in improved definition of environmental classifications addressing biodiversity assessment. It is also important to make progress in establishing the links between biodiversity and the long-term viability of fish stocks under various harvesting strategies. Further, modellers should consider processes from all ecosystem function perspectives, i.e., top-down effects such as predation (e.g., trophic modelling), bottom-up effects such as the environment (e.g., habitat classification based on environmental variables), and mixed effects.

Current and recently started projects are as follows:

ZBD2014-07 Southern Coralline algae habitat

Coralline algae are a structurally important component of coastal habitats, and play an important role in ecosystem processes. This project will document critical baseline information on the diversity of coralline algae in southern New Zealand using morphological and molecular identification, develop coralline reference collection from habitats and regions predicted to experience stress from ocean acidification, and prepare an identification guide for coralline algae of southern New Zealand.

ZBD2014-06 Macroalgae mapping and utility as national indicators

Kelp and fucoid algae form large underwater forests, providing three dimensional structures for fish and invertebrate species and creating canopy for other algae. Because of their central role in a range of ecological processes on temperate reefs and adjacent habitats, loss of canopy-forming algae is likely to be associated with significant loss of associated species and ecological function. Loss of subtidal macroalgal forests is an increasingly common problem in temperate marine ecosystems, particularly on urbanised coasts. This project will summarise historical use of macroalgae in national and international monitoring programmes, investigate available data for laminarians and fucoid algae in New Zealand with respect to use in monitoring programmes, distributional data and response to environmental change, and establish the utility of selected macroalgae as monitoring tools for the assessment of ecosystem health and test methods for mapping distribution for baseline monitoring.

ZBD2014-03 Sub-lethal effects of environment change on fish populations

This project (co-funded with MBIE) will investigate the potential effects of ocean acidification on New Zealand's fish and fisheries, with a focus on snapper. In 2015-16 the project team hosted a workshop of key collaborators, and this year the team have initiated a review of existing information on ocean acidification effects on fish and how these known effects are likely to play out in New Zealand's temperate setting. Following on from the review, tank

experiments will be conducted to assess the response (e.g., mortality, morphology, energy utilisation, behaviour) of snapper larvae to different acidification scenarios. Finally, the findings of the review and tank experiments will be combined in a deterministic model to assess the effects of acidification at the broader population level.

18.4.5 PROGRESS ON MARINE GENETIC BIODIVERSITY

Genetic biodiversity can be measured directly at the scale of genes and chromosomes or indirectly by measuring physical features at the organism scale (assuming that they have a genetic basis).

Genetic diversity is fundamental to the long-term survival, stability and success of a species. Central to this is the 'metapopulation' concept where populations are sufficiently genetically distinct from each other to be identifiable as individual units. A low level of recruitment between populations counters the effects of both random genetic drift and inbreeding depression of genetic diversity.

Human activities can profoundly affect genetic diversity both within populations and between populations. For example, shipping activity (movement across the globe) and aquaculture practices (transfer of organisms to different areas) can increase population connectivity such that genetic biodiversity may decrease between populations. In extreme cases, populations can become the same genetically (homogeneous) although considerable within population diversity may remain. In the event of increased genetic connectivity, a species may become more susceptible to extinction through biological or catastrophic stochasticity. That is, in the absence of between population diversity there is insufficient genetic variance to adapt to the effects of climate change, disease epidemics and so on.

In contrast, under the much more common scenario of habitat fragmentation caused by human activities (fishing, pollution), decreased connectivity between populations will result in greater between-population diversity, but a reduction of within-population diversity. This also results in a decrease in a species survival (fitness) because fragmented or isolated populations may become extinct through environmental and genetic stochasticity or localised depletion. Periodic fluctuations in annual temperature for example can lead to small-scale population extinction, which in the absence of recruitment between populations will result, over time, in the demise of all populations.

To reduce the risk of species loss, information about the genetic diversity both within populations (population isolation) and between populations (population connectivity) is needed. Without such information, the effects of perturbation on a species persistence and survival cannot be predicted. Furthermore, the links between genetic diversity, the dispersal capacity (mode of reproduction and life history development) of a species and the minimum viable population (MVP) size required in the marine environment to ensure population persistence, are little understood. For example, the MVP size for a species with a large dispersal capacity is likely to be quite different from that of a species with a relatively restricted dispersal capacity. Examining the connectivity between populations in the marine environment is fundamental to resolving some of the central challenges in ecology and has almost been ignored in the management of New Zealand fisheries and protection of biodiversity.

Understanding marine genetic diversity is also being enhanced through phylogenetic investigations of the relationships of the New Zealand marine biota using molecular sequence data. With some groups of the flora and fauna, genetic data are essential to understanding relationships and species identities. The research undertaken to date has important applications in both the documentation of diversity and in the recognition of foreign taxa (e.g., central to investigations of diversity of coralline algae in New Zealand: ZBD2001-05, ZBD2004-07; recognition of diversity: D'Archino et al. 2011; distinguishing native and foreign taxa: Heesch et al. 2009). Projects have also interpreted genetic population structure of five coastal species (tuatua, pipi, yellow-bellied flounder and sand flounder, and scallop) (ZBD2009-10) (Gardner et al. 2010, Constable 2014, Hannan 2014, Hannan et al. 2016, Silva et al. 2014), and creating a DNA database for commercial marine fish (Smith et al. 2008).

This Workstream has been integrated as appropriate within the new Workstream structure.

ZBD2013-02 Vulnerable marine ecosystems (VME) genetic connectivity

VMEs are ecosystems comprising species, communities and/or habitats that are highly vulnerable to disturbance, yet little is known about the distribution of biodiversity or

genetic relationships within and between VMEs in the deep seas surrounding New Zealand. This project clarified the spatial relationships and distribution of biodiversity of several protected invertebrate VME species within New Zealand's EEZ and beyond, using genetic analysis of specimens preserved of five species (two stony corals and three black corals) in the NIWA Invertebrate Collection (NIC).

18.4.6 PROGRESS ON EFFECTS OF CLIMATE CHANGE AND VARIABILITY ON MARINE BIODIVERSITY

Cyclical changes or trends in climate and oceanography and associated effects (such as increased ocean acidification) and how they affect the marine ecosystem as a whole have long-term implications for trophic interactions and biodiversity, as well as functional aspects of the system, e.g., biogeochemical processes. With significant improvement in remote sensing tools and global monitoring of climate change, new patterns are emerging indicating that there are long-term cycles. Examples include the Interdecadal Pacific Oscillation as well as shorter periods of change in relation to the El Niño Southern Oscillation that affect ocean ecosystems. Further, physical phenomena such as the deep subtropical gyre 'spin-up' in the South Pacific, which resulted in a warmer ocean around New Zealand from 1996–2002, and recent heat-waves in the Tasman Sea (see Climate Change Chapter 12) can have flow-on effects on ecosystem functioning.

Ocean acidification trends and projections, and implications, have been reviewed extensively in the most recent IPCC report (IPCC 2018). One key projection of increasing ocean acidification is that by 2100 some 70% of cold water corals, a key refuge and feeding ground for some commercial fish species, will be exposed to corrosive waters (see also Tracey et al. 2011b, 2013). In addition, given the current greenhouse gas emission rates, it is predicted that the surface water of the highly productive Arctic Ocean will become under-saturated with respect to essential carbonate minerals by the year 2032, and the Southern Ocean by 2050 with disruptions to large components of the marine food source, in particular those calcifying species, such as foraminifera, pteropods and coccolithophores, which rely on calcium carbonate.

Emerging research suggests that many of the effects of ocean acidification on marine organisms and ecosystems will be variable and complex and will affect different species in different ways. Evidence from naturally acidified locations confirms, however, that although some species may benefit, biological communities in acidified seawater conditions are less diverse and calcifying (calcium-reliant) species are absent whereas algae tend to dominate. BRAG funded projects have improved understanding of the impacts of ocean acidification on deep sea coral growth (ZBD2014-01), testing physiological responses (e.g., growth) to ocean acidification manipulations in a laboratory setting. Another project (ZBD2013-06) has examined shell generation and maintenance of importance aquaculture species including paua and flat oysters.

Many questions remain regarding the biological and biogeochemical consequences of ocean acidification for marine biodiversity and ecosystems, and the impacts of these changes on ecosystems and the services they provide, for example, in fisheries, coastal protection, tourism, carbon sequestration and climate regulation (Tracey et al. 2013).

Studies to predict changes in biodiversity in relation to climate change in more than a rudimentary way are beyond the state of current knowledge in New Zealand. Nevertheless, surveys of biodiversity that have occurred or are planned will provide a snapshot against which future research results or trends can be compared.

Meeting the challenges of climate change and identifying crucial issues for marine biodiversity is an area of high political interest internationally and has been identified as a gap in biodiversity research in New Zealand (Green & Clarkson 2006). A revised action plan (2016–2020) to support the New Zealand Biodiversity Strategy includes a chapter on climate change (Department of Conservation 2016).

Current and recently started projects are as follows:

ZBD2018-02 *Climate change, fish distribution meta-analysis*

Climate change effects on wild renewable marine resources essentially take place through changes in species distribution and their productivity. To date, changes in distribution seem to be the most pronounced and measurable response; for example, warming off southeast Australia with extension of the east Australian Current and associated fauna southwards by about 350 km. Other reports of changes include ranges of some warm-water species, temperature mediated impacts on benthic

invertebrates, and localised regions of warming. Understanding the shifts in New Zealand fish species in relation to fisheries management area boundaries and stock productivity is key to adaptation and management under a changing environment. This project, initiated in December 2018, will evaluate the current and future response of fisheries to changes in New Zealand waters brought about by climate change.

ZBD2018-03 *Climate variability, trends and fish population parameters*

Ongoing climate change is predicted to have both direct and indirect effects on individual organisms, including fish during all life stages, thereby affecting populations of a species, communities and the functioning of ecosystems. However, current methods of stock assessment assume that recruitment and productivity parameters such as natural mortality, steepness, von Bertalanffy growth curves, length-weight relationships, recruitment variability, and the mean number of recruits remain constant over time. This project, to be contracted in early 2019, will investigate whether this assumption is valid as climate change related impacts on the ocean progress, how parameters may change, and what changes are likely to have a significant impact on fisheries sustainability. By identifying suitable indicators to monitor significant change for important fisheries (e.g. the top 20 species by value including finfish, rock lobster and paua), individual stock assessment groups will eventually be able to assess the impacts of environmental change on stock projections.

ZBD2018-05 *Ecosystem function and regime shifts in the Subantarctic*

Anecdotal and scientific information from a range of sources suggests that there have been major environmental shifts in the Subantarctic over the past 40 years, but no meta-analyses or whole of system modeling has been conducted to determine how these shifts may be influencing different biota, including megafauna, top predators, protected species and fish. This project, contracted in November 2018, will examine all potential sources of data and develop a suitable modelling approach to identify synchronous trends, cycles, tipping points, and regime shifts in the Subantarctic. The project will also investigate the feasibility of testing or ground-truthing aspects of the model through survey work in the area. The project will develop ecosystem models to understand the effects of environmental variability and change at the scale of decades affecting ecosystem function in the Subantarctic, including effects on protected species (e.g. New Zealand sea lions, Antipodean albatross, yellow-eyed penguins, other seabirds) and on ecologically and economically important fish (e.g. hoki, squid, southern blue whiting), including potential effects on fish abundance, recruitment and spatio-temporal movements.

ZBD2016-04 Organic Carbon Recycling in Deepwater

Time incremental sediment traps from long-term moorings in subtropical and subantarctic waters north and south of the Chatham Rise have opportunistically collected and preserved mobile nektonic crustaceans over 3 to 16 day sampling periods from 2000 to 2012. This data provides a unique opportunity to examine the ecology and functioning of deep water communities in the “twilight zone”, a habitat that is poorly sampled globally. Throughout this time-series, amphipods have been a conspicuous component of the “trapped” fauna, with notable variations in abundance and speciation depending on water mass and season. The project is investigating seasonal and inter-annual cycles in community composition and abundance to elucidate whether there are biological controls that affect the observed variations between surface production and flux, and to identify pelagic consumer differences between subtropical and subantarctic “twilight zone” ocean ecosystems. These cycles underpin fisheries productivity for species such as orange roughy and oreo.

ZBD2016-03 Multiple stressors on coastal ecosystems in situ

Increasing acidity and water temperatures are two major stressors that will influence the future structure and function of coastal ecosystems. Prior research has primarily focused on the response of different faunal groups to acidified conditions in isolation. To advance understanding and the capacity to predict the future status of coastal ecosystems in New Zealand, a series of long-term mesoscale manipulation studies of coastal planktonic water column, in which pH and other parameters will be altered, will be carried out as part of aligned research in the MBIE funded CARIM Project. In this project, mesocosm experiments will provide the platform for examining the indirect effects of change in food availability and quality in the lower food web on key species such as paua and green lipped mussel.

ZBD2014-09 Climate change risks and opportunities in the marine environment, New Zealand.

The overall aim of this project is to identify risks and opportunities that are likely to arise for the seafood sector

as a consequence of climate change effects in coastal and offshore New Zealand waters. This four-year project will summarise available information on climate change impacts on fisheries in a technical summary, and in consultation with fishers and managers, develop accessible tools with which to disseminate and use this information.

18.4.7 PROGRESS ON BIODIVERSTIY METRICS AND OTHER INDICATORS FOR MONITORING CHANGE

In the mid-1990s, monitoring of marine biodiversity and the marine environment was a topic of considerable discussion, yielding several reports on developing MfE indicators. However, since the publication of MfE's indicators in 2001, a reduced set of core indicators that relate to the marine environment have been reported on (Ministry for the Environment 2007). A new international initiative launched in 2010: 'Biodiversity Indicators Partnership' (https://www.bipindicators.net) provides guidelines and examples of biodiversity indicators developed around the globe, however, Oceania does not appear to have any partnership identified. The link between this initiative and OECD environmental indicators is unclear.

A serious gap identified by Green & Clarkson (2006) in their review of progress on implementation of the NZBS was the lack of development of an integrated national monitoring system (see Biodiversity Research Programme 2010: Part 4). Efforts to respond to this gap within the Biodiversity Programme resulted in the immediate initiation of a five-year Continuous Plankton Recorder project, and a project that convened a series of workshops to determine how best to approach monitoring on a national scale (ZBD2008-14) (Hewitt et al. 2014). One objective of monitoring would be to test the effectiveness of management measures.

Current and recently started projects are as follows:

ZBD2018-01 5 year Continuous Plankton Survey (Phase 3)

The overall objective of the Continuous Plankton Recorder (CPR) series of projects is to map changes in the quantitative distribution of epipelagic plankton, including phytoplankton, zooplankton and euphausiid (krill) life stages, in New Zealand's EEZ and transit to the Ross Sea, Antarctica.

The original project was established in 2008 for a five-year period with sampling carried out annually in the Austral summer. Sanford Limited continues to provide the FV *San Aotea II* and crew to take the samples, and sample analysis is carried out by the laboratory at NIWA Christchurch. A second project further funded this long-term series (ZBD2013-03) (Robinson et al. 2014).

The current project, ZBD2018-01, continues this annual programme of CPR sampling and is funded for a further five years. This will enable a continuation of the data time series and provide a more robust dataset with which to make comparisons with the Southern Ocean CPR survey and potentially determine any trends in the plankton community. Of interest, CPR surveys have detected microplastics, showing the broad impact of these pollutants across the global oceans.

ZBD2014-04 Isoscapes for trophic studies

The main objective of this project is to develop biochemical maps of the South Pacific that can be used to understand the movements and migrations of marine animals. These maps are based on the spatial patterns of isotope values found at the base of the marine food web in different oceanic regions, driven mainly by the dominant biogeochemical provinces. This approach is based on both measured values from the open ocean and recently developed ocean models predicting these values. In addition to using isoscapes and the isotopes of marine animals to determine the movements of individuals in the open ocean, these techniques can also provide information on their diet and dietary changes in space and time.

18.4.8 PROGRESS ON IDENTIFYING THREATS AND IMPACTS TO BIODIVERSITY AND ECOSYSTEM FUNCTIONING

Many marine ecosystems in New Zealand have been modified in some way through the harvesting of marine biota, the selective reduction of certain species and size/age classes, modification of food webs, including the detritus components, and habitat destruction. Epifaunal-dominated benthic communities including seamount communities, volcanic vent communities, bryozoans, corals, hydroids and sponges, and benthic communities dominated by infaunal structure (e.g. shellfish beds, tub-forming polychaetes or amphipods, large burrowing infauna) are vulnerable to human disturbance (Lundquist et al. 2017). The mechanical disturbance of marine habitats that occurs with some activities such as trawling, dredging,

dumping, and oil, gas and mineral exploration and extraction can substantially change the structure and composition of benthic communities. Indirect effects of mechanical disturbance (e.g. sediment resuspension, change in sediment grainsize) also impact on marine communities and their ecological functioning. The invasion of alien species into New Zealand waters is also a real threat, with evidence of nuisance species already well established.

A number of inshore marine ecosystems (especially estuaries and other sheltered waters) have been modified by sediment, contaminants and nutrients derived from human land-use activities (Morrison et al. 2009). Coastal margin development has had a major impact on some inshore marine communities.

A project commissioned by the Aquatic Environment Programme, which identifies key threats to the marine environment (BEN2007-05) is complete and has listed and ranked the top threats to New Zealand's marine environment, as perceived by expert opinion. Relevant findings are that the highest ranking threats are ocean acidification, increasing sea water temperatures and bottom trawling (across all habitats) and that the most threatened habitats are intertidal reef systems in harbours and estuaries (MacDiarmid et al. 2012). Ecological risk assessment (ERA) methods have also been reviewed (under ENV2005-15, Rowden et al. 2008), and a trial Level 2+ assessment completed on Chatham Rise seamounts to estimate the relative risk to seamount benthic habitat from bottom trawling (under ENV2005-16, Clark et al. 2011). An MPI project (DEE2010-04) has resulted in a new ecological risk assessment being developed that is tailored for New Zealand deep water fisheries.

An existing spatially explicit patch dynamic model was expanded upon as a framework to explore effects of disturbance on functional diversity in benthic marine ecosystems, and ultimately, other elements of biodiversity and ecosystem function (such as the abundance of rare species, ecosystem productivity, and the provisioning of biogenic habitat structure) (Lundquist et al. 2013). The model was validated against available inshore (Tasman and Golden Bays) and offshore (Chatham Rise and Challenger Plateau) empirical datasets, demonstrating the value of this tool for investigating disturbance and recovery dynamics in seafloor communities.

18.4.9 PROGRESS ON BIODIVERSITY IN ANTARCTICA

Antarctic research funded by BRAG (historically known as BioRoss) aimed to improve understanding of the biodiversity and functional ecology of selected marine communities in the Ross Sea. The research links directly with ongoing Ross Sea ecosystems research through the Antarctic Working Group, and supporting climate change related research, especially at high latitudes.

Data acquisition from the Antarctic marine environment is logistically difficult and expensive. The BioRoss survey in 2004 and the Latitudinal Gradient Project ICECUBE have provided significant new information on biodiversity, species abundance and distribution that are now facilitating research into functional ecology and longer-term monitoring programmes. This research has the potential to lead into other research on genetic diversity, climate variability and the development of indicators. The research results are also being used in the Fisheries New Zealand Antarctic Research Programme projects on ecosystem modelling of the Ross Sea.

The Fisheries New Zealand Antarctic Research and BioRoss Programmes are also directly involved in supporting the development of protection measures around the Balleny Islands. In 2005 Ministry of Fisheries scientists and Ministry of Foreign Affairs and Trade (MFAT) personnel prepared a paper for submission to CCAMLR justifying MPA designation around the islands to protect ecosystem processes occurring there that may be important for the stability and function of the wider Ross Sea regional ecosystem.

To collect data in support of the MPA proposal, Ministry of Fisheries BioRoss funded a targeted research voyage to the Balleny Islands in February 2006 (ZBD2005-01) (Rowden et al. 2013), and also provided supplementary funding to carry out opportunistic biological sampling at the Balleny Islands on a voyage to the Ross Sea that was primarily funded by LINZ to do bathymetric mapping. The field sampling of these projects were successful, both providing important data and specimens from the Balleny Islands area and supplementary information for the Antarctic Working Group Research Programme.

After a number of unsuccessful proposals for a Ross Sea MPA, CCAMLR agreed in 2016 to implement a 1.55 million km^2 MPA, implemented from 1 December 2017, until the

end of its review period in 2052, at which time it may be renewed or modified as needed. An MBIE proposal was funded in 2017 to address monitoring and evaluation of the success of this MPA.

In addition, BioRoss funded a further ICECUBE project to sample the Antarctic coastline during the summer season of 2006–07 (ZBD2006-03). ICECUBE is a key part of the international Latitudinal Gradient Project to explore hypotheses about environmental drivers of structure and function in sub-tidal ecosystems along the western Ross Sea coastline (Cummings et al. 2008). This project acquired funding for three seasons (2007–08, 2008–09, 2009–10) as part of the MBIE IPY contestable round (see also Cummings et al. 2011 and Thrush & Cummings 2011). Published reports and papers from the Ministry of Fisheries Ross Sea coastal projects include Cummings et al. 2003, 2006b, 2008, 2010, 2011, De Domenico et al. 2006, Grotti et al. 2008, Guidetti et al. 2006, Norkko et al. 2002, 2004, 2005, 2007, Pinkerton et al. 2006, Schwarz et al. 2003, 2005, Sharp et al. 2010, Sutherland 2008, Thrush et al. 2006, 2010).

The New Zealand government provided one-off funding for a Census of Antarctic Marine Life (CAML) survey to the Ross Sea from RV *Tangaroa* as part of New Zealand's involvement in the 2007–08 International Polar Year activities. The CAML Voyage was a large cooperative research effort under the banner of Ocean Survey 20/20 with considerable international collaboration, simultaneously utilising a number of different vessels with different strengths and capabilities.

There are no current projects in the BRAG Workstreams located in Antarctica.

18.5 EVALUATION OF RESEARCH EFFECTIVENESS

Given that the Fisheries New Zealand Biodiversity Programme has been running for more than 18 years, and that a number of new strategic documents and directions are emerging across government, here the programme is evaluated with respect to its initial strategic outcomes, and its alignment with more recent strategic documents.

In 2000, five strategic outcomes were built into the Ministry of Fisheries Biodiversity Research Programme.

That by 2010:

i. *the Ministry of Fisheries Biodiversity programme will have become an integral part of the research effort devoted to understanding New Zealand's marine environment.*
ii. *research planning will benefit from close cooperative relationships within the Ministry of Fisheries, with other government agencies, and with external stakeholders.*
iii. *mutually beneficial collaborative research projects will be carried out alongside other New Zealand and international research providers, especially for vessel-based research.*
iv. *Ministry of Fisheries Biodiversity projects will have contributed substantially to an improved understanding of New Zealand's marine biodiversity and its role in marine ecosystem function, yielding scientifically rigorous outputs for a national and international professional audience.*
v. *results generated by MFish Biodiversity projects will be incorporated into management policy, with clear benefits for the New Zealand marine environment.*

The Biodiversity Programme has been highly effective in delivering on the first four and part of the fifth of these five outcomes, though currently there is no clear metric to evaluate the measure of whether the Programme is providing *'clear benefits for the New Zealand marine environment'*. In recent years, significant whole-of-government projects have been administered through the programme, and one-off funding applications made jointly with other stakeholders have been successful. The programme has made a significant contribution to increasing understanding about biodiversity in the marine environment. Achievements in each outcome are addressed below.

i. *Has the Biodiversity Research Programme become integrated with New Zealand's research effort to understand the marine environment?*

Seven science objectives were developed by multiple stakeholders through the Biodiversity Research Advisory Group. The original objectives included: 1) ecosystem-scale

studies in the New Zealand marine environment, 2) the classification and characterisation of the biodiversity of near-shore and offshore marine habitats, 3) the role of biodiversity in the functional ecology of marine communities, 4) connectivity and genetic marine biodiversity, 5) the assessment of the effects of climate change and increased ocean acidification, 6) identification of indicators of biodiversity that can be used to monitor change, and 7) identification of key threats to biodiversity, identification of threats and impacts to biodiversity and ecosystem functioning beyond natural environmental variation.

Projects ranged from localised experiments on seabed communities of shellfish and echinoderms, to integrated studies of rocky reef systems (Schwarz et al. 2006) and offshore fishery-scale trophic studies. The effects of ocean climate change (temperature, acidification) have been explored on shellfish, rhodolith communities, plankton productivity and the microbial productivity engines of polar waters. A major project to investigate shelf communities in relation to climate over the past 1000 years has resulted in the development of new methods, and insights into past changes and the human impact on New Zealand's marine environment.

A total of 74 projects were commissioned and managed within this 18-year period, yielding over 100 final research reports, most of which have been published through Ministry publications (Marine Biosecurity and Biodiversity Reports and Aquatic Environment and Biodiversity Reports), books, identification guides and mainstream scientific literature. Additional publications continue to be added to the scientific literature. In addition, several workshops have been run through the Programme, including on qualitative modelling techniques, how to set up a marine monitoring programme and predictive modelling. A large number of science providers, including NIWA, Cawthron Institute, University of Auckland, Auckland University of Technology, University of Waikato, Victoria University of Wellington, University of Otago, University of Canterbury and Massey University have been directly commissioned or sub-contracted to take part in or conduct research projects through the Programme during the 10-year period. For some, the projects have provided critical synergies with aligned research, while others have provided one-off opportunities for marine biodiversity investigation or opportunistic leveraging for research voyages.

Research into the biodiversity of habitats such as seamounts has been completed and new methods to assess the vulnerability of seabed habitats have been developed. The land-sea interface is being investigated and projects have shown how land use in a given catchment can affect nutrient transfer and living conditions, and impact the diversity and functioning, of estuarine and coastal organisms. Publication and presentation of the results from these projects has resulted in widespread contribution to the development of marine science in New Zealand. Partnership with overseas researchers and presentations to international meetings and conferences has added to the growing global initiatives on marine biodiversity research questions.

Feedback from stakeholders has indicated that the move to a five-year research planning horizon was welcomed by research providers, but some stakeholders felt that Requests for Proposals should be at a higher level than individual projects to safeguard intellectual property on new ideas and methods.

ii. Does research planning now benefit from close cooperative relationships within the Ministry of Fisheries, with other government agencies, and with external stakeholders?

The Biodiversity Programme is highly cooperative. Of 38 projects underway in the last five years, 14 have formal collaborative components across government departments, with other stakeholders or multiple research providers and 10 have formal linkages to international research programmes. Within the Ministry and with other stakeholders (NGOs, industry, other government departments), the Biodiversity Projects have contributed to discussions about Marine Stewardship Council (MSC) certification, to decision papers on aspects of Antarctic management under CAMLR, fulfilling Ministry commitments to the New Zealand Biodiversity Strategy, and to Ministry progress towards recognising the role of the ecosystem in underpinning sustainable and healthy fisheries production. There are many other examples, e.g., the Programme has contributed towards DOC and Ministry decisions on marine protected areas. The interaction at the research and policy advice stages of resource management feeds back into the BRAG planning for future research.

There are close links with the Fisheries New Zealand Aquatic Environment research programme, the National Aquatic Biodiversity Information System (NABIS), a web-

based interactive data access and mapping tool, and the Fisheries New Zealand Antarctic Research programme. These and other links have enabled contributions resulting from progress on land-sea interface research, habitats of significance to fisheries management, trophic studies (MSC Certification), climate change (effects on shellfish) and habitat classification (fish optimised MEC, testing of MEC and BOMEC). The successful involvement of the Biodiversity Programme in major whole-of-government projects such as Ocean Survey 20/20 and IPY-CAML, has also raised the profile of Fisheries New Zealand and the research it has commissioned both across New Zealand and internationally.

Datasets, voucher specimens and samples from all biodiversity research projects have resulted in a substantial amount of material that has been physically preserved and housed in the Te Papa Fish Collection and NIWA National Invertebrate Collection, and Herbarium (macroalgae). All data are held in databases either at Fisheries New Zealand, NIWA or Te Papa, and accessibility is being improved. The recent Bay of Islands Ocean Survey 20/20 Portal was very well received and nominated for New Zealand Government Open Source awards. It will also incorporate data access from Chatham Challenger and IPY projects. Data from a number of Ministry biodiversity projects have also been entered into international biodiversity databases such as OBIS and from there into the Global Biodiversity Information Facility (GBIF).

Biodiversity Research planning receives regular input from DOC, SeaFIC, MfE, Cawthron Institute, NIWA, GNS, LINZ, MAFBNZ, Te Papa, University of Auckland, AUT, University of Otago, MoRST, MFAT, Regional Councils and others. Research planning for 2018–19 and beyond has included a re-alignment of the current research programme to take account of new developments such as The Future of Our Fisheries, MfE's environmental reporting programme, DOC's integrated marine protected area monitoring programme, and international commitments such as the CBD COP10 Aichi-Nagoya Agreement.

Feedback and support for projects by external stakeholders has shown that the Programme has been effective in promoting inter-agency collaboration.

iii. Have mutually beneficial collaborative research projects been carried out alongside other New Zealand and international research providers, especially for vessel-based research?

As discussed above, collaborative research projects across government and among research providers have resulted in many mutually beneficial data and specimen collections, surveys of New Zealand marine biodiversity in New Zealand Territorial Seas, the EEZ and the Ross Sea, groundbreaking research into seamount biodiversity and the identification of VMEs, and research for international collaboration, particularly vessel-based studies. Large-scale vessel-dependent oceanic research projects have made significant gains in baseline knowledge about the distribution and abundance of biodiversity in the EEZ/Ross Sea region. Vessel-based projects include: NORFANZ (Norfolk Island-Australia-New Zealand survey of biodiversity on Norfolk Ridge and Lord Howe Rise); BioRoss (MFish-LINZ, first New Zealand survey of biodiversity in the Ross Sea); Chatham-Challenger (LINZ-MFish-NIWA-DOC first Ocean Survey 20/20 project); NZ IPY-CAML (MFish-LINZ-NIWA; with international and New Zealand-wide collaboration) survey of the Ross Sea as part of International Polar Year; and biodiversity of seamounts (MFish-NIWA-LINZ-MBIE voyages to the Kermadec Arc and on the Chatham Rise). These projects have generated huge geo-referenced datasets and thousands of specimens for Te Papa and National Invertebrate Collections. They have also resulted in the identification of new species, new genera and new families, as well as new records extending the known distribution of species. These surveys have contributed to habitat classification, identified areas of high biodiversity and challenged paradigms on the environmental drivers that determine biodiversity. More recently they have provided new information on the effects of ocean acidification on the productivity of polar seas, and in New Zealand waters.

Vessel-dependent coastal projects have also generated significant new understanding about the distribution of inshore biota, and the role they play in maintaining a healthy ecosystem. Experimental field work on the productivity of the seabed has been carried out in New Zealand waters (Fiordland, Otago, Bay of Islands, Hauraki Gulf, Kaipara and Manukau Harbours), and along the west coast of the Ross Sea. The impact of land practices on the land-sea interface has also highlighted real downstream effects on the productivity of the coastal environment. These projects have provided new insights into the connectivity between different species groups, and data are being used in a number of ways to assist with spatial planning by RMAs.

Feedback from stakeholders has indicated that the collaborative voyages administered through the Programme have successfully created synergy and opportunity for New Zealand scientists as well as facilitating new international collaborations.

iv. *Have MFish Biodiversity projects contributed substantially to an improved understanding of New Zealand's marine biodiversity and its role in marine ecosystem function, yielding scientifically rigorous outputs for a national and international professional audience?*

In the early years, the Programme focused primarily on taxonomy and the description of marine biodiversity. As the Programme matured, projects to address biodiversity roles in ecosystem function were introduced. Some were experimental and on a local scale while others were on a regional scale. Recent projects have addressed patterns of marine biodiversity in relation to environmental drivers with ecosystem function. This enabled modelling to predict the distribution of biodiversity in unsurveyed areas of ocean, and evaluation of the vulnerability of biodiversity to perturbations such as climate change, as well as the modelling of trophic interactions among key fish species. Presentations of research results have been made to numerous overseas and New Zealand science audiences, and publications in the mainstream literature have been encouraged.

v. *Have results generated by Ministry of Fisheries Biodiversity projects been incorporated into management policy, with clear benefits for the New Zealand marine environment?*

Examples of incorporation into management policy with clear benefits for the marine environment include the increased awareness of research topics initiated in the biodiversity programme by policy analysts to core Aquatic Environment research projects and Fishery Plans (land-use effects, climate change in the ocean, habitat classification); links to the Antarctic research programme and uptake into CCAMLR (ecotrophic studies, ecosystem baselines, VME risk assessment, bioregionalisation), spatial management (seamount closures, BPAs, MPAs, RMAs); and the need by MfE to report on the marine environment at a national scale (plankton recording programme, Marine Environmental Monitoring Programme). Fisheries New Zealand biodiversity advice is frequently requested to contribute to cross-government initiatives including Ocean Survey 20/20, DOC Sub-Antarctic Islands Forum National Monitoring, Statistics New Zealand Tier 1 statistic review and Environmental Domain Stocktake, International Year of Biodiversity, OECD and CBD reports, International Oceans Issues, SPRFMO, NRS marine issues paper, the Antarctic Science Framework, Ocean Fertilisation and IPCC. Finally, the Programme has contributed to New Zealand's efforts in the international Census of Marine Life and an ongoing assessment of New Zealand's progress in Marine Biodiversity has been proposed as a new Tier 1 Environmental Statistic.

However, the benefits to the marine environment are more inferred than demonstrated. There is substantially increased awareness within Fisheries New Zealand and across government, that the health of fisheries and other valued uses of the sea depend on intact ecosystem services provided by the diversity of organisms, the diversity of habitats and the genetic diversity found in the marine environment. Statements of intent and long-term strategic documents such as Fisheries 2030 and Fish Plans have had biodiversity protection and an ecosystem approach to fisheries management objectives explicitly stated. Future research questions will also need to address follow-up of management decisions to assess whether and to what extent the objectives have been achieved.

18.6 CONCLUDING REMARKS

Since its inception 18 years ago, the research in this Biodiversity Programme has evolved from defining and mapping the biological diversity of the sea, to elucidating the role of biodiversity in marine ecosystem function, threats to these roles and how best biodiversity and its successful protection can be measured. Huge advances have been made in providing new identification tools for major groups (e.g., Coralline algae), and progress has been made in raising the profile of biodiversity in coastal and ocean environmental management, in particular fisheries management, and biodiversity research uptake into policy and management decisions within Fisheries New Zealand and across government, in particular within new environmental legislation such as the Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012) and the Environmental Reporting Act (2015).

As the CBD updates the Aichi-Nagoya Agreement with new post-2020 Biodiversity strategy and associated biodiversity and environmental targets, New Zealand must continue to

progress our understanding of the marine biodiversity and the ecosystem services provided by the marine environment in order to inform national and international decision-making. This will continue to require a coordinated approach across government to link science to policy needs. Essentially we need to know four things: what is out there in the marine environment to use, protect, or manage; how does the ecosystem function; what are the impacts of natural- and human-induced changes; and what tools will allow for effective monitoring and management of environmental change? For example, there is a compelling need for large-scale projects such as mapping seafloor habitats and establishing long-term nationwide monitoring and reporting schemes to measure the effects of ocean climate change, regular assessment of the cumulative effects of anthropogenic activities and multiple stressors in the ocean and the effectiveness of their management. Without these, we face the risks that New Zealand's 'green' branding will be increasingly challenged, and that tipping points in the health of the aquatic environment may be reached before evasive action can be taken.

18.7 REFERENCES

Anderson, O F; Bagley, N W; Hurst, R J; Francis, M P; Clark, M R; McMillan, P J (1998) Atlas of New Zealand fish and squid distributions from research bottom trawls. *NIWA Technical Report 42*. 303 p.

Anon (2000) The New Zealand Biodiversity Strategy. Our chance to turn the tide. Department of Conservation and Ministry for the Environment. 146 p.

Appeltans, W, et al. (2012) The magnitude of global marine species diversity. *Current Biology 22*: 1–14.

Arnold, A (Ed.) (2004) Shining a spotlight on the biodiversity of New Zealand's Marine Ecoregion. World Wildlife Fund New Zealand, Wellington, 85 p.

Azam, F; Malfatti, F (2007) Microbial structuring of marine ecosystems. *Nature Reviews in Microbiology* 5: 782.

Bagley, N W; Anderson, O F; Hurst, R J; Francis, M P; Taylor, P J (2000) Atlas of New Zealand fish and squid distributions from midwater trawls, tuna longline sets, and aerial sightings. *NIWA Technical Report 72*. 171 p.

Battley, P F; Melville, D S; Schuckard, R; Ballance, P F (2005) Quantitative survey of the intertidal benthos of Farewell Spit, Golden Bay. *Marine Biodiversity Biosecurity Report No. 7*. 19 p.

Beaumont, J; MacDiarmid, A; D'Archino, R (2010) Mapping the environmental values of New Zealand's marine ecosystem: A meta-analysis. *Biosecurity New Zealand Technical Paper 2010*: 1–76.

Beaumont, J; Oliver, M; MacDiarmid, A (2008) Mapping the values of New Zealand's coastal waters. 1. Environmental values. *Biosecurity New Zealand Technical Paper 2008/16*. 89 p.

Blasiak, R; Spijkers, J; Tokunaga, K; Pittman, J; Yag,i N; Österblom, H (2017) Climate change and marine fisheries: Least developed countries top global index of vulnerability. *PLoS ONE 12 (6)*: e0179632. doi:10.1371/journal.pone.017963.

Bostock, H; Tracey, D; Cummings, V; Mikaloff-Fletcher, S; Williams, M; Guy, C; Neil, H; Currie, K (2012) Ocean acidification in fisheries habitat (ZBD201041). Research Progress Report prepared for the Ministry of Primary Industries. 39 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Bowden, D A (2011) Benthic invertebrate samples and data from the Ocean Survey 20/20 voyages to the Chatham Rise and Challenger Plateau, 2007. *New Zealand Aquatic Environment and Biodiversity Report No. 65*. 40 p.

Bowden, D A; Compton, T J; Snelder, T H; Hewitt, J E (2011a) Evaluation of the New Zealand Marine Environment Classifications using Ocean Survey 20/20 data from Chatham Rise and Challenger Plateau. *New Zealand Aquatic Environment and Biodiversity Report No. 77*. 27 p.

Bowden, D A; Hewitt J (2012) Recommendations for surveys of marine benthic biodiversity: outcomes from the Chatham-Challenger Ocean Survey 20/20 Post-Voyage Analyses Project. *New Zealand Aquatic Environment and Biodiversity Report No. 91*. 34 p.

Bowden, D; Hewitt, J; Lundquist, C (2011b) Chatham-Challenger Ocean Survey 20/20 Post Voyage Analyses: Diversity Metrics. *New Zealand Aquatic Environment and Biodiversity Report No. 83*. 61 p.

Bowden, D A; Hewitt, J; Verdier, A-L; Pallentin, A (2014) Project ZBD2007-01 Objective 14. The potential of multibeam echosounder data for predicting benthic invertebrate assemblages across Chatham Rise and Challenger Plateau. *New Zealand Aquatic Environment and Biodiversity Report No. 126*. 35 p.

Bowden, D A; Schiaparelli, S; Clark, M R; Rickard, G J (2011c) A lost world? Archaic crinoid-dominated assemblages on an Antarctic seamount. *Deepsea Research Part II. Topical Studies in Oceanography 58*: 119–127.

Brown, C J; Fulton, E A; Hobday, A J; Matear, R J, et al. (2010) Effects of climate-driven primary production change on marine food webs: implications for fisheries and conservation. *Global Change Biology 16 (4)*: 1194–1212.

Capson, T L; Guinnotte, J (2014) Future proofing New Zealand's shellfish aquaculture: monitoring and adaptation to ocean acidification. *New Zealand Aquatic Environment and Biodiversity Report No. 136*. 42 p.

Carroll, E; Jackson, J A; Paton, D; Smith, T (2015) Estimating 19th and 20th century right whale catches and removals around east Australia and New Zealand. *New Zealand Fisheries Assessment Report 2015/04*.

Cheung, L; Lam, V W Y; Sarmiento, J L; Kearney, K; Watson, R; Pauly, D (2009) Projecting global marine biodiversity impacts under climate change scenarios. *Fish and Fisheries 10 (3)*: 235–251.

Clark, M R; Bowden, D A; Baird, S J; Stewart, R (2010a) Effects of fishing on the benthic biodiversity of seamounts of the 'Graveyard' complex, northern Chatham Rise. *New Zealand Aquatic Environment and Biodiversity Report No. 46*. 40 p.

Clark, M R; Dunn, M R; McMillan, P J; Pinkerton, M H; Stewart, A; Hanchet, S M (2010b) Latitudinal variation of demersal fish assemblages in the western Ross Sea. CCAMLR document WG-FSA-10/P3. 13 p. (Antarctic Science, 2010. doi: 10.1017/S0954102010000441).

Clark, M R; Leduc, D; Nelson, W; Mills, V (2014) Benthic invertebrate data within New Zealand's Benthic Protection and Seamount Closure Areas. *NIWA Client Report WLG2014-43*. 22 p.

Clark, M R; Roberts, C D (2008) Fish and invertebrate biodiversity on the Norfolk Ridge and Lord Howe Rise, Tasman Sea (NORFANZ voyage, 2003). *New Zealand Aquatic Environment and Biodiversity Report No. 28*. 131 p.

Clark, M R; Rowden, A A (2009) Effect of deepwater trawling on the macro-invertebrate assemblages of seamounts on the Chatham Rise, New Zealand. *Deepsea Research I 56*: 1540–1554.

Clark, M R; Tittensor, D P (2010) An index to assess the risk to stony corals from bottom trawling on seamounts. *Marine Ecology 31 (supplement 1)*: 200–211.

Clark, M R; Tracey, D M; Pallentin, A; Schnabel, K; Anderson, O F; Bowden, D (2009) Voyage report of a survey of 'seamounts' on the northwest and southeast Chatham Rise (TAN0905). 49 p. (Unpublished report available from NIWA, Wellington.)

Clark, M R; Watling, L; Rowden, A A; Guinotte, J M; Smith, C R (2010c) A global seamount classification to aid the scientific design of marine protected area networks. *Journal of Ocean and Coastal Management 54*: 19–36.

Clark, M R; Williams, A; Rowden, A A; Hobday, A J; Consalvey, M (2011) Development of seamount risk assessment: application of the ERAEF approach to Chatham Rise seamount features. *New Zealand Aquatic Environment and Biodiversity Report No. 74*. 18 p.

Compton, T J; Bowden, D A; Pitcher, C R; Hewitt, J E; Ellis, N (2012) Biophysical patterns in benthic assemblage composition across contrasting continental margins off New Zealand. *Journal of Biogeography 40 (1)*: 75–89. doi: 10.1111/j.1365-2699.2012.02761.x.

Constable H B (2014) Population structure, temporal stability and seascape genetics of two endemic New Zealand Pleuronectids, *Rhombosolea plebeia* (sand flounder) and *R. leporina* (yellowbelly flounder). PhD thesis, Victoria University of Wellington.

Costello, M J; Coll, M; Danovaro, R; Halpin, P; Ojaveer H; Miloslavich, P (2010) A Census of Marine Biodiversity Knowledge, Resources, and Future Challenges. *PLoS ONE 5 (8)*: e12110. doi: 10.1371/journal.pone.0012110.

Coutts, A D M; Dodgshun, T J (2007) The nature and extent of organisms in vessel sea-chests: a protected mechanism for marine bioinvasions. *Marine Pollution Bulletin 54*: 875–886.

Cranfield, H J; Manighetti, B; Michael, K P; Hill, A (2003) Effects of oyster dredging on the distribution of bryozoan biogenic reefs and associated sediments in Foveaux Strait, southern New Zealand. *Continental Shelf Research 23*: 1337–1357.

Cryer, M; Hartill, B; O'Shea, S (2002) Modification of marine benthos by trawling: toward a generalization for the deep ocean? *Ecological Applications 12*: 1824–1839.

Cummings, V; Hewitt, J; Van Rooyen, A; Currie, K, et al. (2011) Ocean acidification at high latitudes: potential effects on functioning of the Antarctic bivalve *Laternula elliptica*. *PLoS ONE 6 (1)*: e16069.

Cummings, V; Thrush, S; Andrew, N; Norkko, A; Funnell, G; Budd, R; Gibbs, M; Hewitt, J S; Mercer, S; Marriott, P; Anderson, O (2003) Ecology and biodiversity of coastal benthic communities in McMurdo Sound, Ross Sea: emerging results. Final Research Report for Ministry of Fisheries Research Project ZBD2002/01 Objectives 1 & 2. National Institute of Water and Atmospheric Research. 105 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Cummings, V J; Thrush, S; Chiantore, M; Hewitt, J; Cattaneo-Vietti, R (2010) Macrobenthic communities of the north-western Ross Sea shelf: links to depth, sediment characteristics and latitude. *Antarctic Science 22*: 793–804.

Cummings, V J; Thrush, S F; Marriott, P M; Funnell, G A; Norkko, A; Budd, R G (2008) Antarctic coastal marine ecosystems (ICECUBE). Final Research Report for Ministry of Fisheries Research Project ZBD2006-03. 73 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Cummings, V J; Thrush, S F; Norkko, A; Andrew, N L; Hewitt, J E; Funnell, G A; Schwarz, A-M (2006a) Accounting for local scale variability in benthos: implications for future assessments of latitudinal trends in the coastal Ross Sea. *Antarctic Science 18 (4)*: 633–644.

Cummings, V; Thrush, S; Schwarz, A-M; Funnell, G; Budd, R (2006b) Ecology of coastal benthic communities of the northwestern Ross Sea. *New Zealand Aquatic Environment and Biodiversity Report No. 6*. 67 p.

Currey, R J C; Boren, L J; Sharp, B R; Peterson, D (2012) A risk assessment of threats to Maui's dolphins. Ministry for Primary Industries and Department of Conservation, Wellington. 51 p.

D'Archino, R; Nelson, W A; Zuccarello, G C (2011) Diversity and complexity in New Zealand Kallymeniaceae (Rhodophyta): recognition of the genus *Ectophora* and description of *E. marginata sp. nov. Phycologia 50*: 241–255.

Davies, K; Fisher, K; Foley, M; Greenaway, A; Hewitt, J; Le Heron, R; Mikaere, H; Ratana, K; Spiers, R; Lundquist, C (2018a) Navigating collaborative networks and cumulative effects for Sustainable Seas. *Environmental Science & Policy* 83:22–32.

Davies, K; Murchie, A A; Kerr, V; Lundquist, C (2018b) The evolution of marine protected area planning in Aotearoa New Zealand: Reflections on participation and process. *Marine Policy* 93:113–127.

De Domenico, F; Chiantore, M; Buongiovanni, S; Paola Ferranti, M; Ghione, S; Thrush, S; Cummings, V; Hewitt, J; Kroeger, K; Cattaneo-Vietti, R (2006) Latitude versus local effects on echinoderm assemblages along the Victoria Land coast, Ross Sea, Antarctica. *Antarctic Science 18 (4)*: 655–662.

de Juan, S; Hewitt, J; Thrush, S; Freeman, D (2015) Standardising the assessment of functional integrity in benthic ecosystems. *Journal of Sea Research 98*: 33-41.

Department of Conservation (2010) New Zealand Coastal Policy Statement 2010. Retrieved from http://www.doc.govt.nz/Documents/conservation/marine-and-coastal/coastal-management/nz-coastal-policy-statement-2010.pdf

Department of Conservation (2016) New Zealand Biodiversity Action Plan 2016–2020. Retrieved from https://www.doc.govt.nz/globalassets/documents/conservation/new-zealand-biodiversity-action-plan-2016-2020.pdf

Eakin, R A; Eastman, J T; Near, T J (2009) A new species and a molecular phylogenetic analysis of the Antarctic Fish Genus *Pogonophryne* (Notothenioidei: Artedidraconidae). *Copeia 4*: 705–713.

Farr, T; Broom, J; Hart, D; Neill, K; Nelson, W (2009) Common coralline algae of northern New Zealand: an identification guide. *NIWA Information Series No. 70.*

Fisheries New Zealand (2018) Report from the Fisheries Assessment Plenary, May 2018: stock assessments and yield estimates. Compiled by the Fisheries Science Group, Fisheries New Zealand, Wellington, New Zealand.

Floerl, O; Hewitt, J (2012) Chatham-Challenger OS 20/20 Post Voyage Analyses: Objective 9 – Patterns in species composition. *New Zealand Aquatic Environment and Biodiversity Report No. 97*. 40 p.

Freeman, D J; Marshall, B A; Ahyong, S T; Wing, S R; Hitchmough, R A (2010) The conservation status of New Zealand marine invertebrates, 2009. *New Zealand Journal of Marine and Freshwater Research 44*: 129–148.

Freeman, D; Schnabel, K; Marshall, B; Gordon, D; Wing, S; Tracey, D; Hitchmough, R (2013) Conservation status of New Zealand marine invertebrates, 2013. *New Zealand Threat Classification Series 9.* Department of Conservation, Wellington. 20 p.

Gardner, J P A; Bell, J J; Constable, H B; Hannan, D; Ritchie, P A; Zuccarello, G C (2010) Multi-species coastal marine connectivity: a literature review with recommendations for further research. *New Zealand Aquatic Environment and Biodiversity Report No. 58*. 47 p.

Gordon, D P (Ed.) (2009) New Zealand Inventory of Biodiversity Volume 1. Kingdom Animalia: Radiata, Lophotrochozoa, Deuterostomia. Canterbury University Press, Christchurch. 568+16 p.

Gordon, D P (Ed.) (2010) New Zealand Inventory of Biodiversity Volume 2. Kingdom Animalia: Chaetognatha, Ecdysozoa and Ichnofossils. Canterbury University Press, Christchurch. 528+16 p.

Gordon, D P (Ed.) (2012) New Zealand Inventory of Biodiversity Volume 3. Kingdoms Bacteria, Protozoa, Chromista, Plantae, Fungi. Canterbury University Press, Christchurch. 616+16 p.

Gordon, D P; Beaumont, J; MacDiarmid, A; Robertson, D A; Ahyong, S T (2010) Marine Biodiversity of Aotearoa New Zealand. *PLoS ONE 5 (8)*: e10905. doi:10.1371/journal.pone.0010905.

Gould, B; Ahyong, S T (2008) Marine Invasive Taxonomic Service. *Biosecurity 85*: 18–19.

Grange, K R; Tovey, A; Hill, A E (2003) The spatial extent and nature of the bryozoan communities at Separation Point, Tasman Bay. *Marine Biodiversity Biosecurity Report No. 4*. 22 p.

Green, W; Clarkson, B (2006) Review of the New Zealand Biodiversity Strategy Themes. 242 p. Retrieved from http://doc.govt.nz/documents/conservation/nz-biodiversity-strategy-themes.pdf.

Grotti, M; Soggia, F; Lagomarsino, C; Dalla Riva, S; Goessler, W; Francesconi, K A (2008) Natural variability and distribution of trace elements in marine organisms from Antarctic coastal environments. *Antarctic Science 20 (1)*: 39–51.

Guidetti, M; Marcato, S; Chiantore, M; Patarnello, T; Albertelli, G; Cattaneo-Vietti, R (2006) Exchange between populations of *Adamussium colbecki* (Mollusca: Bivalvia) in the Ross Sea. *Antarctic Science 18 (4)*: 645–653.

Guinotte, J M; Bartley, J D; Iqbal, A; Fautin, D G; Buddemeier, R W (2006) Modelling habitat distribution from organism occurrences and environmental data: case study using anemonefishes and their sea anemone hosts. *Marine Ecology Progress Series 316*: 269–283.

Hanchet, S M (Comp.) (2009) New Zealand IPY-CAML Progress Report. August 2009. 76 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Hanchet, S M (Comp.) (2010) New Zealand IPY-CAML Progress Report. July 2010. (Unpublished report held by Fisheries New Zealand, Wellington.)

Hanchet, S M; Fu, D; Dunn, A (2008a) Indicative estimates of biomass and yield of Whitson's grenadier (*M. whitsoni*) on the continental slope of the Ross Sea in Subareas 88.1 and 88.2. WG-FSA-08/32.

Hanchet, S M; Mitchell, J; Bowden, D; Clark, M; Hall, J; O'Driscoll, R (2008b) Ocean survey 20/20: New Zealand IPY-CAML Final Voyage Report. *NIWA Client Report WLG2008-74*. 193 p.

Hanchet, S M; Mitchell, J; Bowden, D; Clark, M; Hall, J; O'Driscoll, R; Pinkerton, M; Robertson, D (2008c) Preliminary report of the New Zealand RV *Tangaroa* IPY-CAML survey of the Ross Sea region, Antarctica in February–March 2008. Unpublished NIWA report to CCAMLR working group on ecosystem monitoring and management. WG-EMM-08/18. 15 p.

Hanchet, S M; Stevenson, M L; Jones, C; Marriott, P M; McMillan, P J; O'Driscoll, R L; Stevens, D; Stewart, A L; Wood, B A (2008d) Biomass estimates and size distributions of demersal finfish on the Ross Sea shelf and slope from the New Zealand IPY-CAML survey, February–March 2008. WG-FSA-08/31.

Hanchet, S M; Stewart, A L; McMillan, P J; Clark, M; O'Driscoll, R L; Stevenson, M L (2013) Diversity, relative abundance, new locality records, and updated fish fauna of the Ross Sea, Antarctica. *Antarctic Science 25*: 619–636. doi:10.1017/S0954102012001265.

Hannan, D A (2014) Population genetics and connectivity in *Paphies subtriangulata* and *Paphies australis* (Bivalvia: Mesodesmatidae). PhD thesis, Victoria University of Wellington.

Hannan, D A; Constable, H B; Silva, C N S; Bell, J J; Ritchie, P A; Gardner, J P A (2016) Genetic Connectivity Amongst New Zealand's Open Sandy Shore and Estuarine Coastal Taxa. *New Zealand Aquatic Environment and Biodiversity Report No. 172*. 93 p.

Harper, M A; Patterson, J E; Harper, J F (2009) New diatom taxa from the world's first marine Bioblitz held in New Zealand: *Skeletomastus* a new genus, *Skeletomastus coelatus* nov. comb. and *Pleurosigma inscriptura* a new species. *Acta Botanica Croatica 82*: 339–349.

Harvey, A S; Woelkerling, W J; Farr, T J; Neill, K F; Nelson, W A (2005) Coralline algae of central New Zealand: an identification guide to common 'crustose' species. *NIWA Information Series No. 57*.

Heesch, S; Broom, J E S; Neill, K; Farr, T J; Dalen, J L; Nelson, W A (2009) Ulva, Umbraulva and Gemina: genetic survey of New Zealand taxa reveals diversity and introduced species. *European Journal of Phycology 44 (2)*: 143–154. doi:10.1080/09670260802422477.

Hewitt, J E; Bell, R; Costello, M; Cummings, V, et al. (2014) Development of a National Marine Environment Monitoring Programme (MEMP) for New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 141*. 126 p.

Hewitt, J; Julian, K; Bone, E K (2011a) Chatham–Challenger Ocean Survey 20/20 Post-Voyage Analyses: Objective 10 – Biotic habitats and their sensitivity to physical disturbance. *New Zealand Aquatic Environment and Biodiversity Report 81*. 36 p.

Hewitt, J; Lundquist, C J; Bowden, D A (2011b) Chatham-Challenger Ocean Survey 20/20 Post-Voyage Analyses: Diversity Metrics. *New Zealand Aquatic Environment and Biodiversity Report No. 83*. 62 p.

Hewitt, J E; Thrush, S F; Cummings, V J; Turner, S J (1998) The effect of changing sampling scales on our ability to detect effects of large-scale processes on communities. *Journal of Experimental Marine Biology and Ecology 227*: 251–264.

Hewitt, J E; Thrush, S F; Dayton, P K; Bonsdorff, E (2007) The effect of spatial and temporal heterogeneity on the design and analysis of empirical studies of scale-dependent systems. *The American Naturalist 168 (3)*: 398–408.

IPCC (2018) Summary for Policymakers. In: Global warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty [V. Masson-Delmotte, P. Zhai, H. O. Pörtner, D. Roberts, J. Skea, P. R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J. B. R. Matthews, Y. Chen, X. Zhou, M. I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, T. Waterfield (eds.)]. World Meteorological Organization, Geneva, Switzerland, 32 p.

Kingsford, R T; Watson, J E M; Lundquist, C J; Venter, O; Hughes, L; Johnston, E L; Atherton, J; Gawel, M; Keith, D A; Mackey, B G; Morley, C; Possingham, H P; Raynor, B; Recher, H F; Wilson, K A (2009). Major conservation policy issues for biodiversity in Oceania. *Conservation Biology 23 (4)*: 834–840.

Koubbi, P; Masato, M; Duhamel, G; Goarant, A; Hulley, P-A; O'Driscoll, R; Takashi, I; Pruvost, P; Tavenier, E; Hosie, G (2011) Ecological importance of micronektonic fish for the ecoregionalisation of the Indo-Pacific sector of the Southern Ocean: role of myctophids. *Deepsea Research II 58*: 170–180.

Law, C S; Rickard, G J; Mikaloff-Fletcher, S E; Pinkerton, M H; Gorman, R; Behrens, E; Chiswell, S M; Bostock, H C; Anderson, O; Currie, K (2016) The New Zealand EEZ and South West Pacific. Synthesis Report RA2, Marine Case Study. Climate Changes, Impacts and Implications (CCII) for New Zealand to 2100. MBIE contract C01X1225. 41 p. https://ccii.org.nz/app/uploads/2017/04/RA2-MarineCaseStudySynthesisReport.pdf

Le Heron, R; Lewis, N; Fisher, K; Thrush, S; Lundquist, C; Hewitt, J; Ellis, J (2016) Non-sectarian scenario experiments in socio-ecological knowledge building for multi-use marine environments: Insights from New Zealand's Marine Futures project. *Marine Policy* 67: 10–21. doi:10.1016/j.marpol.2016.01.022

Le Quesne, W J F; Pinnegar, J K (2012) The potential impacts of ocean acidification: scaling from physiology to fisheries. *Fish and Fisheries 13*: 333–344. doi:10.1111/j.1467-2979.2011.00423.x.

Leathwick, J R; Dey, K L; Julian, K (2006a) Development of a marine environmental classification optimised for demersal fish. *NIWA Client Report HAM2006–063*.

Leathwick, J R; Elith, J; Francis, M P; Hastie, T; Taylor, P (2006b) Variation in demersal fish species richness in the oceans surrounding New Zealand: an analysis using boosted regression trees. *Marine Ecology Progress Series 321*: 267–281.

Leathwick, J; Francis, M; Julian, K (2006c) Development of a demersal fish community map for New Zealand's Exclusive Economic Zone. *NIWA Client Report HAM2006-062*, prepared for Department of Conservation. National Institute of Water & Atmospheric Research, Hamilton, New Zealand.

Leathwick, J R; Rowden, A; Nodder, S; Gorman, R; Bardsley, S; Pinkerton, M; Baird, S J; Hadfield, M; Currie, K; Goh, A (2010) Development of a benthic-optimised marine environment classification for waters within the New Zealand EEZ. Final Research Report for Ministry of Fisheries Research Project BEN200601, Objective 5. (Unpublished report held by Fisheries New Zealand, Wellington.)

Link, J (1999) Reconstructing Food Webs and Managing Fisheries. Ecosystem Approaches for Fisheries Management. Proceedings of the 16th Lowell Wakefield Fisheries Symposium. AK-SG-99-01:571-588.

Lörz, A N; Berkenbusch, K; Nodder, S; Ahyong, S; Bowden, D; McMillan, P; Gordon, D; Mills, S; Mackay, K (2012) A review of deepsea benthic biodiversity associated with trench, canyon and abyssal habitats below 1500 m depth in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 92*. 133 p.

Lörz, A-N; Maas, E; Linse, K; Coleman, C O (2009) Do circum-Antarctic species exist in peracarid Amphipoda? A case study in the genus *Epimeria* Costa, 1851 (Crustacea: Peracarida: Epimeriidae). *Zookeys (18)*: 91–128.

Lundquist, C J; Bulmer, R H; Clark, M R; Hillman, J; Nelson, W A; Norrie, C R; Rowden, A A; Tracey, D M; Hewitt, J E (2017) Challenges for the conservation of marine small natural features. *Biological Conservation 211*: 69–79. http://dx.doi.org/10.1016/j.biocon.2016.12.027

Lundquist, C; Davies, K; McCartain, L (2015) Best practice guidelines for marine protected area network design and evaluation. *NIWA Client Report HAM2015-051*, prepared for Ministry for the Environment project MFE15206. 87 p.

Lundquist, C J; Fisher, K; Le Heron, R; Lewis, N; Ellis, J; Hewitt, J; Greenaway, A J; Cartner, K J; Burgess-Jones, T C; Schiel, D R; Thrush, S F (2016) Science and societal partnerships to address cumulative impacts. *Frontiers in Marine Science* 3 (2): 1-12.

Lundquist, C J; Julian, K; Costello, M; Gordon, D; Mackay, K; Mills, S; Neill, K; Nelson, W; Thompson, D (2014) Development of a Tier 1 National Reporting Statistic for New Zealand's Marine Biodiversity. *New Zealand Aquatic Environment and Biodiversity Report No. 147.* 61 p.

Lundquist, C J; Pritchard, M; Thrush, S F; Hewitt, J E; Greenfield, B L; Halliday, N J; Lohrer, A M (2013) Bottom disturbance and seafloor landscapes: Development of a model of disturbance and recovery dynamics for marine benthic ecosystems. *New Zealand Aquatic Environment and Biodiversity Report No. 118*. 58 p.

Lundquist, C J; Ramsey, D; Bell, R; Swales, A; Kerr, S (2011) Predicted impacts of climate change on New Zealand's biodiversity. *Pacific Conservation Biology 17*: 179–191.

MacDiarmid, A B; Abraham, E; Baker, C S; Carroll, E, et al. (2016a) Taking Stock – the changes to New Zealand marine ecosystems since first human settlement: synthesis of major findings, and policy and management implications. *New Zealand Aquatic Environment and Biodiversity Report No. 170*. 48 p.

MacDiarmid, A; Beaumont, J; Bostock, H; Bowden, D; Clark, M; Hadfield, M; Heath, P; Lamarche, G; Nodder, S; Orpin, A; Stevens, C; Thompson, D; Torres, L; Wysoczanski, R (2011) Expert risk assessment of activities in the New Zealand Exclusive Economic Zone and Extended Continental Shelf. NIWA Client Report WLG2011-39, prepared for Ministry for the Environment. 145 p.

MacDiarmid, A B; Boschen, R; Bowden, D; Clark, M; Hadfield, M; Lamarche, G; Nodder, S; Pinkerton, M; Thompson, D (2014) Environmental risk assessment of discharges of sediment during prospecting and exploration for seabed minerals. NIWA Client Report WLG2013-66, prepared for Ministry for the Environment. 53 p.

MacDiarmid, A; Cleaver, P; Stirling, B (2015) Historical evidence for the state and exploitation of marine fish and invertebrate resources in the Hauraki Gulf and along the Otago-Catlins shelf 1769 – 1950. *New Zealand Aquatic Environment and Biodiversity Report.* 615 p.

MacDiarmid, A; McKenzie, A; Abraham, E (2016b) Top-down effects on rocky reef ecosystems in north-eastern New Zealand: a historic and qualitative modelling approach. *New Zealand Aquatic Environment and Biodiversity Report No. 171*. 20 p.

MacDiarmid, A; McKenzie, A; Sturman, J; Beaumont, J; Mikaloff-Fletcher, S; Dunne, J (2012) Assessment of anthropogenic threats to New Zealand marine habitats. *New Zealand Aquatic Environment and Biodiversity Report No. 93*. 255 p.

MacDiarmid, A; Stewart, R (2015) Ross Sea and Balleny Islands biodiversity: routine observations and opportunistic sampling of biota made during a geophysical survey to the Ross Sea in 2006. *New Zealand Aquatic Environment and Biodiversity Report No. 153. 97 p.*

McMillan, P J; Francis, M P; James, G D; Paul, L J; Marriott, P J; Mackay, E; Wood, B A; Griggs, L H; Sui, H; Wei, F (2011a) New Zealand

fishes. Volume 1: A field guide to common species caught by bottom and midwater fishing. *New Zealand Aquatic Environment and Biodiversity Report No. 68*. 329 p.

McMillan, P J; Francis, M P; Paul, L J; Marriott, P J; Mackay, E; Baird, S J; Griggs, L H; Sui, H; Wei, F (2011b) New Zealand fishes. Volume 2: A field guide to less common species caught by bottom and midwater fishing. *New Zealand Aquatic Environment and Biodiversity Report No. 78*. 181 p.

McMillan, P J; Griggs, L H; Francis, M P; Marriott, P J; Paul, L J; Mackay, E; Wood, B A; Sui, H; Wei, F (2011c) New Zealand fishes. Volume 3: A field guide to common species caught by surface fishing. *New Zealand Aquatic Environment and Biodiversity Report No. 69*. 145 p.

McWethy, D B; Whitlock, C; Wilmshurst, J M; McGlone, M S; Fromont, M; Li, X; Dieffenbacher-Krall, A; Hobbs, W O; Fritz, S C; Cook, E R (2010) Rapid landscape transformation in South Island, New Zealand, following initial Polynesian settlement. *Proceedings of the national Academy of Sciences 107*: 21343–21348.

Maxwell, K; MacDiarmid, A B (2016) Taking Stock: Oral histories about changes in marine fauna observed locally in the Hauraki Gulf and along the Otago-Catlins coast, New Zealand, 1926–2008. *New Zealand Aquatic Environment and Biodiversity Report No. 173*. 45 p.

Ministry for the Environment (2007) Environment New Zealand 2007. Wellington, 460 p. Retrieved from http://www.mfe.govt.nz/sites/default/files/environment-nz07-dec07.pdf

Ministry for the Environment (2011) Proposed National Policy Statement on Indigenous Biological Diversity. Retrieved from http://www.mfe.govt.nz/publications/land-biodiversity/proposed-national-policy-statement-indigenous-biodiversity.

Ministry for the Environment; Statistics New Zealand (2015) New Zealand's Environmental Reporting Series: Environment Aotearoa 2015. Available from www.mfe.govt.nz and www.stats.govt.nz

Ministry for the Environment (2016). Our marine environment 2016: Data to 2015. Retrieved from http://www.mfe.govt.nz/sites/default/files/media/Environmental%20reporting/our-marine-environment.pdf

Ministry for the Environment & Department of Conservation (2017). Conservation and Environment Science Roadmap. Ministry for the Environment & Department of Conservation, Wellington, 78 p.

Ministry for Primary Industries (2013). National plan of action – 2013 to reduce the incidental catch of seabirds in New Zealand fisheries. Ministry for Primary Industries, Wellington, New Zealand, 63 p.

Ministry for Primary Industries & Department of Conservation (2013). National plan of action for the conservation and management of sharks. Ministry for Primary Industries, Wellington, New Zealand, 32 p.

Ministry of Business, Innovation and Employment (2017) The Business Growth Agenda 2017. Wellington, 56 p.

Ministry of Fisheries (2009). Fisheries 2030. New Zealanders maximizing benefits from the use of fisheries within environmental limits. Ministry of Fisheries, Wellington. 15 p.

Mitchell, J (2008) Initial environmental evaluation (IEE) NZ IPY-CAML voyage 2008 report. Report prepared for Antarctic Policy Unit, Ministry of Foreign Affairs and Trade. 35 p.

Morrison, M; Lowe, M; Parsons, D; Usmar, N; McLeod, I (2009) A review of land-based effects on coastal fisheries and supporting biodiversity in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 37*. 100 p.

Neill, K; D'Archino, R; Farr, T; Nelson, W (2012) Macroalgal diversity associated with soft sediment habitats in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 87*. 127 p.

Neill, K F; Nelson, W A; D'Archino, R; Leduc, D; Farr, T J (2014) Northern New Zealand rhodoliths: assessing faunal and floral diversity in physically contrasting beds. *Marine Biodiversity 45 (1)*: 63–75. doi:10.1007/s12526-014-0229-0.

Nelson, W A (2009) Calcified macroalgae – critical to coastal ecosystems and vulnerable to change: A review. *Marine and Freshwater Research 60*: 787–801.

Nelson, W A (2013) New Zealand Seaweeds – An illustrated guide. Te Papa Press, Wellington. 328 p.

Nelson, W; Cummings, V; D'Archino, R; Halliday, J; Marriott, P; Neill, K (2010) Macroalgae and benthic biodiversity of the Balleny Islands, Southern Ocean. *New Zealand Aquatic Environment and Biodiversity Report No. 55*. 99 p.

Nelson, W A; D'Archino, R; Neill, K F; Farr, T J (2014) Macroalgal diversity associated with rhodolith beds in northern New Zealand. *Cryptogamie, Algologie 35*: 27–47.

Nelson, W A; Neill, K; Farr, T; Barr, N; D'Archino, R; Miller, S; Stewart, R (2012) Rhodolith beds in northern New Zealand:characterisation of associated biodiversity and vulnerability to environmental stressors. *New Zealand Aquatic Environment and Biodiversity Report 99*. 102 p.

Nodder, S D (2008) OS 20/20 Chatham Rise & Challenger Plateau Hydrographic, Biodiversity & Seabed Habitats. *NIWA Client Report WLG2008-27*, NIWA, Wellington, New Zealand.

Nodder, S; Maas, E; Bowden, D; Pilditch, C (2011) Physical, Biogeochemical and Microbial Characteristics of Sediment Samples from the Chatham Rise and Challenger Plateau. *New Zealand Aquatic Environment and Biodiversity Report No. 70*. 34 p.

Norkko, A; Andrew, N; Thrush, S; Cummings, V; Schwarz, A-M; Hawes, I; Mercer, S; Budd, R; Gibbs, M; Funnell, G; Hewitt, J; Goring, D

(2002) Ecology and biodiversity of coastal benthic communities in McMurdo Sound, Ross Sea: development of sampling protocols and initial results. Final Research Report for Ministry of Fisheries Research Project ZBD2001/02, Objectives 1, 2 & 3. 119 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Norkko, J; Norkko, A; Thrush, S F; Cummings, V J (2005) Growth under environmental extremes: spatial and temporal patterns in nucleic acid ratios in two Antarctic bivalves. *Journal of Experimental Marine Biology and Ecology 326*: 114–156.

Norkko, A; Thrush, S F; Cummings, V J; Funnell, G A; Schwarz, A-M S; Andrew, N L; Hawes, I (2004) Ecological role of *Phyllophora antarctica* drift accumulations in coastal soft-sediment communities of McMurdo Sound, Antarctica. *Polar Biology 27*: 482–494.

Norkko, A; Thrush, S F; Cummings, V J; Gibbs, M M; Andrew, N L; Norkko, J; Schwarz, A-M (2007) Trophic structure of coastal Antarctic food webs associated with changes in sea ice and food supply. *Ecology 88*: 2810–2820.

O'Driscoll, R L (2009) Preliminary acoustic results from the New Zealand IPY-CAML survey of the Ross Sea region in February–March 2008. Final Research Report for Ministry of Fisheries project IPY200701, Objective 8. 14 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

O'Driscoll, R L; Hanchet, S M; Miller, B S (2012) Can acoustic methods be used to monitor grenadier (Macrouridae) abundance in the Ross Sea region? *Journal of Ichthyology 53*: 700–708.

O'Driscoll, R L; Macaulay, G J; Gauthier, S; Pinkerston, M; Hanchet, S (2011) Distribution, abundance, and acoustic properties of Antarctic silverfish (*Pleuragramma antarcticum*) in the Ross Sea. *Deepsea Research II 58*: 181–195.

O'Loughlin, M P; Paulay, G; Davey, N; Michonneau, F (2011) The Antarctic region as a marine biodiversity hotspot for echinoderms: Diversity and diversification of sea cucumbers. *Deepsea Research II 58*: 264–275.

Office of the Prime Minister's Science Advisory Committee (2013) New Zealand's changing climate and oceans: The impact of human activity and implications for the future. Retrieved from http://www.pmcsa.org.nz/wp-content/uploads/New-Zealands-Changing-Climate-and-Oceans-report.pdf

Opresko, D; Tracey, D; Mackay, E (2014) Antipatharia (black corals) for the New Zealand region. A field guide of commonly sampled New Zealand black corals including illustrations highlighting technical terms and black coral morphology. *New Zealand Aquatic Environment and Biodiversity Report No. 136*. 20 p.

Page, M J; Alcock, N; Gordon, D; Kelly-Shanks, M; Nelson, W; Neill, K; Watson, J (2001) Preliminary assessment of the biodiversity of benthic macrofauna of the western Ross Sea, Antarctica. Final Research Report to the Ministry of Fisheries. 29 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Phillips, B; Perez-Ramirez, M (Eds.) (2017) Climate Change Impacts on Fisheries and Aquaculture: A Global Analysis. Volume 1. Wiley-Blackwell.

Pinkerton, M; Hanchet, S; Bradford-Grieve, J; Cummings, V; Wilson, P; Williams, M (2006) Modelling the effects of fishing in the Ross Sea. Final Research Report for Ministry of Fisheries Research Project ANT2004-05. 169 p. (Unpublished report held by Fisheries New Zealand, Wellington.)

Pinkerton, M H; MacDiarmid, A; Beaumont, J; Bradford-Grieve, J, et al. (2015) Changes to the food-web of the Hauraki Gulf during the period of human occupation: a mass-balance model approach. *New Zealand Aquatic Environment and Biodiversity Report No. 160*. 346 p.

Poloczanska, E S; Brown, C J; Sydeman, W J; Kiessling, W, et al. (2013) Global imprint of climate change on marine life. *Nature Climate Change 3 (10)*: 919–925.

Ramirez-Llodra, E; Tyler, P A; Baker, M C; Bergstad, O A; Clark, M R; Escobar, E; Levin, L A; Menot, L; Rowden, A A; Smith, C R; Van Dover, C L (2011) Man and the Last Great Wilderness: Human Impact on the Deepsea. *PLoS ONE 6 (8)*: e22588. doi:10.1371/journal.pone.0022588.

Richard, Y; Abraham, E R; Berkenbusch, K (2017). Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006–07 to 2014–15. *New Zealand Aquatic Environment and Biodiversity Report No. 191*. 104 p.

Robinson, K V; Pinkerton, M H; Hall, J A; Hosie, G H (2014) Continuous Plankton Recorder sampling between New Zealand and the Ross Sea, 2006–2013. *New Zealand Aquatic Environment and Biodiversity Report No. 128*. 70 p.

Rogers, A D; Laffoley, D (2011) International Earth system expert workshop on ocean Stresses and impacts. Summary report. IPSO Oxford. 18 p.

Rowden, A, et al. (2012a) Biodiversity of macroinvertebrates and fish assemblages of the northwestern Ross Sea shelf. *New Zealand Aquatic Environment and Biodiversity Report No. 101*. 111 p.

Rowden, A A; Berkenbusch, K; Brewin, P E; Dalen, J; Neill, K F; Nelson, W A; Oliver, M D; Probert, P K; Schwarz, A-M; Sui, P H; Sutherland, D (2012b) A review of the marine soft-sediment assemblages of New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 96*. 165 p.

Rowden, A A; Clark, M R (2010) Benthic biodiversity of seven seamounts on the southern end of the Kermadec volcanic arc, northeast New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 62*. 31 p.

Rowden, A A; Clark, M R; O'Shea, S (2004) Benthic biodiversity of seamounts on the Northland Plateau. *Marine Biodiversity Biosecurity Report No. 5*. 21 p.

Rowden, A A; Clark, M R; O'Shea, S; McKnight, D G (2003) Benthic biodiversity of seamounts on the southern Kermadec volcanic arc. *Marine Biodiversity Biosecurity Report No. 3*. 23 p.

Rowden, A A; Kroger, K; Clark, M (2013) Biodiversity of macroinvertebrates and fish assemblages of the Balleny Islands seamounts. *New Zealand Aquatic Environment and Biodiversity Report No. 115*. 76 p.

Rowden, A A; Oliver, M; Clark, M R; Mackay, K (2008) New Zealand's 'SEAMOUNT' database: recent updates and its potential use for ecological risk assessment. *New Zealand Aquatic Environment and Biodiversity Report No. 27*. 49 p.

Rowden, A A; O'Shea, S; Clark, M R (2002) Benthic biodiversity of seamounts on the northwest Chatham Rise. *Marine Biodiversity Biosecurity Report 2*: 1–21.

Royal Society of New Zealand (2012) Future marine resource use: emerging issues. Royal Society of New Zealand, retrieved from www.royalsociety.org.nz.

Russell, L K; Hepburn, C D; Hurd, C L; Stuart, M D (2008) The expanding range of *Undaria pinnatifida* in southern New Zealand: distribution, dispersal mechanisms and the invasion of wave-exposed environments. *Biological Invasions 10*: 103–115.

Schwarz, A-M; Hawes, I; Andrew, N; Mercer, S; Cummings, V; Thrush, S (2005) Primary production potential of non-geniculate coralline algae at Cape Evans, Ross Sea, Antarctica. *Marine Ecology Progress Series 294*: 131–140.

Schwarz, A-M; Hawes, I; Andrew, N; Norkko, A; Cummings, V; Thrush, S (2003) Macroalgal photosynthesis near the southern global limit for growth; Cape Evans, Ross Sea, Antarctica. *Polar Biology 26*: 789–799.

Schwarz, A; Taylor, R; Hewitt, J; Phillips, N; Shima, J; Cole, R; Budd, R (2006) Impacts of terrestrial runoff on the biodiversity of rocky reefs. *New Zealand Aquatic Environment and Biodiversity Report No 7*. 109 p.

Sewell, M A (2005) Examination of the meroplankton community in the southwestern Ross Sea, Antarctica, using a collapsible plankton net. *Polar Biology 28*: 119–131.

Sewell, M A (2006) The meroplankton community of the northern Ross Sea: a preliminary comparison with the McMurdo Sound region. *Antarctic Science 18*: 595–602.

Sewell, M A; Lavery, S; Baker, C S (2006) Whose larva is that? Molecular identification of planktonic larvae of the Ross Sea. *New Zealand Aquatic Environment and Biodiversity Report No. 3*. 57 p.

Sharp, B R; Parker, S J; Pinkerton, M H, et al. (2010) Bioregionalisation and Spatial Ecosystem Processes in the Ross Sea Region. CCAMLR document WG-EMM-10/30, Hobart, Australia.

Silva, C N S; Gardner, J P A (2014) Development and characterisation of 12 microsatellite markers for the New Zealand endemic scallop *Pecten novaezelandiae*. *Conservation Genetics Resources 6 (2)*: 327–328.

Smith, A M; Gordon, D (2011) Bryozoans of southern New Zealand Field Identification Guide. *New Zealand Aquatic Environment and Biodiversity Report No. 75*. 64 p.

Smith, I W G (2005) Retreat and resilience: Fur seals and human settlement in New Zealand. *In*: Monks, G G (Ed.) The Exploitation and Cultural Importance of Marine Mammals. pp 6–18. Oxbow Books, Oxford.

Smith, I W G (2011) Estimating the magnitude of pre-European Maori marine harvest in two New Zealand study areas. *New Zealand Aquatic Environment and Biodiversity Report No. 82*. 72 p.

Smith, P J; Steinke, D; McMillan, P J; McVeagh, S M; Struthers, C D (2008) DNA database for commercial marine fish. *New Zealand Aquatic Environment and Biodiversity Report No. 22*. 62 p.

Snelder, T; Leathwick, J; Dey, K; Rowden, A; Weatherhead, M (2006) Development of an ecological marine classification in the New Zealand region. *Environmental Management 39*: 12–29.

Soga, M; Gaston, K J (2018) Shifting baseline syndrome: causes, consequences, and implications. *Frontiers in Ecology and the Environment* 16: 222–230.

Sogin, M L; Morrison, H G; Huber, J A; Welch, D M; Huse, S M; Neal, P R; Arrieta, J M; Herndl G J (2006) Microbial diversity in the deepsea and the underexplored 'rare biosphere'. *Proceedings Natural Academy Science 103 (32)*: 12115–12120.

Statistics New Zealand, Ministry for the Environment, Department of Conservation (2013) Environment domain plan 2013: Initiatives to address our environmental information needs. Retrieved from www.stats.govt.nz.

Sutherland, D L (2008) Surface-associated diatoms from marine habitats at Cape Evans, Antarctica, including the first record of living *Eunotogramma marginopunctatum*. *Polar Biology 31*: 879–888. doi:10.1007/s00300-008-0426-z.

Thrush, S F; Cummings, V J (2011) Massive icebergs, alteration in primary food resources and change in benthic communities at Cape Evans, Antarctica. *Marine Ecology 32 (3)*: 289–299. doi:10.1111/j.1439-0485.2011.00462.x.

Thrush, S F; Dayton, P K (2002) Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. *Annual Review of Ecology and Systematics 33*: 449–473.

Thrush, S F; Dayton, P K; Cattaneo-Vietti, R; Chiantore, M; Cummings, V J; Andrew, N L; Hawes, I; Kim, S; Kvitek, R; Schwarz, A -M (2006) Broad-scale factors influencing the biodiversity of coastal benthic communities of the Ross Sea. *Deepsea Research II 53*: 959–971.

Thrush, S F; Hewitt, J E; Cummings, V J; Green, M O; Funnell, G A; Wilkinson, M R (2000) The generality of field experiments: interactions between local and broad-scale processes. *Ecology 81*: 399–415.

Thrush, S F; Hewitt, J; Cummings, V J; Norkko, A; Chiantore, M (2010) β-Diversity and species accumulation in Antarctic coastal

benthos: Influence of habitat, distance and productivity on ecological connectivity. *PLoS ONE 5*: e11899.

Thrush, S F; Pridmore, R D; Bell, R G; Cummings, V J, et al. (1997) The sandflat habitat: Scaling from experiments to conclusions. *Journal of Experimental Marine Biology and Ecology 216*: 1–9.

Tracey, D M; Anderson, O F; Clark, M R; Oliver, M D (2005) A guide to common deepsea invertebrates in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 1*. 160 p.

Tracey, D M; Anderson, O F; Naylor, J R (2007) A guide to common deepsea invertebrates in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 10*. 282 p.

Tracey, D M; Anderson, O F; Naylor, R J (2011a) A guide to common deepsea invertebrates in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 86*. 317 p.

Tracey, D; Baird, S J; Sanders, B M; Smith, M H (2011b) Distribution of protected corals in relation to fishing effort and assessment of accuracy of observer identification. *NIWA Client Report WLG2011-33*, prepared for Department of Conservation, Wellington. 74 p.

Tracey, D; Bostock, H; Currie, K; Mikaloff-Fletcher, S; Williams, M; Hadfield, M; Neil, H; Guy, C; Cummings, V (2013) The potential impact of ocean acidification on deepsea corals and fisheries habitat in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 117*. 101 p.

Williams, G C; Tracey, D; Mackay, E (2014) Pennatulacea (sea pens) descriptions for the New Zealand region. A field guide of commonly sampled New Zealand sea pens including illustrations highlighting technical terms and sea pen morphology. *New Zealand Aquatic Environment and Biodiversity Report No. 135*. 22 p.

Williams, R; Gould, B; Christian, S (2008) Shipwrecks — an international biosecurity risk? *Surveillance 35*: 4–6.

Wing, S R (2005) Fiordland Biodiversity Research Cruise Final Research Report ZBD2003-04. (Unpublished report held by Fisheries New Zealand, Wellington.)

.

19 APPENDICES

Status of chapter	This chapter has been fully updated for AEBAR 2018.

19.1 TERMS OF REFERENCE FOR THE AQUATIC ENVIRONMENT WORKING GROUP (AEWG) FOR 2017 ONWARDS

OVERALL PURPOSE

The purpose of the AEWG is to assess, based on scientific information, the effects of (and risks posed by) fishing (for all fisheries in which New Zealand engages), aquaculture, and enhancement on the aquatic environment including:

- bycatch and unobserved mortality of protected species (e.g., seabirds and marine mammals), fish, and other marine life, and consequent impacts on populations;
- effects on benthic ecosystems, species, and habitat;
- effects on biodiversity, including genetic diversity;
- changes to ecosystem structure and function from fishing, including trophic effects; and
- effects of aquaculture and fishery enhancement on the environment and on fishing.

Where appropriate and feasible, such assessments should explore the implications of the effect, including with respect to government standards, other agreed reference points, or other relevant indicators of population or environmental status. Where possible, projections of future status under alternative management scenarios should be made.

AEWG does not make management recommendations or decisions (this responsibility lies with MPI fisheries managers and the Minister responsible for fisheries).

MPI also convenes a Biodiversity Research Advisory Group (BRAG) which has a similar review function to the AEWG. Projects reviewed by BRAG and AEWG have some commonalities however, the key focus of projects considered by BRAG is on the functionality of the marine ecosystem and its productivity, whereas projects considered by AEWG more commonly focus on the direct effects of fishing, aquaculture or enhancement.

PREPARATORY TASKS

1. Prior to the beginning of AEWG meetings each year, MPI fisheries scientists will produce a list of issues for which new assessments or evaluations are likely to become available that year.
2. The Ministry's research planning processes should identify most information needs well in advance but, if urgent issues arise, MPI-Fisheries or aquaculture staff will alert the relevant AEWG chair prior to the required meetings of items that could be added to the agenda. AEWG Chairs will determine the final timetables and agendas for meetings.

TECHNICAL OBJECTIVES

3. To review any new research information on fisheries, aquaculture or enhancement impacts, including risks of impacts, and the relative or absolute sensitivity or susceptibility of potentially affected species, populations, habitats, and systems.
4. To estimate and derive appropriate reference points for determining population, system, or environmental status, noting any relevant draft or published management policies (e.g., National Plan of Action or Threat Management Plan). To conduct environmental assessments or evaluations for selected species, populations, habitats, or systems in order to determine their status relative to appropriate reference points and Standards, where such exist.
5. In addition to determining the status of the species, populations, habitats, and systems relative to reference points, and particularly where the status is unknown, AEWG should explore the potential for using existing data and analyses to draw conclusions about likely future

trends in fishing effects or status if current fishing methods, effort, catches, and catch limits are maintained, or if fishers or fisheries managers are considering modifying them in other ways.

6. Where appropriate and practical, to conduct or request projections of likely future status using alternative management actions, based on input from AEWG, fisheries plan advisers and fisheries and standards managers, noting any draft or published Standards.
7. For species or populations deemed to be depleted or endangered, to develop ideas for alternative rebuilding scenarios to levels that are likely to ensure long-term viability based on input from AEWG, fisheries managers, noting any draft or published management policies (e.g., National Plan of Action or Threat Management Plan).
8. For species, populations, habitats, or systems for which new assessments are not conducted in the current year, to review and update any existing Fisheries Assessment Plenary report text in order to determine whether the latest reported status summary is still relevant; else to revise the evaluations based on new data or analyses, or other relevant information.
9. To review and revise existing environmental and ecosystem consideration sections of Fisheries Assessment Plenary report text based on new data or analyses, or other relevant information.

WORKING GROUP INPUT TO ANNUAL AQUATIC ENVIRONMENT AND BIODIVERSITY ANNUAL REVIEW

10. To include in contributions to the Aquatic Environment and Biodiversity Annual Review (AEBAR) summaries of information on selected issues that may relate to species, populations, habitats, or systems that may be affected by fishing, aquaculture or enhancement. These contributions are analogous to Working Group reports from the Fisheries Assessment Working Groups.
11. To provide information and scientific advice on management considerations (e.g., area boundaries, bycatch issues, effects of fishing on habitat, other sources of mortality, and input controls such as mesh sizes and minimum legal sizes) that may be relevant for setting sustainability measures.
12. To summarise the assessment methods and results, along with estimates of relevant standards, references points, or other metrics that may be used as benchmarks or to identify risks to the aquatic environment.
13. It is desirable that full agreement among technical experts is achieved on the text of contributions to the AEBAR. If full agreement among technical experts cannot be reached, the Chair will determine how this will be depicted in the AEBAR, will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes.
14. To advise the Principal Advisor Fisheries Science and Aquatic Environment Team Manager, about issues of particular importance that may require independent review or updating in the AEBAR. The general criterion for determining which issues should be discussed by a wider group or text changed in the AEBAR is that new data or analyses have become available that alter the previous assessment of an issue, particularly assessments of population status or projection results. Such information could include:

    - New or revised estimates of environmental reference points, recent or current population status, trend, or projections;
    - The development of a major trend in bycatch rates or amount;
    - Any new studies or data that extend understanding of population, system, or environmental susceptibility to an effect or its recoverability, fishing patterns, or mitigation measures that have a substantial implications for a population, system, or environment or identify risks associated with fishing activity, aquaculture or enhancement; and
    - Consistent performance outside accepted reference points or goals as defined by relevant draft or published management policies (e.g., National Plan of Action or Threat Management Plan).

AEWG MEMBERSHIP 2017

CONVENORS: Rich Ford, Nathan Walker

Membership lists those that have attended at least one meeting in the 2017 calendar year.

MEMBERS: Ed Abraham, Owen Anderson, Karen Baird, Suze Baird, Stephen Beatson, Tiffany Bock, Laura Boren, Christine Bowden, Erin Breen, Susan Chalmers, Tom Clark, Katie Clemens-Seely, Deanna Clement, Richard Cuthbert, Igor Debski, Charlie Edwards, Jack Fenaughty, Dave Foster, Malcolm Francis, Dave Goad, Katrina Goddard, Judi Hewitt, Kristina Hillock, Freydis Hjorvarsdottis, Lyndsey Holland, Sunkita Howard, Krista Hupman, Rosie Hurst, James Jolly, Emma Jones, Jo Lambie, Amanda Leathers, Mary Livingston, Greg Lydon, Lucy Manning, Jennifer Matthews, Andy Mckay, Darryl McKenzie, David Middleton, Rikki Mules, Conor Nielson, Jenny Oliver, Tracey Osborne, Enrique Pardo, Heiko Phillipi, Kris Ramm, Amanda Richards, Yvan Richard, Jim Roberts, Bruce Robertson, Marie-Julie Roux, Paul Scofield, Ben Sharp, Liz Slooten, Geoff Tingley, Rob Tinkler, Ian Tuck, Anton Van Heldon, D'Arcy Webber, Barry Weeber, Jody Weir, Richard Wells, Sam Whinam.

19.2 TERMS OF REFERENCE FOR THE BIODIVERSITY RESEARCH ADVISORY GROUP (BRAG) FOR 2017 ONWARDS

OVERALL PURPOSE

Since 2000, the objectives of the Biodiversity Research Programme have been drawn directly from MFish commitments to Theme 3 of the New Zealand Biodiversity Strategy. Within this framework, the Biodiversity Medium Term Research Plan has been adapted over time as new issues emerge, to build on synergies with other research programmes and work where biodiversity is under greatest threat from fishing or other anthropogenic activity.

Within the constraints of the overall purpose of the Programme,

> 'To improve our understanding of New Zealand marine ecosystems in terms of species diversity, marine habitat diversity, and the processes that lead to healthy ecosystem functioning, and the role that biodiversity has for such key processes'[1]

and the NZBS definition of biodiversity (the variability among living organisms from all sources including inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystem) the science currently commissioned broadly aims to:

- describe and characterise the distribution and abundance of fauna and flora, as expressed through measures of biodiversity, and improving understanding about the drivers of the spatial and temporal patterns observed;
- determine the functional role of different organisms or groups of organisms in marine ecosystems, and assess the role of marine biodiversity in mitigating the impacts of anthropogenic disturbance on healthy ecosystem functioning;
- identify which components of biodiversity must be protected to ensure the sustainability of a healthy marine ecosystem as well as to meet societal values on biodiversity.

MPI also convenes the Aquatic Environment Working Group (AEWG; see above), which has a similar review function to the BRAG. Projects reviewed by BRAG and AEWG have some commonalities in that they relate to aspects of the marine environment. However, the key focus of projects considered by BRAG is on marine issues related to the functionality of the marine ecosystem and its productivity, whereas projects considered by AEWG are more commonly focused on the direct effects of fishing.

BRAG may identify natural resource management issues that extend beyond fisheries management and make recommendations on priority areas of research that will inform other MPI sectors or other government departments of emerging science results that require the attention of managers, policymakers and decision-makers in the marine sector. BRAG does not make management recommendations or decisions (this responsibility lies with the MPI Fisheries Management Group and the Minister responsible for fisheries).

[1] See MFish Biodiversity Research Programme 2010: Part 1. Context and Purpose.

PREPARATORY TASKS

1. Prior to the beginning of BRAG meetings each year, MPI fisheries scientists will produce a list of issues for which new research projects are likely to be required in the forthcoming financial year. The BRAG Chair will determine the final timetables and agendas.
2. The Ministry's research planning processes should identify most information needs well in advance but, if urgent issues arise, MPI fisheries managers will alert the Aquatic Environment and Biodiversity Science Manager and the Principal Advisor Fisheries Science at least three months prior to the required meetings where possible.

BRAG TECHNICAL OBJECTIVES

3. To review, discuss and convey views on the results of marine biodiversity research projects contracted by MPI (formerly Ministry of Fisheries).

It is the responsibility of the BRAG to review, discuss, and convey views on the results of marine biodiversity research projects contracted by MPI and the former Ministry of Fisheries. The review process is an evaluation of how existing research results can be built upon to address emerging research issues and needs. It is essentially an evaluation of 'what we already know' and how this can be used to obtain 'what we need to know'. This information should be used by the BRAG to identify gaps in our knowledge and for developing research plans to address these gaps.

4. Discuss, evaluate, make recommendations and convey views on a 3- to 5-year Medium Term Research Plan.

It is the responsibility of BRAG participants to discuss, evaluate, make recommendations and convey views on a 3- to 5-year Medium Term Research Plan for its particular research area as required. Individual related projects on a species or fishery or research topic need to be integrated into Medium Term Research Plans. The Medium Term Research Plans should encompass research needs and directions for at least the next 3 to 5 years.

The Biodiversity Medium Term Research Plan is aligned to relevant strategic and policy directions such as the 'MPI Statement of Intent' and any Strategic Research Plan (Fisheries 2030, Deepwater 10-year research plan) and fisheries plans developed for the appropriate species/fishery or research area, including biodiversity.

The recommendations on project proposals for the next financial year will be submitted via the Chair of BRAG to the Principal Science Advisor Fisheries (MAF).

5. The Biodiversity Research Programme includes research in New Zealand's TS, EEZ, Extended Continental Shelf, the South Pacific Region and the Ross Sea region and has seven scientific work streams as follows:
   i. To develop ecosystem-scale understanding of biodiversity in the New Zealand marine environment.
   ii. To classify and characterise the biodiversity, including the description and documentation of biota, associated with nearshore and offshore marine habitats in New Zealand.
   iii. To investigate the role of biodiversity in the functional ecology of nearshore and offshore marine communities.
   iv. To assess developments in all aspects of biodiversity, including genetic marine biodiversity and identify key topics for research.
   v. To determine the effects of climate change and increased ocean acidification on marine biodiversity, as well as effects of incursions of non-indigenous species, and other threats and impacts.
   vi. To develop appropriate diversity metrics and other indicators of biodiversity that can be used to monitor change.
   vii. To identify threats and impacts to biodiversity and ecosystem functioning beyond natural environmental variation.

BRAG INPUT TO MPI 'AQUATIC ENVIRONMENT AND BIODIVERSITY ANNUAL REVIEW'

6. To contribute to and summarise progress on biodiversity research in the Aquatic Environment and Biodiversity Annual Review. This contribution is analogous to Working Group Reports from the Fishery Assessment Working Groups.
7. To summarise the assessment methods and results, along with estimates of relevant standards, references points, or other metrics that may be relevant to biodiversity objectives by MPI,

the Biodiversity Strategy and international obligations.

8. It is desirable that full agreement among technical experts is achieved on the text of these contributions. If full agreement among technical experts cannot be reached, the Chair will determine how this will be depicted in the Aquatic Environment and Biodiversity Annual Review, will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes.

9. To advise the Principal Science Advisor Fisheries (MPI) about issues of particular importance that may require review by a plenary meeting or summarising in the Aquatic Environment and Biodiversity Annual Review. The general criterion for determining which issues should be discussed by a wider group include:

    - emerging issues, recent or current biodiversity status assessments, trends, or projections;
    - the development of a major trend in the marine environment that will impact on marine productivity or ecosystem resilience to stressors;
    - any new studies or data that impact on international obligations.

BRAG ATTENDANCE 2016–17

CONVENOR: Mary Livingston

MEMBERS: Sara Bury, David Bowden, Malcolm Clark, Vonda Cummings, Roberta D'Archino, Moira Decima, Britt Graham, Barb Hayden, Cliff Law, Daniel Leduc, Carolyn Lundquist, Alison McDiarmid, Sara Mikaloff-Fletcher, Sadie Mills, Wendy Nelson, Scott Nodder, Darren Parsons, Jim Roberts, Di Tracey, Benton Twist, Ashley Rowden, Matt Pinkerton (NIWA), Rochelle Constantine (University of Auckland), Shane Geange, Debbie Freeman (DOC), Amanda Leathers (WWF), Colin Johnston (AQNZ), Richard Wells (Independent), Sharleen Gargiulo (Deepwater Group), Rich Ford, Malindi Gammon, Lyndsey Holland, Greg Lydon Jennifer Matthews, Tiffany Bock (MPI), Jonathan Gardner (VUW).

19.3 TERMS OF REFERENCE FOR THE ANTARCTIC WORKING GROUP (ANTWG) FOR 2017 ONWARDS

OVERALL PURPOSE

The purpose of the ANTWG is to review science and research information intended for submission to or use by the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR). CCAMLR is an inter-governmental organisation that is committed to conserving the marine life of the Southern Ocean while allowing rational use of marine resources, including commercial fishing. The CCAMLR Convention requires that management considers the effects of fishing on dependent and associated species as well as on the target species. The area of jurisdiction of the CCAMLR Convention is approximately south of the circumpolar Antarctic Polar Front in the Southern Ocean. Science and research requested or used by CCAMLR may include, inter alia, fishery characterisations, abundance indices, catch-at-age or catch-at-length data, and stock assessment modelling to assess the status of fish stocks managed by CCAMLR; bycatch and unobserved mortality of protected species, fish, and other marine life; effects on biodiversity and benthic biodiversity, species, and habitat; and changes to ecosystem structure and function as a result of fishing, including trophic effects. The ANTWG also undertakes scientific review of documents and papers that may be submitted to the scientific working groups of CCAMLR to aid and inform its management. The ANTWG does not make management recommendations or decisions; these responsibilities lie with CCAMLR's Scientific Committee and the Commission.

PREPARATORY TASKS

1. Prior to the first meeting of the ANTWG each year, the ANTWG Chair will produce a list of stocks/issues for which new stock assessments, evaluations, impact assessments, risk assessments, or other scientific analyses have been requested by the CCAMLR Scientific Committee or the Commission (including its contributing bodies), fishing industry, or other stakeholders. The ANTWG Chair will determine the final timetables and agendas of the working group each year, taking account of the available time and resources.

TECHNICAL OBJECTIVES

2. To review new research information on stock structure, productivity, abundance and related topics for each fish stock or environmental issue under the purview of the ANTWG.
3. Where possible, to derive yields or reference points requested by CCAMLR's Scientific Committee or Commission related to fish stocks or environmental issues relevant to CCAMLR fisheries.
4. To conduct stock assessments or evaluations for selected stocks in order to determine the precautionary yields and status of the stocks relative to the requested reference points or, if no such reference points are specified by CCAMLR, MSY-compatible reference points and associated limits, based on the 'Guide to Biological Reference Points for Fisheries Assessment Meetings' and New Zealand's Harvest Strategy Standard.
5. For stocks where the status is unknown, the ANTWG should, where possible, use any existing data and analyses to draw conclusions about likely future trends in biomass levels and/or fishing mortality (or exploitation) rates if current catches and/or TACs are maintained, or if fishers or CCAMLR are considering modifying them in other ways.
6. Where requested by the CCAMLR Scientific Committee or Commission, to conduct projections of likely future stock status using alternative fishing mortality (or exploitation) rates or catches and other relevant management actions, based on input from the ANTWG and any guidance from the CCAMLR Scientific Committee or Commission.
7. Where requested by the CCAMLR Scientific Committee or Commission, in relation to specified stocks, to develop and report on alternative rebuilding scenarios.
8. To conduct environmental impact assessments and qualitative or quantitative risk assessments in relation to bycatch species, other species of concern, benthic systems, or vulnerable marine ecosystems to support the work of the CCAMLR Scientific Committee and Commission.

WORKING GROUP REPORTS

9. To review, and update if necessary, the 'Status of the Stocks' tables in the Fisheries Assessment Plenary report based on new data or analyses, or other relevant information.
10. To complete (and/or update) the Status of Stocks tables using the template provided in the Introductory chapter of the most recent May Plenary report.
11. It is desirable that full agreement amongst technical experts is achieved on the text of the ANTWG reports. If full agreement amongst technical experts cannot be reached, the Chair will determine how this will be depicted in the ANTWG report, will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes.

PAPERS AND REPORTS TO CCAMLR

12. Papers and reports summarising work reviewed by the ANTWG are generally submitted to CCAMLR's Scientific Committee, and their content varies widely. It is desirable that full agreement amongst technical experts is achieved on the content of such papers or reports, noting that deadlines for submission to CCAMLR may require the Chair to finalise text after a meeting of the ANTWG has considered and resolved scientific issues. If full agreement amongst technical experts cannot be reached, the Chair will determine how this will be depicted in the paper or report to be submitted to CCAMLR. In such cases, the Chair will also document the extent to which agreement or consensus was achieved and record and attribute any residual disagreement in the meeting notes.

ANTWG ATTENDANCE 2017–18

CONVENOR: Marine Pomarède

MEMBERS: David Bowden, Brit Finucci, Malcolm Francis, Stuart Hanchet, Sophie Mormede, Richard O'Driscoll, Steve Parker, Matt Pinkerton, Darren Stevens (all NIWA), Debbie Freeman, Greig Funnell (DOC), Barry Weeber (ECO), Brodie Plum, Andy Smith (Talleys), Darryn Shaw (Sanford), Regina Eisert (University of Canterbury), Jack Fenaughty (Silvifish Resources Ltd), Matt Baird, Martin Cryer, Alistair Dunn, Lesley Gould, Gretchen Skea, Karen Tunley, Kalolaine Vaipuna (all MPI), Bob Zuur.

19.4 GENERIC TERMS OF REFERENCE FOR FISHERIES ASSESSMENT WORKING GROUPS (FAWGS) FOR 2017 ONWARDS

OVERALL PURPOSE

The purpose of the FAWGs is to assess the status of fish stocks managed within the Quota Management System, as well as other important species of interest to New Zealand. Based on scientific information the FAWGs assess the current status of fish stocks or species relative to MSY-compatible reference points and other relevant indicators of stock status, conduct projections of stock size and status under alternative management scenarios, and review results from relevant research projects. They do not make management recommendations or decisions (this responsibility lies with MPI fisheries managers and the Minister responsible for fisheries).

PREPARATORY TASKS

1. Prior to the beginning of the main sessions of FAWG meetings (January to May and September to November), MPI fisheries scientists will produce a list of stocks and issues for which new stock assessments or evaluations are likely to become available prior to the next scheduled sustainability rounds. This list will include stocks for which the fishing industry and others intend to directly purchase scientific analyses. It is therefore incumbent on those purchasing research to inform the relevant FAWG chair of their intentions at least three months prior to the start of the sustainability round. FAWG Chairs will determine the final timetables and agendas for each Working Group.
2. At least six months prior to the main sessions of FAWG meetings, MPI fisheries managers will alert MPI science managers and the Principal Advisor Fisheries Science to unscheduled special cases for which assessments or evaluations are urgently needed.

TECHNICAL OBJECTIVES

3. To review any new research information on stock structure, productivity, abundance and related topics for each fish stock/issue under the purview of individual FAWGs.
4. Where possible, to estimate appropriate MSY-compatible reference points [2] for selected fish stocks for use as reference points for determining stock status, based on the Harvest Strategy Standard for New Zealand Fisheries[3] (the Harvest Strategy Standard).
5. To conduct stock assessments or evaluations for selected fish stocks in order to determine the status of the stocks relative to MSY-compatible reference points[1] and associated limits, based on the 'Guide to Biological Reference Points for Fisheries Assessment Meetings', the Harvest Strategy Standard, and relevant management reference points and performance measures set by fisheries managers.
6. For stocks where the status is unknown, FAWGs should use existing data and analyses to draw logical conclusions about likely future trends in biomass levels and/or fishing mortality (or exploitation) rates if current catches and/or TACs/TACCs are maintained, or if fishers or fisheries managers are considering modifying them in other ways.
7. Where appropriate and practical, to conduct projections of likely future stock status using alternative fishing mortality (or exploitation) rates or catches and other relevant management actions, based on the Harvest Strategy Standard and input from the FAWG and fisheries managers.
8. For stocks that are deemed to be depleted or collapsed, to develop alternative rebuilding scenarios based on the Harvest Strategy Standard and input from the FAWG and fisheries managers.
9. For fish stocks for which new stock assessments are not conducted in the current year, to review the existing Fisheries Assessment Plenary report

[2] MSY-compatible reference points include those related to stock biomass (i.e., B_{MSY}), fishing mortality (i.e., F_{MSY}) and catch (i.e., MSY itself), as well as analytical and conceptual proxies for each of the three of these quantities.

[3] Link to the Harvest Strategy Standard: http://fs.fish.govt.nz/Page.aspx?pk=61&tk=208&se=&sd=Asc&filSC=&filAny=False&filSrc=False&filLoaded=False&filDCG=9&filDC=0&filST=&filYr=0&filAutoRun=1.

text on the 'Status of the Stocks' in order to determine whether the latest reported stock status summary is still relevant; else to revise the evaluations of stock status based on new data or analyses, or other relevant information.

WORKING GROUP REPORTS

10. To include in the Working Group report information on commercial, Māori customary, non-commercial and recreational interests in the stock; as well as all other mortality to that stock caused by fishing, which might need to be allowed for before setting a TAC or TACC.
11. To provide information and advice on other management considerations (e.g., area boundaries, bycatch issues, effects of fishing on habitat, other sources of mortality, and input controls such as mesh sizes and minimum legal sizes) required for specifying sustainability measures. Sections of the Working Group reports related to bycatch and other environmental effects of fishing will be reviewed by the Aquatic Environment Working Group although the relevant FAWG is encouraged to identify to the AEWG Chair any major discrepancies between these sections and their understanding of the operation of relevant fisheries.
12. To summarise the stock assessment methods and results, along with estimates of MSY-compatible references points and other metrics that may be used as benchmarks for assessing stock status.
13. To review, and update if necessary, the 'Status of the Stocks' sections of the Fisheries Assessment Plenary report for all stocks under the purview of individual FAWGs (including those for which a full assessment has not been conducted in the current year) based on new data or analyses, or other relevant information.
14. For all important stocks, to complete (and/or update) the Status of Stocks Stocks tables using the template provided in the introductory chapter of the most recent May and November Plenary reports.
15. It is desirable that full agreement amongst technical experts is achieved on the text of the FAWG reports, particularly the 'Status of the Stocks' sections, noting that the AEWG will review sections on bycatch and other environmental effects of fishing. If full agreement amongst technical experts cannot be reached, the Chair will determine how this will be depicted in the FAWG report, will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes.

WORKING GROUP INPUT TO THE PLENARY

16. To advise the Principal Advisor Fisheries Science about stocks requiring review by the Fisheries Assessment Plenary and those stocks that are not believed to warrant review by the Plenary. The general criteria for determining which stocks should be discussed by the Plenary are that (i) the assessment is controversial and Working Group members have had difficulty reaching consensus on a base case, (ii) the assessment is the first for a particular stock or the methodology has been substantially altered since the last assessment, and (iii) new data or analyses have become available that alter the previous assessment, particularly assessments of recent or current stock status, or projections of likely future stock status. Such information could include:

    - new or revised estimates of MSY-compatible reference points, recent or current biomass, productivity or yield projections;
    - the development of a major trend in the catch or catch per unit effort; or
    - any new studies or data that extend understanding of stock structure, fishing patterns, or non-commercial activities, and result in a substantial effect on assessments of stock status.

19.5 MEMBERSHIP AND PROTOCOLS FOR ALL SCIENCE WORKING GROUPS

This document summarises the protocols for membership and participation in all Science Working Groups including Fisheries Assessment Working Groups (FAWGs), the Aquatic Environment Working Group (AEWG), the Biodiversity Research Advisory Group (BRAG), the Highly Migratory Species Working Group (HMS), the South Pacific Working Group (SPACWG), the Antarctic Working Group

(ANTWG), and the Marine Amateur Fisheries Working Group (MAFWG).

WORKING GROUP CHAIRS

1. The Ministry will select and appoint the Chairs for Science Working Groups. The Chair will be an MPI fisheries or marine scientist who is an active participant in the Working Group, providing technical input, rather than simply being a facilitator. Working Group Chairs will be responsible for:

    - ensuring that Working Group participants are aware of the Terms of Reference for the Working Group, and that the Terms of Reference are adhered to by all participants;
    - setting the rules of engagement, facilitating constructive questioning, and focusing on relevant issues;
    - ensuring that all peer review processes are conducted in accordance with the Research and Science Information Standard for New Zealand Fisheries[4] (the Research Standard), and that research and science information is reviewed by the relevant Working Group against the *P R I O R* principles for science information quality (page 6) and the criteria for peer review (pages 12–16) in the Standard;
    - requesting and documenting the affiliations of participants at each Working Group meeting that have the potential to be, or to be perceived to be, a conflict of interest of relevance to the research under review (refer to page 15 of the Research Standard). Chairs are responsible for managing conflicts of interest, and ensuring that fisheries management implications do not jeopardise the objectivity of the review or result in biased interpretation of results;
    - ensuring that the quality of information that is intended or likely to inform fisheries management decisions, the development of environmental standards or the formulation of relevant fisheries policy is ranked in accordance with the information ranking guidelines in the Research Standard (page 21–23), and that resulting information quality ranks are appropriately documented in the Plenary and the Aquatic Environment and Biodiversity Annual Review (AEBAR);
    - striving for consensus while ensuring the transparency and integrity of research analyses, results, conclusions and final reports; and
    - reporting on Working Group recommendations, conclusions and action items; and ensuring follow-up and communication with the MPI Principal Advisor Fisheries Science, relevant MPI fisheries management staff, and other key stakeholders.

WORKING GROUP MEMBERS

2. Membership of Science Working groups will be open to any participant with the agreement of the Working Group Chair.
3. Working Groups will consist of the following participants:

    - MPI fisheries science chair – required;
    - research providers – required (may be the primary researcher, or a designated substitute capable of presenting and discussing the agenda item);
    - other scientists not conducting the presented research to act in a peer review capacity;
    - representatives of relevant MPI fisheries management teams; and
    - any interested party who agrees to the standards of participation below.

4. Working Group participants must commit to:

    - participating appropriately in the discussion;
    - resolving issues;

[4] Ministry of Fisheries (2011) Research and Science Information Standard for New Zealand Fisheries. Retrieved from https://www.mpi.govt.nz/dmsdocument/3692-research-and-science-information-standard-for-new-zealand-fisheries.

- following up on agreements and tasks;
- maintaining confidentiality of Working Group discussions and deliberations (unless otherwise agreed in advance, and subject to the constraints of the Official Information Act);
- adopting a constructive approach;
- avoiding repetition of earlier deliberations, particularly where agreement has already been reached;
- facilitating an atmosphere of honesty, openness and trust;
- respecting the role of the Chair; and
- listening to the views of others, and treating them with respect.

5. Participants in Working Group meetings will be expected to declare their sector affiliations and contractual relationships to the research under review, and to declare any substantial conflicts of interest related to any particular issue or scientific conclusion.

6. Working Group participants must adhere to the requirements of independence, impartiality and objectivity listed under the Peer Review Criteria in the Research Standard (pages 12–16). It is understood that Working Group participants will often be representing particular sectors and interest groups, and may be expressing the views of those groups. However, when participating in the review of science information, representatives are expected to step aside from their sector affiliations, and to ensure that individual and sector views do not result in bias in the science information and conclusions.

7. Participants in each Working Group will have access to the corresponding sections of the Science Working Group website including the Working Group papers and other information provided in those sections. Access to Science Working Group websites will generally be restricted to those who have a reasonable expectation of attending at least one meeting of a given Science Working Group each year.

8. Working Group members who do not adhere to the standards of participation (paragraph 4), or who use Working Group papers and related information inappropriately (see paragraph 10), may be requested by the Chair to leave a particular meeting or to refrain from attending one or more future meetings. In more serious instances, members may be removed from the Working Group membership and denied access to the Working Group website for a specified period of time.

WORKING GROUP PAPERS AND RELATED INFORMATION

9. Working Group papers will be posted on the MPI-Fisheries website prior to meetings if they are available. As a general guide, PowerPoint presentations and draft or discussion papers should be available at least two working days before a meeting, and near-final papers should be available at least five working days before a meeting if the Working Group is expected to agree to the paper. However, it is also likely that some papers will be made available for the first time during the meeting due to time constraints. If a paper is not available for sufficient time before the meeting, the Chair may provide for additional time following the meeting for additional comments from Working Group members.

10. Working Group papers are 'works in progress' intended to facilitate the discussion of analyses by the Working Groups. They often contain preliminary results that are receiving peer review for the first time and, as such, may contain errors or preliminary analyses that will be superseded by more rigorous work. **For these reasons, no-one may release the papers or any information contained in these papers to external parties. In general, Working Group papers should not be cited.** Exceptions may be made in rare instances by obtaining permission in writing from the Principal Advisor Fisheries Science, and the authors of the paper. It is also anticipated that Working Group participants who are representing others at a particular Working Group meeting or series of such meetings may wish to communicate preliminary results to the people they are representing. Participants, along with recipients of the information, are required to exercise discretion in doing this, and to guard against preliminary results being made public.

11. From time to time, MPI commissions external reviews of analyses, models or issues. Terms of Reference for these reviews and the names of external reviewers may be provided to the Working Group for information or feedback. It is extremely important to the proper conduct of these reviews that all contact with the reviewers is through the Chair of the Working Group or the Principal Advisor Fisheries Science. Under no circumstances should Working Group members approach reviewers directly until after the final report of the review has been published.

WORKING GROUP MEETINGS

12. Meetings will take place as required, generally January–April and July–November for FAWGs and throughout the year for other Working Groups (AEWG, BRAG, HMSWG, SPACWG, ANTWG and MAFWG).
13. A quorum will be reached when the Chair, the designated presenter, and at least three other technical experts are present. In the absence of a quorum, the Chair may decide to proceed as a sub-group, with outcomes being discussed with the wider Working group via email or taken forward to the next meeting at which a quorum is formed.
14. The Chair is responsible for deciding, with input from the entire Working Group, but focusing primarily on the technical discussion and the views of technical expert members:

    - the quality and acceptability of the information and analyses under review;
    - the way forward to address any deficiencies;
    - the need for any additional analyses;
    - contents of research reports, Working Group reports and AEBAR chapters;
    - choice of best models and sensitivity analyses to be presented; and
    - the status of the stocks, or the status/performance in relation to any relevant environmental standards or targets.

15. The Chair is responsible for facilitating a consultative and collaborative discussion.
15. Working Group meetings will be run formally, with agendas pre-circulated, and formal records kept of recommendations, conclusions and action items.
16. A record of recommendations, conclusions and action items will be posted on the MPI-Fisheries website after each meeting has taken place.
17. Data upon which analyses presented to the Working Groups are based must be provided to MPI in the appropriate format and level of detail in a timely manner (i.e., the data must be available and accessible to MPI; however, data confidentiality concerns mean that some data may not necessarily be made available to Working Group members).
18. Working Group processes will be evaluated periodically, with a view to identifying opportunities for improvement. Terms of Reference and the Membership and Protocols may be updated as part of this review.
19. MPI fisheries scientists and science officers will provide administrative support to the Working Groups.

INFORMATION QUALITY RANKING

20. Science Working Groups are required to rank the quality of research and science information that is intended or likely to inform fisheries management decisions, in accordance with the science information quality ranking guidelines in the Research Standard (pages 21–23). Information quality rankings should be documented in Working Group reports and, where appropriate, in Status of Stock summary tables. Note that:

    - Working Groups are not required to rank all research projects and analyses, but key pieces of information that are expected or likely to inform fisheries management decisions, the development of environmental decisions or the formulation of relevant policy should receive a quality ranking;
    - explanations substantiating the quality rankings will be included in Working Group reports. In particular, the quality shortcomings and concerns for moderate/mixed and low quality information should be documented; and
    - the Chair, working with participants, will determine which pieces of information require a quality ranking. Not all information resulting from a particular research project

would be expected to achieve the same quality rank, and different quality ranks may be assigned to different components, conclusions or pieces of information resulting from a particular piece of research.

RECORD KEEPING

21. The overall responsibility for record-keeping rests with the Chair of the Working Group, and includes:

- keeping notes on recommendations, conclusions and follow-up actions for all Working Group meetings, and to ensure that these are available to all members of the Working Group and the Principal Advisor Fisheries Science in a timely manner. If full agreement on the recommendations or conclusions cannot readily be reached amongst technical experts, then the Chair will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes; and
- compiling a list of generic assessment issues and specific research needs for each stock, species or environmental issue under the purview of the Working Group, for use in subsequent research planning processes.

19.6 FISHERIES 2030

USE OUTCOME – Fisheries resources are used in a manner that provides the greatest overall economic, social, and cultural benefit. This means having:

- An internationally competitive and profitable seafood industry that makes a significant contribution to our economy.
- High-quality amateur fisheries that contribute to the social, cultural, and economic well-being of all New Zealanders.
- Thriving customary fisheries, managed in accordance with kaitiakitanga, supporting the cultural well-being of iwi and hapū.
- Healthy fisheries resources in their aquatic environment that reflect and provide for intrinsic and amenity value.

GOVERNANCE CONDITIONS – Fundamental to achieving our goal is the recognition that our approach must be based on sound governance. This means having arrangements that lead to:

- The Treaty partnership being realised through the Crown and Māori clearly defining their respective rights and responsibilities in terms of governance and management of fisheries resources.
- The public having confidence and trust in the effectiveness and integrity of the fisheries and aquaculture management regimes.
- All stakeholders having rights and responsibilities related to the use and management of fisheries resources that are understood and for which people can be held individually and collectively accountable.
- Having an enabling framework that allows stakeholders to create optimal economic, social, and cultural value from their rights and interests.
- An accountable, responsive, dynamic, and transparent system of management.

Fisheries 2030 draws on a number of values and principles. These seek to outline the behaviour and approach that should be used to undertake the actions, make decisions, and achieve the goal for New Zealand fisheries.

VALUES

- Tikanga: the Mäori way of doing things; correct procedure, custom, habit, lore, method, manner, rule, way, code, meaning, reason, plan, practice, convention. It is derived from the word tika meaning ‘right’ or ‘correct’.
- Kaitiakitanga: The root word in kaitiakitanga is tiaki, which includes aspects of guardianship, care, and wise management. Kaitiakitanga is the broad notion applied in different situations.
- Kotahitanga: Collective action and unity.
- Manaakitanga: Manaakitanga implies a duty to care for others, in the knowledge that at some time others will care for you. This can also be translated in modern Treaty terms as ‘create no further grievances in the settlement of current claims’.
- Integrity: Be honest and straightforward in our dealings with one another. If we agree to do something we will carry it out.

- Respect: Treat each other with courtesy. We will respect each other's right to have different values and hold different opinions.
- Constructive relationship: Strive to build and maintain constructive ways of working with each other, which can endure.
- Achieving results: Focus on producing a solution rather than just discussing the problem.

PRINCIPLES

- Ecosystem-based approach: We apply an ecosystem-based approach to fisheries management decision-making.
- Conserve biodiversity: Use should not compromise the existence of the full range of genetic diversity within and between species.
- Environmental bottom lines: Biological standards define the limits of extraction and impact on the aquatic environment.
- Precautionary approach: Particular care will be taken to ensure environmental sustainability where information is uncertain, unreliable, or inadequate.
- Address externalities: Those accessing resources and space should address the impacts their activities have on the environment and other users.
- Meet Settlement obligations: Act in ways that are consistent with the Treaty of Waitangi principles and deliver settlement obligations.
- Responsible international citizen: Manage in the context of international rights, obligations, and our strategic interests.
- Inter-generational equity: Current use is achieved in a manner that does not unduly compromise the opportunities for future generations.
- Best available information: Decisions need to be based on the best available and credible biological, economic, social, and cultural information from a range of sources.
- Respect rights and interests: Policies should be formulated and implemented to respect established rights and interests.
- Effective management and services: Use least-cost policy tools to achieve objectives where intervention is necessary and ensure services are delivered efficiently.
- Recover management costs for the reasonable expenses of efficiently provided management and services, from those who benefit from use, and those who cause the risk or adverse effect.
- Dynamic efficiency: Frameworks should be established to allow resources to be allocated to those who value them most.

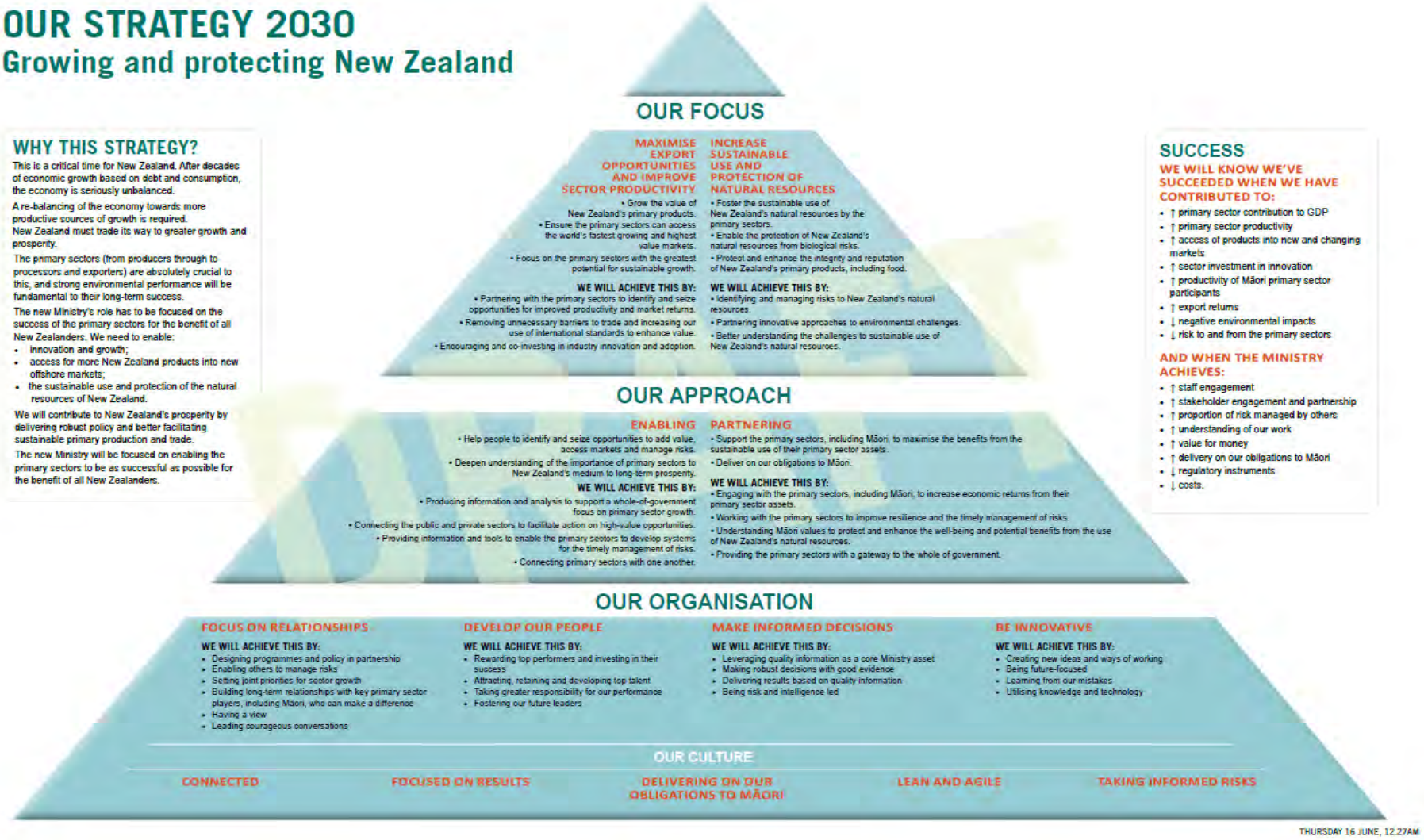
Fisheries 2030 includes a 'plan of action' for the five years from 2009, including: improving the management framework; supporting aquaculture and international objectives; ensuring sustainability of fish stocks; improving fisheries information; building sector leadership and capacity; meeting obligations to Māori; and enabling collective management responsibility. The key components guiding this document are ensuring sustainability of fish stocks and improving fisheries information.

ENSURING SUSTAINABILITY OF FISH STOCKS

- Setting and implementing fisheries harvest strategy standards.
- Setting and monitoring environmental standards, including for threatened and protected species and seabed impacts.
- Enhancing the framework for fisheries management planning, including the use of decision rules to adjust harvest levels over time.

IMPROVING FISHERIES INFORMATION

- Determining best options for information collection on catch from amateur fisheries, including the implementation of charter boat reporting.
- Improving our knowledge of fish stocks and the environmental impacts of fishing through long-term research plans.
- Gaining access to increased research and development funding.



Appendix 19.6.1: Our Strategy 2030: Growing and protecting New Zealand.

19.7 OTHER STRATEGIC POLICY DOCUMENTS

19.8.1 BIODIVERSITY STRATEGY

New Zealand's Biodiversity Strategy was launched in 2000 in response to the decline of New Zealand's indigenous biodiversity — described in the State of New Zealand's Environment report as our 'most pervasive environmental issue'. It can be found on the government's biodiversity website at: http://www.doc.govt.nz/biodiversity.

The Strategy also reflects New Zealand's commitment, through ratification of the international Convention on Biological Diversity, to help stem the loss of biodiversity worldwide. Strategic Priority 7 of the strategy was 'To manage the marine environment to sustain biodiversity'. Fishing practices, the effects of activities on land, and biosecurity threats are identified as constituting the areas of greatest risk to marine biodiversity. Pertinent objectives and summarised actions from the strategy are as follows:

Objective 3.1: Improving our knowledge of coastal and marine ecosystems (Substantially increase our knowledge of coastal and marine ecosystems and the effects of human activities on them, especially assessing the importance of, and threats facing, marine biodiversity, and establishing environmental monitoring capabilities to assess the effectiveness of measures to avoid, remedy or mitigate impacts on marine biodiversity).

Objective 3.4: Sustainable marine resource use practices (Protect biodiversity in coastal and marine waters from the adverse effects of fishing and other coastal and marine resource uses, especially maintaining harvested species at sustainable levels, integrating marine biodiversity protection into an ecosystem approach, applying a precautionary approach, identifying marine species and habitats most sensitive to disturbance, and integrating environmental impact assessments into fisheries management decision making.)

Objective 3.6: Protecting marine habitats and ecosystems (Protect a full range of natural marine habitats and ecosystems to effectively conserve marine biodiversity, using a range of appropriate mechanisms, including legal protection, especially establishing a network of areas that protect marine biodiversity.)

Objective 3.7: Threatened marine and coastal species management (Protect and enhance populations of marine and coastal species threatened with extinction, and prevent additional species and ecological communities from becoming threatened.)

In addition to its annual reviews, the Biodiversity Strategy was reviewed by Green and Clarkson at the end of its 5-year term. This review was published in 2006 (http://www.doc.govt.nz/documents/conservation/nzbs-report.pdf). Most relevant to this synopsis were their findings on Objective 3.4 (Sustainable marine resource use) where they cited 'Moderate progress'. *'The policy move towards adopting a more ecosystem approach to fisheries management should be encouraged and strengthened. We acknowledge, however, the difficulties associated with obtaining the necessary information to make this approach effective. There are links to Objective 3.1 and the need for a more coordinated approach to identifying priority areas for marine research.'*

19.8.2 BIOSECURITY STRATEGY

In its 2003 Biosecurity Strategy, the Ministry of Agriculture and Forestry's Biosecurity NZ defined biosecurity as 'the exclusion, eradication or effective management of risks posed by pests and diseases to the economy, environment and human health'. New Zealand is highly dependent on effective biosecurity measures because our indigenous flora, fauna, biodiversity, and, consequently, our primary production industries, including fisheries are uniquely at risk from invasive species. A complementary Biosecurity Science Strategy for New Zealand was developed in 2007 to address the science expectations of the Biosecurity Strategy. The science strategy identified the need to:

- prioritise science needs;
- minimise biosecurity risks at the earliest stage possible by increasing focus on research that is strategic and proactive;
- improve planning, integration and communication in the delivery of science;
- ensure research outputs can be used effectively to improve biosecurity operations and decision making.

19.8.3 MARINE PROTECTED AREAS POLICY

The Marine Protected Areas (MPA) Policy and Implementation Plan was released for consultation in December 2005 jointly by the Ministry of Fisheries and Department of Conservation. It confirmed Government's commitment to ensuring that New Zealand's marine biodiversity was protected, and established MPA Policy as a key component of that commitment. The MPA Policy objective is to protect marine biodiversity by establishing a network of Marine Protected Areas that is comprehensive and representative of New Zealand's marine habitats and ecosystems. The Policy involved a four-stage approach to implementation:

Stage 1: Development of the approach to classification, formulation of a standard of protection, and mapping of existing protected areas and/or mechanisms. Scientific workshops will be used to assist with the process, and the results will be put on the website for comment.

Stage 2: Development of the MPA inventory, identification of gaps in the MPA network, and prioritisation of new MPAs.

Stage 3: Establishment of new MPAs to meet gaps in the network. This will be undertaken at a regional level and a national process will be followed for offshore MPAs.

Stage 4: Evaluation and monitoring.

Stage 1 and the inventory specified for Stage 2 are complete and regional forums were established for the Subantarctic and West Coast bioregions.

The link for the stage 2 report is at:

http://www.doc.govt.nz/publications/conservation/marine-and-coastal/marine-protected-areas/coastal-marine-habitats-and-marine-protected-areas-in-the-new-zealand-territorial-sea-a-broad-scale-gap-analysis.

In June 2009, these planning forums released consultation documents on implementation of the MPA Policy in their bioregions:

Consultation Document – Implementation of the Marine Protected Areas Policy in the Territorial Seas of the Subantarctic Biogeographic Region of New Zealand:

http://www.marinenz.org.nz/documents/subantarctics-mpa-policy-consultation-document.pdf.

The MPA Classification, Protection Standard, Implementation Guidelines, together with a summary of subsequent consultation processes around implementing the policy can be found at:

http://www.doc.govt.nz/Documents/conservation/marine-and-coastal/marine-protected-areas/mpa-classification-protection-standard.pdf.

19.8.4 REVISED COASTAL POLICY STATEMENT

The revised New Zealand Coastal Policy Statement (NZCPS) came into force in December 2010, replacing the original 1994 NZCPS. The statement is to be applied, as required by the Resource Management Act 1991 (RMA), by persons exercising functions and powers under that Act. The documentation can be read on the Department of Conservation's website at:

http://www.doc.govt.nz/publications/conservation/marine-and-coastal/new-zealand-coastal-policy-statement/new-zealand-coastal-policy-statement-2010.

The NZCPS does not directly apply to fisheries management decision-making, although the Minister of Fisheries is required to have regard to the Statement when making decisions on sustainability measures under s11 of the Fisheries Act. In addition, this synopsis include chapters on land use issues and habitats of particular significance for fisheries management for which the main threats are managed under the RMA (e.g., land-use practices could increase sedimentation and affect the estuarine nursery grounds of important fishstocks). In other areas, management of effects under the RMA can complement management of the effects of fishing (e.g., complementary management of the habitat and bycatch of a protected species). The following objectives and policies are considered relevant (numbering as per NZCPS, text in parentheses summarises subheadings in the Statement of most relevance to fisheries values):

Objective 1: To safeguard the integrity, form, functioning and resilience of the coastal environment and sustain its ecosystems, including marine and intertidal areas, estuaries, dunes and land (especially by maintaining or enhancing natural biological and physical processes in the coastal environment).

Objective 6: To enable people and communities to provide for their social, economic, and cultural wellbeing and their health and safety, through subdivision, use, and development (especially by recognising that the protection of habitats of living marine resources contributes to social, economic and cultural wellbeing and that the potential to utilise coastal marine natural resources should not be compromised by activities on land).

Policy 5: Land or waters managed or held under other Acts (especially to consider effects on coastal areas held or managed under other Acts with conservation or protection purposes and to avoid, remedy or mitigate adverse effects of activities in relation to those purposes).

Policy 8: Aquaculture: Recognise the significant existing and potential contribution of aquaculture to the social, economic and cultural well-being of people and communities (especially by taking account of the social and economic benefits of aquaculture, recognising the need for high water quality, and including provision for aquaculture in the coastal environment).

Policy 11: Indigenous biodiversity: To protect indigenous biological diversity in the coastal environment (especially by avoiding, remedying or mitigating adverse effects on: habitats that are important during the vulnerable life stages of indigenous species; ecosystems and habitats that are particularly vulnerable to modification; and habitats of indigenous species that are important for recreational, commercial, traditional or cultural purposes).

Policy 21: Enhancement of water quality: Where the quality of water in the coastal environment has deteriorated so that it is having a significant adverse effect on ecosystems, natural habitats, or water based recreational activities, or is restricting existing uses, such as aquaculture, shellfish gathering, and cultural activities, give priority to improving that quality.

Policy 22: Sedimentation (especially with respect to impacts on the coastal environment).

Policy 23: Discharge of contaminants (especially with respect to impacts on ecosystems and habitats).

19.8.5 MANAGEMENT OF ACTIVITIES IN THE EEZ

Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012. The Act manages the environmental effects of activities in New Zealand's oceans. The legislation aims to protect our oceans from the potential environmental risks of activities like petroleum exploration activities, seabed mining, marine energy generation and carbon capture developments.

The Resource Management Act regulates natural resource management activities on land and in the Territorial Sea out to 12 nautical miles. Fishing and shipping are also regulated by other Acts. The EEZ Act does not override these other controls that already exist in the EEZ. Beyond 12 nautical miles New Zealand has historically had no means to assess and regulate the environmental effects of many other activities. The EEZ Act fills that regulatory gap and manages the previously unregulated adverse environmental effects of activities in the EEZ and continental shelf. Before the EEZ Act was passed there was a gap in our domestic legislation.

The EEZ Act sets up a framework for managing the effects of activities in the EEZ and continental shelf. The text of the Act can be found on the New Zealand Legislation website.

The EEZ legislation to manage effects other than those caused by fishing do not directly apply to fisheries management decision-making under the Fisheries Act. However, there are issues around the management of cumulative effects (e.g., of more than one activity on benthic communities) and around effects of any proposed new activities in the EEZ on fishing activity already occurring. Some projects already completed or currently underway are likely to be useful for these processes (e.g., detailed maps of fishing effort produced under ENV2001/07 and BEN2006/01 and enhancements of the Marine Environment Classification produced under ZBD2005-02 for demersal fishes and BEN2006/01A for benthic invertebrates).

19.8.6 NATIONAL PLAN OF ACTION TO REDUCE THE INCIDENTAL CATCH OF SEABIRDS IN NEW ZEALAND FISHERIES

New Zealand released its first National Plan of Action (NPOA) to reduce the Incidental Catch of Seabirds in New

Zealand Fisheries in April 2004. That document is available online at:

http://www.doc.govt.nz/documents/conservation/native-animals/birds/npoa.pdf.

A completely revised and refreshed NPOA-Seabirds was released in March 2013. A resources page was added to the MPI (Fisheries) website to provide access to this plan, its supporting risk assessment documents, a web-based reporting system for protected species captures, and information on MPI's fisheries planning processes that will be the vehicle for implementation:

https://fs.fish.govt.nz/Page.aspx?pk=108.

The 2013 NPOA-Seabirds can be found at:

https://www.mpi.govt.nz/dmsdocument/3962-national-plan-of-action-2013-to-reduce-the-incidental-catch-of-seabirds-in-new-zealand-fisheries.

The 2013 NPOA covers all New Zealand fisheries and has a long-term objective that *'New Zealand seabirds thrive without pressure from fishing related mortalities, New Zealand fishers avoid or mitigate against seabird captures and New Zealand fisheries are globally recognised as seabird friendly.'*

There are high-level subsidiary objectives related to practical aspects, biological risk, research and development, and international issues.

i. Practical objective: All New Zealand fishers implement current best-practice mitigation measures relevant to their fishery and aim through continuous improvement to reduce and where practicable eliminate the incidental mortality of seabirds.
ii. Biological risk objective: Incidental mortality of seabirds in New Zealand fisheries is at or below a level that allows for the maintenance at a favourable conservation status or recovery to a more favourable conservation status for all New Zealand seabird populations.
iii. Research and Development objectives:
    a. the testing and refinement of existing mitigation measures and the development of new mitigation measures results in more practical and effective mitigation options that fishers readily employ;
    b. research and development of new observation and monitoring methods results in improved cost effective assurance that mitigation methods are being deployed effectively; and
    c. research outputs relating to seabird biology, demography and ecology provide a robust basis for understanding and mitigating seabird incidental mortality.
iv. International objective: In areas beyond the waters under New Zealand jurisdiction, fishing fleets that overlap with New Zealand breeding seabirds use internationally accepted current best practice mitigation measures relevant to their fishery.

19.8.7 NEW ZEALAND NATIONAL PLAN OF ACTION FOR THE CONSERVATION AND MANAGEMENT OF SHARKS

The New Zealand National Plan of Action (NPOA) for the Conservation and Management of Sharks (2013) was approved by the Minister of Fisheries on 9 January 2014. The purpose of the NPOA-Sharks is to ensure the conservation and management of sharks and their long-term sustainable use. It also contains a set of actions in order to meet this purpose. The document is available online at:

http://www.mpi.govt.nz/dmsdocument/1138-national-plan-of-action-for-the-conservation-and-management-of-sharks-2013.

19.8.8 NATIONAL SCIENCE CHALLENGES

The National Science Challenges were conceived to tackle some of the biggest science-based issues and opportunities facing New Zealand. They were designed to take a more strategic approach to the government's science investment by targeting a series of goals, which, if achieved, would have major and enduring benefits for New Zealand. The Challenges provide an opportunity to align and focus New Zealand's research on large and complex issues by drawing scientists together from different institutions and across

disciplines to achieve a common goal through collaboration.

Many of the issues facing New Zealand require new knowledge obtained through science and research. The government has launched the Challenges to provide a means to address the most pressing of these complex issues. The Challenges will seek answers to questions of national significance to New Zealand by focusing effort and providing additional focus on key areas. The Challenges provide an opportunity to identify which issues are most important to New Zealand and will allow government to take a targeted, cross-government approach to addressing them.

Each Challenge includes both new funding and funds that will become available as current MBIE research contracts mature. Relevant CRI core funding will also be invested in Challenges, where CRIs are part of a Challenge collaboration. The new Challenge money comprises $73.5 million over four years in Budget 2013, in addition to the $60 million allocated in Budget 2012, and $30.5 million per year thereafter.

The eleven research areas identified for National Science Challenge funding (asterisks mark those Challenges potentially relevant to fisheries and the marine environment) were:

1. High Value Nutrition
2. The Deep South *
3. New Zealand's Biological Heritage *
4. Sustainable Seas *
5. A Better Start
6. Resilience to Nature's Challenges *
7. Science for Technological Innovation
8. Ageing Well
9. Healthier Lives
10. Our Land and Water *
11. Building Better Homes, Towns and Cities

See also: http://www.mbie.govt.nz/info-services/science-innovation/national-science-challenges.

The Ministry for Business, Innovation and Employment administers the Challenges and issued Requests for Proposals for four of the Challenges in October 2013 and for the remainder in February 2014. Given that the Challenges represent a radically different approach to research in New Zealand, and required substantial collaboration between science organisations, it is perhaps not surprising that designing and contracting the work has taken some time. The following Challenges of relevance to fisheries and marine systems have been launched (as at December 2014, listed in order of their launch):

The Deep South — Te Kōmata o Te Tonga — was launched on 5 August 2014 with a headline of *Understanding the role of the Antarctic and the Southern Ocean in determining our climate and our future environment*. The mission of this Challenge is to transform the way New Zealanders adapt, manage risk, and thrive in a changing climate. Working with communities and industry we will bring together new research approaches to determine the impacts of a changing climate on our climate-sensitive economic sectors, infrastructure and natural resources to guide planning and policy. This will be underpinned by improved knowledge and observations of climate processes in the Southern Ocean and Antarctica – our Deep South – and will include development of a world-class earth systems model to predict Aotearoa/New Zealand's climate. Further information can be found at: http://www.deepsouthchallenge.co.nz.

New Zealand's Biological Heritage — Ngā Koiora Tuku Iho — was launched on 29 August 2014 with a headline of *Protecting and managing our biodiversity, improving our biosecurity, and enhancing our resilience to harmful organisms*. This Challenge does not consider marine systems as such, but includes estuarine systems and close liaison between this Challenge and Sustainable Seas will be necessary to ensure important biological systems and processes are covered. Further information can be found at: http://www.biologicalheritage.nz.

Sustainable Seas — Ko ngā moana whakauka — was launched on 4 September 2014 with a headline of *Enhance utlilisation of our marine resources within environmental and biological constraints*. The aim of this Challenge is to enhance use of New Zealand’s vast marine resources, while ensuring that our marine environment is understood, cared for, and used wisely for the benefit of all, now and in the future. This requires a new way of managing the many uses of our marine resources that combines the aspirations and experience of Māori, communities, and industry with the evidence of scientific research to transform New Zealand into a world-leader in sustainable marine economic development. Thus, this is the Challenge most closely

associated with fisheries management. Further information can be found at: http://sustainableseaschallenge.co.nz.

19.8 APPENDIX OF AQUATIC ENVIRONMENT AND BIODIVERSITY FUNDED AND RELATED PROJECTS

The following listing of projects are those relevant to aquatic environment research that have been through research planning and subsequently been funded by the Ministry of Fisheries (MFish), the Ministry for Primary Industries (MPI) or the fishing industry. These projects have been ordered by the research themes:

1. Protected species (PRO)
2. Non-protected bycatch (NPB)
3. Benthic impacts (BEN)
4. Ecosystem effects (ECO)
5. Biodiversity (ZBD)

Within these themes projects are ordered chronologically (from the most recent to the oldest). A list of references cited within the table is included at the end of this appendix.

Each project or row of the table is described by a project number (used by MFish/MPI), a project title, specific objectives (where there are many objectives and some are clearly not relevant to aquatic environment research they may not be listed), project status and any relevant citations from the project.

Citations listed below can be accessed differently depending upon the type of output. Finalised FARs (Fisheries Assessment Reports) and AEBRs (Aquatic Environment and Biodiversity Reports), historical FARDs (Fisheries Assessment Research Documents) and MMBRs (Marine Biodiversity and Biosecurity Reports), and some FRRs (final Research Reports) can be found at: http://fs.fish.govt.nz/Page.aspx?pk=61&tk=209.
Increasingly, reports will be available from the MPI website at: https://www.mpi.govt.nz/news-and-resources/publications. For unpublished documents or those not available on either of these websites please contact Science.Officer@mpi.govt.nz. Every attempt has been made to make this table comprehensive and correct, but if any errors are found please send suggested corrections or additions through to Science.Officer@mpi.govt.nz.

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	PRO2017-01A	Research into the demographic parameters for at-risk seabirds as identified by the Risk Assessment (black petrels)	1. To collect information on population size, adult survival, age at first reproduction and key demographic parameters for black petrel to reduce uncertainty or bias in estimates of risk. 2. To collect spatial data to allow refinement of the spatial overlap with fishing, with tracking devices to be deployed on a wider range of ages/breeding stages.	Contracted, in progress	
PRO	PRO2017-01B	Research into the demographic parameters for at-risk seabirds as identified by the Risk Assessment (Southern Buller's/Snares)	To collect information on population size, adult survival, age at first reproduction and key demographic parameters for southern Buller's on The Snares to reduce uncertainty or bias in estimates of risk.	Approved, not yet contracted	
PRO	PRO2017-04	Risk Assessment to support the development of revised NPOA seabirds	1. To explore the recommendations made by the expert review of the risk assessment framework due to occur in June 2017, via trialling any suggested changes to the methodology or undertaking sensitivity runs. 2. Following the methods as described in the AEBAR and agreed changes from the expert review, construct a spatially explicit fisheries risk assessment for seabirds.	Approved, not yet contracted	
PRO	PRO2017-05A	Population specific modelling of adult survival of black petrels	To update previous population modelling of black petrels to produce an updated population trend and estimate of adult survival.	Contracted	
PRO	PRO2017-05B	Population specific modelling of adult survival of Chatham island albatross	To compile all mark-recapture data collected for the Chatham Island albatross and produce an adult survival estimate for the time period corresponding to that used by the risk assessment.	Contracted, in progress	
PRO	PRO2017-06	Characterisation of yellow eyed penguin / fishery interactions	To undertake a review and characterisation of all available information to better understand when, where and how yellow-eyed penguins become caught in set nets, and to the extent possible, the frequency of occurrence.	Approved, not yet contracted	
PRO	PRO2017-08A	Research into the demographic parameters for at-risk marine mammals as identified by the marine mammal risk	Characterise population structure and estimate population size for New Zealand common dolphin population(s), with an emphasis on populations that are most exposed to fisheries risk.	Contracted, in progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		assessment (common dolphins)			
PRO	PRO2017-08C	Research into the demographic parameters for at-risk marine mammals as identified by the marine mammal risk assessment (sea lions)	1. To investigate the likely causes and consequences of changing New Zealand sea lion pup mass and pup survival, using biological and population monitoring data collected at the Auckland islands both before and after the advent of population decline (i.e., pre- and post-2000). 2. To investigate the extent to which indices derived from the analyses in Objective 1 can be used to predict sea lion population trends, by incorporation into the existing sea lion population model and/or as a relevant indicator for future population monitoring efforts.	Contracted, in progress	
PRO	PRO2017-10	Analysis of New Zealand sea lion tracking data to estimate overlap with fisheries	1. Characterise the foraging behaviour of Auckland Islands' sea lions in a spatially and temporally explicit manner using available satellite telemetry data. 2. Apply spatial overlap methods to inform improved estimation of encounter rate, strike rate, and cryptic mortality rate of Auckland Islands' sea lions with commercial fisheries over time, including for fishing effort with and without the use of Sea Lion Exclusion Devices (SLED). 3. Apply estimates from Objective 2 (with uncertainty) to inform spatially explicit estimates of fishery related deaths in association with current fishing effort patterns.	Contracted, in progress	
PRO	PRO2017-12	Hector's and Māui dolphin multi-threat risk assessment to support review of the TMP	1. Construct population models for Māui dolphins, and for Hector's dolphins in those regional sub-populations where data are sufficient. 2. Map potential non-fishery threats to Māui and Hector's dolphins and estimate the overlap between dolphin distributions and both fishery and non-fishery threats. 3. Apply the Spatially Explicit Fisheries Risk Assessment (SEFRA) method to estimate fisheries impact and risk to Māui and Hector's dolphins, using the new information in the objectives above, including at a regional sub-population level. This analysis should include estimation and partition of total mortalities attributable to different threats (with uncertainty) at a regional sub-population level. 4. In consultation with government scientists and managers examine alternate spatial management scenarios through both modelling and participation in a multi-threat risk assessment workshop.	Contracted, in progress	
PRO	PRO2017-15	Use of innovative tag technology to examine foraging patterns of seabirds and association with fishing vessels	To undertake tagging programs alongside other field programs monitoring populations of relevant seabird species using tags that can detect radar strength and potentially depth of dives to examine the relative occurrence of seabird foraging close to fishing vessels.	Contracted, in progress	
PRO	PRO2017-19	Factors affecting capture rate of black petrels and flesh-footed shearwaters	Build a spatially and temporally explicit commercial fisheries risk model estimating capture/ rates of black petrels and flesh-footed shearwaters as a function of multiple spatial, temporal, and vessel- or effort-specific variables potentially affecting capture rates, for fishery groups generating considerable risk to these species. Identify what factors most strongly drive fisheries risk, and evaluate risk reduction options.	Contracted, in progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	SEA2017-10	Black petrel Electronic Monitoring; Audit and Analysis	Audit, data analysis, and report of the footage of seabird captures recorded electronically and by on-board observers collected as a result of the collaborative Black Petrel Electronci Monitoring trial.	Contracted, in progress	
PRO	SEA2017-08	A synthesis of the population work carried out as part of PRO2006-01	Preparation of a report which summerises work under the PRO2006-01 A to E Mfish contracts including White-chinned petrel and grey petrel on Antipodes Island, Salvin's albatross on Snares, Northern buller's albatross, northern royal albatross and northern giant petrel on Fourty-fours and Sisters, and Chatham island albatross on the Pyramid.	Contracted, in progress	
PRO	SEA2017-03	Shark qualitative analysis for risk assessment rerun	Collection from post November 2014 for each shark species assessed in Ford et al. (2015) of plenary chapters, data files, summaries and maps of reported captures over last 5 complete fishing years up to 30 September 2016, heat maps as generated for NABIS layers, trawl survey iformation on distribution and trends and papers or summaries of biology, age, growth, fecundity and general productivity.	Completed	
PRO	PRO2016-01A	Demographic parameters of black petrels	To collect or analyse information on population size, distribution, or key demographic parameters to reduce uncertainty or bias in estimates of risk for selected at-risk seabirds.	Contracted, in progress	
PRO	PRO2016-02	Factors affecting capture rate of black petrels and flesh-footed shearwaters	Build a spatially and temporally explicit commercial fisheries risk model estimating capture/kill rates of black petrels and flesh-footed shearwaters as a function of multiple spatial, temporal, and vessel- or effort-specific variables potentially affecting capture rates, for fishery groups generating considerable risk to these species. Identify what factors most strongly drive fisheries risk, and evaluate alternate risk reduction options.	Withdrawn	
PRO	PRO2016-03	Estimation of captures of protected species in New Zealand Fisheries	To summarise fishing effort, observer effort, and observer reported captures in trawl, longline, set net and purse seine fisheries within the New Zealand EEZ, for the 2016/17, 2016/17 and 2017/18 fishing years. To estimate capture rates and total captures of protected species by method, area, and target fishery, and where possible, by species for the 2016/17, 2016/17 and 2017/18 fishing years.	Contracted, in progress	
PRO	PRO2016-04	Characterisation and quantification of non-fishing threats on seabirds	To characterise and quantify the non-fishing threats to seabirds.	Withdrawn	
PRO	PRO2016-06	Spatially explicit risk assessment query and simulation tool	Build an interactive user-driven query and simulation tool to enable MPI fisheries managers and government scientists to: i) access, query, display, and disaggregate spatially explicit data layers and outputs of the L2 seabird risk assessment for user-defined combinations of fishery groups, species, and/or areas; ii) define and examine the consequences of alternate assumptions and alternate risk management scenarios, including spatial and temporal effort controls, mitigation uptake, and/or new research to reduce biological and statistical uncertainty -- and iii) for each alternate scenario, estimate seabird captures, fatalities, uncertainty, and corresponding risk; iv) simulate the assignment of fisheries observer coverage within user defined scenarios	Contracted, in progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			and estimate the power to accurately estimate seabird captures and risk under different observer coverage levels.		
PRO	PRO2016-09	Abundance and distribution of Hector's and Māui dolphins	1. To develop and refine designs and methods for summer and winter surveys for Hector's dolphins along the SCSI. 2. To estimate the abundance of Hector's dolphins along the SCSI applying an agreed survey and analysis methodology. 3. To estimate the distribution of Hector's dolphins along the SCSI applying an agreed survey and analysis methodology.	Contracted, in progress	
PRO	SEA2016-29	Analysis for the White-capped albatross Aerial Survey	1. Prepare photo montages and count nesting and loafing albatross according to existing methodology (as presented to CSP TWG and AEWG previously) for the aerial surveys undertaken during the summers of 2015/16 and 2016/17 (undertaken under contract to DOC, Deepwater Group and MPI), 2. Based on the counts in objective 1 and previous counts, assess the population trend, taking into account the proportion of loafers identified in the photo montages and by ground counts. 3. Analyse the trends shown by sub-areas for the entire time series of aerial surveys to assess whether selected sub-areas could be monitored and represent the trend of the wider population.	Contracted, in progress	
PRO	SEA2016-19	Spatial methods development to support risk assessment (part II). Estimation of capture and retention efficiency for non-target fish species in commercial trawl fisheries	1. Exploration and testing of alternative methods for species density estimation. 2. Incorporation and propagation of uncertainty in species density estimates. 3. Application of the model with environmental attribute data assigned to individual fishing events.	Contracted	
PRO	SEA2016-20	Helicopter based aerial surveys of the Auckland Islands	Helicopter based aerial surveys of the Auckland Islands.	Complete	
PRO	SEA2016-21	Stocktake of Status of development of Mitigation measures applicable to New Zealand commercial fisheries	Stocktake of Status of development of Mitigation measures applicable to New Zealand commercial fisheries.	Complete	
PRO	SEA2016-21	Stocktake of Status of development of	Report summarising (by fishing method) the bycatch mitigation measures and hurdles to uptake in New Zealand, including development, testing and cost.	Complete	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		Mitigation Measures applicable to New Zealand commercial fisheries			
PRO	SEA2016-24	SEA2016-24 Supplemental sea lion population modelling to support an updated Squid Trawl Fishery Operational Plan	Update the existing Auckland Islands sea lion population model with two additional years of pup count data and estimate population trajectories corresponding to alternate management and hypothetical mortality scenarios.	Contracted	
PRO	SEA2016-26	SEA2016-26 SPRFMO bottom fishing impact assessment	1. Use NZ bottom trawl data from the entire SPRFRMO area and apply the plotImpact method developed for CCAMLR to the SPRFMO area using NZ bottom trawl data. 2. The overall dataset will be divided into the type of fishing (slope, seamount, or mixed) and the impact summaries and histograms of percent impact will be generated for each fishing type at the four spatial scales. 3. The relationship between the cell size and estimated percentage impact will be evaluated for two selected habitats (slope and seamount). For this exercise, the data for a given habitat will be summarised as a distribution of impact percentages for the cells included, and profiled across cell sizes starting at 100m and with increasing cell size to show the relationship between estimated percent impact and cell size. 4. If time allows, illustrate the potential impact and recovery dynamics of an example VME taxon using an assumed spatial distribution and demographic parameters in one area of interest, and an assumed single move on rule (trigger and distance) to redistribute fishing effort to the remaining areas.	Complete	
PRO	SEA2016-30	Hector's and Māui dolphin risk disaggregation tool	1. Expand the custom risk assessment disaggregation and query tool (contract PRO2016-06) to include Hector's and Māui dolphins, incorporating all data inputs for Hector's and Māui dolphins utilised in the Marine Mammal Risk Assessment. Risk estimation will be carried out via a single-species application of the SEFRA method (2017 MPI AEBAR, Chapter 3) for both the setnet and inshore trawl fishery groups. 2. Expand the capability of the risk query tool to include a sub-population definition function that subdivides a population according to user-defined boundaries, and automatically generates separate outputs for each. 3. Expand the capability of the risk query tool to include analysis of hypothetical scenarios using alternate species spatial distribution layers and/or alternate fishing effort distribution layers (provided as user-defined inputs, e.g., as GIS layers) with standardised diagnostic outputs comparing alternate scenarios with the base case scenario. 4. Expand the capability of the risk query tool to allow the fishing effort and observed captures database query and display function to be applied to any protected species in the groomed and linked effort and captures database. This objective is for the visual display outputs only; it does not include risk assessment modelling for species other than Hectors and Māui dolphins.	Contracted	
PRO	DAE2015-01	Characterisation of seabird capture data	To collate and characterise the seabird capture information from deepwater trawl fisheries to improve understanding of potential risk factors for captures of seabirds, with a focus on net captures.	Contracted, in progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	PRO2015-01	Improving estimates of cryptic mortality for use in seabird risk assessments	1. To develop guidelines for the production of estimates of total seabird captures from observer data, with methods varying based on the level and quality of data. 2. To increase the capability of other countries to produce robust estimates of seabird captures.	Contracted, in progress	
PRO	PRO2015-04	Addressing key information gaps for Māui dolphins	1. To collect information on spatial distribution and overlap with fisheries to decrease uncertainty in our understanding and estimates of risk to Māui dolphins.	Contracted, in progress	
PRO	SEA2015-06	Additional aerial survey effort for Hector's dolphins on the West Coast South Island	Following increased Hectors dolphins sightings over the summer survey this project allows for extra effort in the Grey (0-4nm), Hector (4-12nm) and Okarito (4-12nm) strata (Plan A). If time and weather permits this will also allow for extra effort in the 4-12nm strata off Whanganui, Jackson Bay and Milford.	Completed as part of PRO2013-06	Clement & MacKenzie 2016
PRO	SEA2015-10	Sea lion prey survey	1. Undertake a demersal trawl survey of the Auckland Islands and Stewart/Snares shelf to determine the spatial and bathymetric distribution and abundance of the main prey species of NZ sea lions in the areas used by benthic and pelagic foraging lactating females. 2. Conduct a potting feasibility study to determine the distribution, abundance and biology of yellow octopus (Enteroctopus zealandicus). 3. Conduct a benthic habitat characterisation based on acoustic swath mapping of the seafloor in the area immediately surrounding demersal trawl stations. 4. Deploy underwater cameras to visually survey seafloor habitat and sea lion prey species at a representative subsample of habitat types, identified from the acoustic swath habitat characterisation. 5. Make oceanographic observations to quantify physical characteristics of sea lion foraging habitat.	Complete	
PRO	SEA2015-12	Potential impacts of fisheries restrictions for the NZ sea lion TMP	1. To estimate the likely impact on catch rates and total catches of squid, scampi, and hoki of a range of specified potential fishing restrictions. 2. To estimate the likely impact on sea lion interactions and captures of the specified fishing restrictions.	Complete	
PRO	SEA2015-15	Stewart Island sea lion survey	1. To determine the feasibility of monitoring pup production in January, concurrent with the monitoring of other colonies. 2. While testing the feasibility of using thermal technology in finding sea lion pupping locations. 3. To screen for Klebsiella pneumonia. 4. Determine common causes of death for Stewart Island pups.	Complete	Boren et al. 2016
PRO	PRO2014-06	Update of level-2 seabird risk assessment	1. To update the level-2 seabird risk assessment using all new information on bird population size, productivity, and distribution, and all relevant fishing effort and observer data for the 2009/10 to 2013/14 fishing years. 2. To identify key drivers of uncertainty and opportunities to reduce uncertainty in the risk ratios for species at high or very high risk.	Complete	Richard et al. 2017

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			3. To participate in, and provide data for, a workshop to review the findings relative to other available data and results. 4. To update the level-2 seabird risk assessment using all new information on bird population size, productivity, and distribution, and all relevant fishing effort and observer data for the 2010/11 to 2014/15 fishing years. 5. To identify key drivers of uncertainty and opportunities to reduce uncertainty in the risk ratios for species at high or very high risk.		
PRO	PRO2014-03	Research in response to advice from the Māui's dolphin research advisory group	1. To be developed through the MRAG process: agreed project was genetic mark recapture estimation of Māui dolphin abundance with field effort in 2014/15 and 2015/16.	Complete	
PRO	PRO2014-02	Risk assessment modelling for fishing-related mortality of sea lions to underpin the TMP	1. To review existing models of New Zealand sea lions that have been used to estimate key demographic rates and their variability. 2. Based on the results of Objective 1, develop an operating model of the Auckland Island population of New Zealand sea lions suitable for use in management strategy evaluation. 3. To use a management strategy evaluation to assess the risk posed by commercial fishing to New Zealand sea lions, including assessing the likely performance of candidate management approaches against current or agreed performance criteria. 4. To extend the modelling to other populations and risks as information permits.	Complete	Roberts et al. 2016
PRO	PRO2014-05 Co-funded with DOC POP2015-01	Reducing uncertainty in biological components of the risk assessments for at-risk seabird species	1. Species, population, and information requirements to be determined based on the prioritisation procedures in the NPOA-seabirds and the table of priorities from the outputs of the review workshop.	Complete	Bell et al. 2016
PRO	PRO2014-01	Improving information on the distribution of key protected species	1. To produce an agreed list of seabird and marine mammal species for inclusion and compile all available spatial data for these species. 2. To model and map the distribution of the species identified in objective 1 from available spatial data, reflecting any temporal changes (seasonality or trends). 3. To refine the results of the mapping for priority species by developing and implementing predictive habitat distribution models.	Ongoing	
PRO	SEA2014-12	NZ sea lion stable isotope analysis	1. Locate the ideal NZ sea lion teeth for stable isotope analysis from that will provide the best temporal coverage. 2. Prepare and micro-drill the NZ sea lion teeth within the annual bands to document changes in their foraging and changes in ocean conditions through time using stable isotope analysis. 3. Assess the stable isotope datasets in combination with existing diet studies, prey abundance estimates and climate indices to best examine temporal patterns.	Ongoing	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	SEA2014–15	Sensitivity of the Seabird Risk Assessment to selected scenarios	1. Assess the sensitivity of the Seabird Risk Assessment to assumptions about Buller's albatrosses. 2. Assess the ability of the Seabird Risk Assessment to detect changes in the capture rates. 3. Assess the sensitivity of the risk assessment to live captures.	Complete	Abraham & Richard 2017
PRO	SEA2014-16	Observer coverage power analysis for NPOA	1. Assess the level of observer coverage required to detect a change in the estimated risk of fisheries to New Zealand seabirds, for varying levels of decrease in fishing-related fatalities, for selected seabird species and fisheries.	Complete	
PRO	SEA2014-19	Development and production of smaller hook pods for trial in NZ	To modify the current Hook Pod to New Zealand version without the LED incorporated. This will result in a smaller and more robust Hook Pod that will be equally effective at reducing seabird bycatch in New Zealand's surface longline fisheries.	Complete	
PRO	SEA2014-21	Additional analyses to support the New Zealand sea lion risk assessment	1. The effect of past mortality resulting from key threats for which data are available (such as disease and fishing mortality), or for which plausible estimates are available (such as cryptic mortality), will be explored by fitting the historical demographic model including data on mortality arising from known threats to estimate starting (1960) and current population structure. Threat-derived mortality will then be excluded from the model and re-run from the estimated starting population to predict population structure in the absence of such mortality. 2. Questions were raised about the most appropriate way to deal with animals of unknown pupping status in the model. At present, decision rules are used to determine pupping status from observations (observed suckling, at least 3 sightings with a pup or 3 sightings without a pup) to determine pupping status, with the remaining animals classified as unknown and divided in the proportion of known pupping / non-pupping. Exclusion of animals of unknown status results in increased estimates of pupping rate. Alternative approaches should be considered and the sensitivity of pupping rate to relaxing the decision rules should be explored, such as relaxing the decision rules used to determine pupping status to 2 or 1 observations with or without a pup, or use of other information such as females calling to pups. (Linked to the following item). 3. Similar questions were raised about determining pupping status before an animal that has moved between colonies is used to estimate migration (translocation) rates. As an alternative, this requirements could be relaxed to include animals simply observed (but not confirmed to be pupping) at another colony to be included in migration rate estimation. 4. The assumption of a CV of 0.06 for pup census indices, as the only way of specifying a relative weighting between census and tag-recapture data, was questioned. Alternative CVs and weighting approaches should be determined using something like standard deviations of Pearson residuals. 5. Incorporation of time-varying re-sighting probability was noted to improve model fits, indicating that that re-sighting probabilities did vary over time. One could explore whether the number of days on which re-sightings were conducted each year are correlated with effort days, in which case effort days could be used to estimate re-sight probabilities for recent years that have not been back-corrected.	Complete PRO2014-02.	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			6. It was recommended that the effect of incorporation of 'phantom tags' on parameters such as re-sighting probability should be explored. An alternative approach would be to simply multiply the survival rate from tagging to age 1 yr by the directly estimated proportion of pups that die prior to tagging. The latter is, after all, the basis for how many phantom tags are added.		
PRO	SEA2014-23	An assessment of thermal aerial survey techniques on fur seals	1. Undertake field work component. 2. Submit draft report and present to the Aquatic Environment Working Group.	Complete	
PRO	SEA2014-25	Black petrel distribution and fisheries overlap	1. Geographical Information Systems (GIS) data, giving the distribution of black petrel. 2. GIS data giving the overlap of black petrel with bottom longline, surface longline, set net, and trawl fisheries.	Complete	
PRO	SEA2013-06	Black Petrel Distribution Modelling	1. To use the best available information to develop a spatial and seasonal distribution of black petrel, in New Zealand waters.	Complete	Abraham et al. 2015
PRO	SEA2013–14	Re-Run of Level-2 Seabird Risk Assessment 2014	1. To provide an update of the Seabird Risk Assessment, including observer and fisheries data to the end of the 2012/13 fishing year.	Complete	Richard & Abraham 2015
PRO	SEA2013-08	Data preparation for protected species bycatch estimation	1. Groom catch effort, observer, and protected species capture data. 2. Provide web-based interface to allow exploration, display, and reporting on the data.	Completed: preparation for PRO2013-01	
PRO	PRO2013-01	Protected species capture estimation	1. To estimate capture rates and total captures of seabirds, marine mammals, turtles, and protected fish species by method, area, and target fishery, and where possible, by species for the fishing years 2012/13, 2013/14 and 2014/15. 2. To estimate factors associated with the capture of seabirds and marine mammals. 3. To estimate, where possible, the nature and rate of warp strike incidents and total number of seabirds affected.	In progress	Abraham et al. 2016, 2017
PRO	PRO2013-06	Abundance and distribution of WCSI Hector's dolphins	1. To develop and refine designs and methods for summer and winter aerial surveys for Hector's dolphins along the WCSI consistent with the recent ECSI surveys. 2. To estimate the abundance of Hector's dolphins along the WCSI in summer 2013/14 applying an agreed aerial survey methodology. 3. To estimate the distribution of Hector's dolphins along the WCSI in summer 2013/14 applying an agreed aerial survey methodology. 4. To estimate the abundance of Hector's dolphins along the WCSI in winter 2014 applying an agreed aerial survey methodology.	Complete	Clement & MacKenzie 2016

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			5. To estimate the distribution of Hector's dolphins along the WCSI in winter 2014 applying an agreed aerial survey methodology.		
PRO	PRO2013-08	Reanalysis of Hector's dolphin line transect aerial survey data	1. To collate sightings and effort data for all Hector's dolphin aerial surveys that applied different approaches to estimating the detection function. 2. To assess the impact of different approaches to estimating the detection function on estimates of abundance and distribution and develop correction factors. 3. To reanalyse all relevant survey data to estimate Hector's dolphin abundance and distribution applying the agreed approach to estimating the detection function	Included in PRO2013-06	
PRO	PRO2013-13	Global seabird risk assessment (for New Zealand species)	1. Evaluate relative exposure to commercial fisheries at a global scale for New Zealand seabird populations applying a seasonally-disaggregated spatial overlap approach (i.e., accessing global seabird spatio-temporal distribution data and compiling comprehensive global fisheries effort databases) for different categories of fishing effort. 2. Apply estimates of population PBR (from the updated NZ-EEZ seabird risk assessment, including uncertainty) and species- or guild-specific estimates of seabird Vulnerability (i.e., as estimated in the updated NZ-EEZ seabird risk assessment, modified to the extent possible by data indicative of relative seabird bycatch rates in comparable fishing effort inside vs. outside the New Zealand EEZ, including uncertainty) to estimate global fisheries risk for New Zealand seabird populations. 3. For each New Zealand seabird population estimate what proportion of global fisheries risk is attributable to mortalities occurring inside vs. outside the NZ-EEZ, and what proportion is likely to be unaccounted for in the analysis (e.g., due to incomplete global fisheries data or risk from IUU fishing). 4. For that portion of species risk outside the NZ-EEZ, summarise the source of that risk to the extent possible, for example by RFMO (or other relevant management agency), and by fishery group, geographic area, season, vessel size, and other relevant categories.	Contracted, ongoing	Abraham et al. 2017 (CCBST ERS paper)
PRO	PRO2013-17	Repeat quantitative modelling of southern Buller's albatross	1. To update the fully quantitative population model of southern Buller's albatross to assess population trend and key demographic rates for this population. 2. To use the model to predict future trends assuming recent average demographic rates.	Complete	Fu and Sagar 2016
PRO	PRO2013-18	Authoritative Sea Lion Capture List	To produce a definitive data set of New Zealand sea lion captures and to reconcile data from the different sources, and resolve any discrepancies.	Complete	Thompson et al. 2015
PRO	SEA2013-08	Data preparation for protected species bycatch estimation	1. Groom catch effort, observer, and protected species capture data 2. Provide web-based interface to allow exploration, display, and reporting on the data	Completed: preparation for PRO2013-01	
PRO	No project number	A risk assessment of threats to Māui's dolphins	To evaluate of the risks posed to Māui's dolphin to support the review of the TMP.	Complete	Currey et al. 2012

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	PRO2012-02	Assessment of the risk to marine mammal populations from New Zealand commercial fisheries	1. To scope the risk assessment, including producing an agreed list of marine mammal populations (in concert with MAF and DOC). 2. To review the literature, compile the required information and evaluate the appropriate level of risk assessment for the marine mammal populations identified in objective 1. 3. To conduct a risk assessment for the marine mammal populations identified in objective 1 using, where possible, a risk index reflecting the ratio of fisheries-related mortality to the level of potential biological removal. 4. To refine the results of the risk assessment for priority marine mammal populations by incorporating spatially and temporally-explicit abundance, distribution and capture information.	Complete	Berkenbusch et al. 2013; Abraham et al. 2017
PRO	PRO2012-07	Cryptic mortality of seabirds in trawl and longline fisheries	1. To review available information from international literature and unpublished sources to characterise and inform estimation of cryptic mortality and live releases for at-risk seabirds in New Zealand trawl and longline fisheries 2. To review the extent to which fisheries observer data informing current estimates of seabird captures may be used to also estimate cryptic mortalities in different fishery groups in the seabird risk assessment, and identify key assumptions and associated uncertainty in the estimation of cryptic mortalities. 3. To identify those species and/or fishery groups for which current uncertainty regarding cryptic mortality contributes most strongly to high risk scores for at-risk seabird species, and recommend options to improve estimation of cryptic mortality for those species / fishery group combinations.	Complete	Pierre et al. 2015
PRO	PRO2012-10	Level 3 risk assessment for Antipodean albatross	1. Develop an Antipodean albatross population model. 2. Assess the effect of fisheries mortality on population viability. 3. As information permits, assess the effect of alternative management strategies.	Complete	Edwards et al. 2017
PRO	ENV2011-01	NPOA-sharks science review	1. To collate and summarise information in support of a review of the National Plan of Action for the Conservation and Management of Sharks (NPOA-sharks). 2. To identify research gaps from objective 1 and suggest cost-effective ways these could be addressed.	Complete	Francis & Lyon 2012, 2013
PRO	SEA2011-14	CCSBT Seabird risk assessment	To undertake an Ecological Risk Assessment for seabird interactions in surface longline fisheries managed under the Convention for the Conservation of Southern Bluefin Tuna.	Complete	Waugh et al. 2012
PRO	SRP2011-03	Probabilistic modelling of sea lion interactions	1. Estimate the probability that a sea lion suffers mild head trauma following a collision with a SLED grid.	Complete	Abraham 2011
PRO	SRP2011-04	HSL Modelling	1. Revise Breen-Fu-Gilbert sea lion model.	Complete	Breen et al. 2010
PRO	PRO2010-01	Estimating the nature and extent of incidental captures of seabirds, marine mammals and turtles	1. To estimate the nature and extent of captures of seabirds, marine mammals and turtles, and the warp strikes of seabirds in New Zealand fisheries for the fishing years 2009/10, 2010/11 and 2011/12.	Complete	Thompson et al. 2012, 2013, 2014; Richard & Abraham 2015

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		in New Zealand commercial fisheries			
PRO	PRO2010-02	Research into key areas of uncertainty or development of mitigation techniques for the revised NPOA-seabirds	1. To provide the information necessary to underpin the revised NPOA-seabirds or develop mitigation techniques to reduce risk identified via the revised NPOA-seabirds.	Complete	Richard & Abraham 2013a, 2013b, 2013c; Berkenbusch et al. 2013
PRO	No project number	A risk assessment framework for incidental seabird mortality associated with New Zealand fishing in the New Zealand EEZ	To describe the conceptual and methodological framework of this risk assessment approach to guide the completion of similar risk assessments elsewhere.	Complete	Sharp et al. 2011
PRO	SRP2010-03	Fur Seal interactions with a SED excluder device	1. Fur seal interactions with SED excluder device (Dr J Lyle).	Complete	Lyle 2011
PRO	SRP2010-05	Fur seal interaction with an SLED excluder device	1. Using a series of 10-15 impact tests at a maximum collision speed of 5 or 6 ms-1, develop a 'HIC map' for the SLED grid to enable the consequences of collisions with different parts of the grid by sea lions of different head masses to be predicted (scaling values (for eq 3) will include -1/3, -2/3, and -3/4). 2. Using a small number of collision tests, verify that the HIC for a glancing blow can be predicted with sufficient accuracy by resolving vectors. 3. Calculate the maximum possible sensitivity to different boundary conditions using the relative masses of the SLED grid and sea lion heads. 4. Clarify in the final research report that undertaking tests in air (as opposed to underwater) should not affect the results.	Complete	Ponte et al. 2011
PRO	IPA2009-09	Sea Lion bioenergetics modelling	1. To review and collate data on growth, metabolism, diet and reproductive parameters of NZ sea lions or, if data are inexistent, of other sea lions species. 2. To analyse the energy density of various NZ sea lion prey items. 3. To incorporate the data acquired in Objectives 1. and 2. into a bioenergetics model to estimate the energy and food requirements of NZ sea lions.	Complete	Meynier 2010
PRO	IPA2009-16	Preliminary impact assessment of NZ sea lion interaction with SLEDS	1. Preliminary impact assessment of New Zealand sea lion interactions with SLEDs.	Complete	Ponte et al. 2010

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	IPA2009-19/20	Level 2 seabird risk assessment rerun	1. To examine the risk of incidental mortality from commercial fishing for 64 seabird species in New Zealand trawl and longline fisheries.	Complete	Richard et al. 2011
PRO	No project number	External review of NZ sea lion bycatch necropsy data and methods	The primary purposes of this review were to determine whether, in the opinion of a group of independent experts: - the interpretation of necropsy findings and trauma classification system used by Dr Wendi Roe are valid - sea lions recovered from trawl nets have sustained clinically significant trauma - some or all of the sea lions exiting through SLEDs are likely to survive.	Complete	Roe 2010a
PRO	PRO2009-01A	Abundance & distribution of Hector's & Māui's dolphins (5 year project)	1. To estimate the distribution of the South Coast South Island Hector's dolphin sub-population in both winter and summer. 2. The work for this sub-project was subsequently extended to include data collection necessary to estimate abundance.	Complete	Clement & Mattlin 2010
PRO	PRO2009-01B	Abundance, distribution, and productivity of Hector's (and Māui's) dolphins	1. To estimate the likely precision of abundance estimates from summer aerial surveys for Hector's dolphins along the East Coast South Island (ECSI; from Farewell Spit to Nugget Point) under different levels of sampling intensity and stratification. 2. To estimate the likely precision of abundance estimates and the likely quality of distribution information from winter aerial surveys for Hector's dolphins along the ECSI under different levels of sampling intensity and stratification. 3. To identify and quantify trade-offs between the precision of abundance estimates and the quality of distribution information as well as between overall precision and likely cost (e.g., based on the number of flying hours required). 4. To identify key areas and times for which it would be particularly useful to have information on Hector's dolphin distribution (e.g., where risk may come from overlap with particular fisheries) and quantify trade-offs between the precision of ECSI-wide surveys and collecting such fine-scale information. 5. Assess the extent to which two-phase or adaptive approaches would be useful to improve the surveys' utility for assessing dolphin distribution, particularly the seaward limit.	Complete	MacKenzie et al. 2013
PRO	PRO2009-01C	Abundance, distribution and productivity of Hector's (and Māui) dolphins	1. To estimate critical aspects of the biology, abundance and distribution of Hector's and Māui's dolphin populations to assess the effects of fishing-related mortality on these populations including the abundance of Hector's dolphins along the ECSI in summer 2012/13 applying an agreed aerial survey methodology. 2. To estimate critical aspects of the biology, abundance and distribution of Hector's and Māui's dolphin populations to assess the effects of fishing-related mortality on these populations including the distribution of Hector's dolphins along the ECSI in summer 2012/13 applying an agreed aerial survey methodology.	Complete	MacKenzie & Clement 2014

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			3. To estimate critical aspects of the biology, abundance and distribution of Hector's and Māui's dolphin populations to assess the effects of fishing-related mortality on these populations including the abundance of Hector's dolphins along the ECSI in winter 2013 applying an agreed aerial survey methodology. 4. To estimate critical aspects of the biology, abundance and distribution of Hector's and Māui's dolphin populations to assess the effects of fishing-related mortality on these populations including the distribution of Hector's dolphins along the ECSI in winter 2013 applying an agreed aerial survey methodology.		
PRO	PRO2009-04	Development and efficacy of seabird mitigation measures	1. To test the efficacy of a variety of configurations of mitigation techniques at reducing seabird mortality (or appropriate proxies for mortality) in longline fisheries.	Complete	No reports specified as required output
PRO	ENV2008-03	Bycatch of basking sharks in New Zealand fisheries	1. To review the productivity of basking sharks. 2. To describe the nature and extent of fishery-induced mortality of basking sharks in New Zealand waters and recommend methods of reducing the overall catch.	Complete	Francis & Smith 2010
PRO	PRO2008-01	Risk assessment of protected species bycatch in NZ fisheries	1. To provide an assessment of the risk posed by different fisheries to the viability of New Zealand protected species, and to assign a risk category to all New Zealand fishing operations.	Complete	Waugh et al. 2009
PRO	PRO2008-03	Necropsy of marine mammals captured in New Zealand	1. To necropsy marine mammals captured incidentally to New Zealand fishing operations in the SQU6T fishery during the 2008/09 fishing year to determine life-history characteristics such as sex- reproductive status and the likely cause of mortality- and to determine the species- and sex of captured animals returned for necropsy. 2. To determine- through examination of returned carcasses- the species- sex- reproductive status- and age-class of sea lions and fur seals captured in the SQU6T New Zealand fishery. 3. To detail any injuries and- where possible- the cause of mortality of sea lions and fur seals returned from New Zealand fisheries- and examine relationships between injuries and body condition- breeding status- and other associated demographic characteristics. 4. To review and collate data from previous NZ sea lion autopsy programmes.	Complete	Roe 2010b; Roe & Meynier 2012
PRO	SAP2008-14	Sea lion population modelling, additional	1. To assess the likely performance of different bycatch control rules for the SQU6T fishery. 2. To correct and update the Breen-Fu-Gilbert (2008) sea lion model- including assessment of the performance of 200-series and 300-series management control rules. 3. To document the development of the model- including all four objectives of project IPA2006/09 and objective 1 of this project- in a single report suitable for an international review.	Complete	Breen et al. 2010
PRO	Deepwater Group	Necropsy of marine mammals captured in New Zealand fisheries in the 2007–08 fishing year	Necropsy of marine mammals captured in New Zealand fisheries in the 2007–08 fishing year.	Complete	Roe 2009a
PRO	IPA2007-09	Protected species risk assessment	To provide an assessment of the risk posed by different fisheries to the viability of NZ protected species- and to assign a risk category to all NZ fishing operations.	Complete	Waugh et al. 2008

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	PRO2007-01	Estimating the nature and extent of incidental captures of seabirds in New Zealand commercial fisheries	1. Estimate capture rates per unit effort and total captures of seabirds for the New Zealand EEZ and in selected fisheries by method, area, target fishery, in relation to mitigation methods in use, and, where possible, by seabird species for the fishing year 2006/07, 2007/08 and 2008/09. 2. Examine the incidence of seabird warp strike in trawl fisheries where these data are available from fisheries observers, and estimate the rate of incidents (birds affected per hour) and total number of seabirds affected by fishery, area and method. Examine the factors (fishery, environmental, seasonal, mitigation, area) that influence the probability of warp-strike occurring.	Complete	Abraham 2010; Abraham & Thompson 2009a, 2010, 2011a, 2011b; Thompson & Abraham 2009a; Abraham et al. 2010b
PRO	PRO2007-02	Estimating the nature and extent of incidental captures of seabirds in New Zealand commercial fisheries	1. Estimate capture rates per unit effort and total captures of seabirds for the New Zealand EEZ and in selected fisheries by method, area, target fishery, in relation to mitigation methods in use, and, where possible, by seabird species for the fishing year 2006/07, 2007/08 and 2008/09. 2. Examine the incidence of seabird warp strike in trawl fisheries where these data are available from fisheries observers, and estimate the rate of incidents (birds affected per hour) and total number of seabirds affected by fishery, area and method. Examine the factors (fishery, environmental, seasonal, mitigation, area) that influence the probability of warp-strike occurring.	Complete	Abraham et al. 2010a; Thompson & Abraham 2009a, 2009b, 2009c, 2010, 2011; Thompson et al. 2010a, 2010b
PRO	ENV2006-05	The use of electronic monitoring technology in New Zealand longline fisheries	1. Trial the deployment of electronic monitoring systems in selected longline fisheries, monitoring incidental take of protected species. 2. Evaluate the efficacy of electronic monitoring in allowing enumeration and identification of protected species captures. 3. Recommend options for data management and information transfer arising from the deployment of electronic monitoring in selected fisheries.	Complete	McElderry et al. 2008
PRO	IPA2006-02	The efficacy of warp strike mitigation devices: trials in the 2006 squid fishery	1. Groom the mitigation trial data and produce a summary of the data. 2. Examine strike rates and capture rates on warps and mitigation devices. 3. Determine the relative efficacy of mitigation devices tested in the trial. 4. Make recommendations regarding future trials. 5. Compare seabird warp strike data for 2005 and 2006. 6. Work with SeaFIC and the mitigation trials TAG to produce analyses and outputs.	Complete	Middleton & Abraham 2007
PRO	IPA2006-09	Modelling interactions between trawl fisheries and New Zealand Sea lion interactions	1. Model the New Zealand sea lion population and explore alternative management procedures for controlling New Zealand sea lion bycatch in the SQU 6T fishery. 2. Collate and review all available sea lion biological data- fisheries data- and sea lion bycatch data relevant to a population model and management strategy evaluation for the Auckland Islands sea lion population. 3. Update and improve the existing Breen and Kim sea lion population model (2003) to incorporate all relevant data and address model uncertainties including but not necessarily limited to those identified by the AEWG. 4. Fit the revised model to all available data and test sensitivity including but not necessarily limited to runs identified by the AEWG. 5. Test a range of management procedures (rules) with the model to determine if they meet agreed management criteria.	Complete	Breen 2008

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	IPA2006-13	Identification of Marine Mammals Captured in New Zealand Fisheries	1. To determine, through examination of returned marine mammal carcasses, the species, sex, reproductive status, and age-class of marine mammals returned from New Zealand fisheries. 2. To detail any injuries and, where possible, the cause of mortality of marine mammals returned from New Zealand fisheries, and examine relationships between injuries and body condition, breeding status, and other associated demographic characteristics.	Complete	Roe 2009b
PRO	PRO2006-01	Data collection of demographic, distributional and trophic information on selected seabird species to allow estimation of effects of fishing on population viability	1 To gather demographic, distributional and dietary information on selected seabird species to allow assessment of effects of fishing on population viability.	Complete	Sagar & Thompson 2008; Sagar et al. 2009a, 2009b, 2010a, 2010b, 2010c; Baker et al. 2008, 2009, 2010
PRO	PRO2006-02	Modelling of the effects of fishing on the population viability of selected seabirds	1. Model the effects of fisheries mortalities on population viability compared with other sources of mortality or trophic effects of fishing. 2. Examine the overlap of fishing activity with species distribution at sea for different stages of the breeding and life-cycle and for different sexes, and assess the likely risk to species or populations from fisheries (by target species fisheries, fishing methods, area and season) in the New Zealand EEZ.	Complete	Francis & Bell 2010; Francis 2012; Francis et al. 2015
PRO	PRO2006-04	Estimation of the nature and extent of incidental captures of seabirds in New Zealand commercial fisheries	1. To estimate the nature and extent of captures and warp-strikes of seabirds in New Zealand fisheries for the fishing year 2005/06.	Complete	Baird & Smith 2008
PRO	PRO2006-05	Estimating the nature and extent of marine mammal captures in New Zealand commercial fisheries	1. To estimate and report the total numbers, releases and deaths of marine mammals where possible by species, fishery and fishing method, caught in commercial fisheries for the years 1990 to the end of the fishing year 2005/06. 2. To analyse factors affecting the probability of fur seal captures for the years 1990 to the end of the fishing year 2005/06. 3. To classify fishing areas, seasons and fishing methods into different risk categories in relation to the probability of marine mammal incidental captures for the years from 1990 through to the end of the fishing year 2005/06.	Complete	Mormede et al. 2008; Baird 2008a, 2008b, 2011; Smith & Baird 2009, 2011
PRO	PRO2006–07	Characterise non-commercial fisheries interactions	1. To characterise non-commercial fisheries interactions with seabirds and marine mammals. 2. Characterise non-commercial fisheries risk to seabirds and marine mammals by area and method.	Complete	Abraham et al. 2010a; Thompson & Abraham 2009a,

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			Recommend mitigation measures appropriate for uptake in non-commercial fisheries in which seabird or marine mammal captures occur.		2009b, 2009c, 2010, 2011; Thompson et al. 2010a, 2010b, 2010c
PRO	ENV2005-01	Estimation of the nature and extent of incidental captures of seabirds in New Zealand fisheries	1. To estimate the nature and extent of captures of seabirds in selected New Zealand fisheries for the fishing year 2004/05.	Complete	Baird & Smith 2007a; Baird & Gilbert 2010
PRO	ENV2005-02	Estimation of the nature and extent of marine mammal captures in New Zealand fisheries	To examine the nature and extent of the captures of marine mammals in New Zealand fisheries, for the whole New Zealand EEZ, by Fishery Management Area and fishing season, and by smaller metric as appropriate for the fishing year 2004/05. 2. Examine alternative methods for estimating sea lion captures and recommend one or more alternative standardised methods for describing and estimating sea lion captures in the SQU 6T fishery.	Complete	Abraham 2008; Baird 2007; Smith & Baird 2007b; Baird & Smith 2007b
PRO	ENV2005-04	Identification of marine mammals captured in New Zealand	1. To determine the species- sex- and where possible- age and reproductive status of marine mammals captured in New Zealand fisheries. 2. To necropsy marine mammals captured incidentally to New Zealand fishing operations to determine life-history characteristics and the likely cause of mortality. 3. To determine- through examination of returned marine mammal carcasses- the taxon to species-level- sex- and reproductive status- and age-class of marine mammals captured in New Zealand fisheries. 4. To detail the injuries and where possible the cause of mortality of marine mammals returned from New Zealand fisheries- along with their body condition and breeding status- and other associated demographic characteristics. 5. To detail the protocol used for the necropsy of marine mammals- to provide a standardised procedure for autopsy to determine species- age- sex and associated demographic characteristics for fishery-killed specimens.	Complete	Roe 2007
PRO	ENV2005–06	Estimation of protected species captures in longline fisheries using electronic monitoring	1. To provide estimates of seabird and marine mammal mortalities from longline fisheries in New Zealand using electronic monitoring systems and to recommend deployment and data management options for ongoing use of these systems for estimation of protected species incidental take.	Complete	McElderry et al. 2007
PRO	ENV2005-09	Data collection to estimate key performance indicators in the	1. To gather data on key population parameters for Chatham albatross *Diomedea eremita*- to enable population viability to be assessed- and the responses of key parameters to fisheries mortality and fisheries management activities to mitigate fisheries related risk. 2. To undertake field research to collect data on population growth rates- adult survival- inter-breeding season	Complete	No reports specified as required output

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		Chatham albatross, *Diomedea eremita*.	survival- mortality due to predation at the colony- fecundity and associated parameters for Chatham Albatross- following the study design project. 3. To undertake field research to determine the range and extent foraging movements of Chatham albatrosses within New Zealand fishing waters- and examine the nature and extent of any association between Chatham albatrosses and fishing activities.		
PRO	ENV2005-13	Assessment of risk to yellow-eyed penguin Megady-ptes antipodes from fisheries incidental mortality	1. To review existing data on yellow-eyed penguin M. antipodes population performance and fisheries information and provide an analysis of the potential effect of fishing mortality and other factors on population viability. 2. To recommend data collection requirements and protocols for the assessment of the effects of fishing on yellow-eyed penguins.	Complete	Maunder 2007
PRO	ENV2004-02	Estimation of New Zealand sea lion incidental captures in New Zealand Fisheries	1. To estimate the level of New Zealand sea lion (*Phocartos hookeri*) incidental capture in New Zealand fisheries	Complete	Smith & Baird 2007a
PRO	ENV2004-04	Characterisation of seabird captures in New Zealand fisheries	1. Characterisation of seabird captures in New Zealand fisheries.	Complete	MacKenzie & Fletcher 2006
PRO	ENV2004–05	Modelling of impacts of fishing-related mortality on New Zealand seabird populations	1. To examine and identify modelling approaches to analyse seabird demographic impacts that may be occurring as a result of fisheries mortality. 2. To compile databases of available demographic and distributional data on selected seabirds affected by fisheries mortality and New Zealand fisheries and estimate key population parameters and seasonal distribution for each species. 3. To estimate rates of removals related to fishing activities in New Zealand for selected seabird species, where possible by age class and sex. 4. To describe the spatial overlap of seabird distributions at sea, with fisheries where the risk of incidental mortality has been demonstrated to be moderate to high. 5. To examine the potential for factors other than fisheries removals within the New Zealand zone to influence the population dynamics of the selected study species. 6. To characterise selected seabird populations' abilities to sustain removals related to fishing operations within the New Zealand EEZ, and to recommend, where possible environmental standards for assessing the sustainability of selected fishing operations in relation to impacts on seabird populations.	Complete	Fletcher et al. 2008
PRO	ENV2004-06	Māui's dolphin study	1. To quantify and compare summer and winter distribution of Māui's dolphin.	Complete	Slooten et al. 2005
PRO	IPA2004-14	Seabird warp strike in the southern squid trawl fishery	1. To document seabird warp strike in the southern squid trawl fishery, 2004–05.	Complete	Abraham & Kennedy 2008

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	ENV2003-05	Review of the Current Threat Status of Associated or Dependent Species	1. To assess the current threat status of selected associated or dependent species.	Complete	Baird et al. 2010
PRO	No project number	QMA SQU6T New Zealand sea lion incidental catch and necropsy data for the fishing years 2000–01, 2001–02 and 2002–03	Report on New Zealand sea lion incidental catch and necropsy data for the fishing years 2000–01, 2001–02 and 2002–03	Complete	Mattlin 2004
PRO	MOF2002–03L	Exploring alternative management procedures for controlling bycatch of Hooker's sea lions in the SQU 6T squid fishery	Report on exploring alternative management procedures for controlling bycatch of Hooker's sea lions in the SQU 6T squid fishery.	Complete	Breen & Kim 2006
PRO	ENV2001-01	Estimation of seabird incidental captures in New Zealand fisheries	1. To estimate the level of seabird incidental capture in New Zealand fisheries. 2. To recommend appropriate levels of observer coverage for estimation of seabird incidental capture in New Zealand fisheries.	Complete	Baird 2004a, 2004b, 2004c; Smith & Baird 2008b
PRO	ENV2001–02	Incidental capture of *Phocarctos hookeri* (New Zealand sea lions) in New Zealand commercial fisheries, 2001–02.	1. To estimate and report the total numbers of captures, releases, and deaths of *Phocarctos hookeri* caught in fishing operations, including separate estimates for SQU 6T and other areas, as appropriate, during the 2001102 fishing year, including confidence limits and an investigation of any statistical bias in the estimate.	Complete	Baird 2005a, 2005b; Baird & Doonan 2005
PRO	ENV2001-03	Estimation of *Arctocephalus forsteri* (New Zealand fur seal) incidental captures in New Zealand fisheries	1. To estimate the level of *Arctocephalus forsteri* incidental capture in New Zealand fisheries. 2. To recommend appropriate levels of observer coverage for estimation of *Arctocephalus forsteri* incidental capture in New Zealand fisheries.	Complete	Smith & Baird 2008a; Baird 2005c, 2005d, 2005e
PRO	ENV2000–01	Protected species bycatch	1. To estimate the total numbers of captures, releases, and deaths of seabirds and marine mammals – by species – caught in fishing operations during the 1999–2000 fishing year.	Complete	Baird 2003

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	ENV2000-02	Estimation of incidental mortality of New Zealand sea lions in New Zealand fisheries	1. To examine the factors that may influence the level of incidental mortality of New Zealand sea lion in New Zealand fisheries. 2. To recommend appropriate levels of observer coverage for estimation of incidental mortality of New Zealand sea lion in New Zealand sea lion fisheries.	Complete	Doonan 2001; Bradford 2002; Smith & Baird 2005a, 2005b
PRO	ENV2000-03	ENV 2000-A Estimation of seabird and marine mammal incidental capture in New Zealand fisheries	1. To estimate the level of seabird and marine mammal incidental capture in New Zealand fisheries. 2. To determine the factors that influence the level of seabird and marine mammal incidental capture in New Zealand fisheries. 3. To recommend appropriate levels of observer coverage for estimation of seabird and marine mammal incidental capture in New Zealand fisheries.	Complete	Bradford 2002, 2003; Francis et al. 2004
PRO	ENV99-01	Incidental capture of seabirds, marine mammals and sealions in commercial fisheries in New Zealand waters	To estimate the level of seabird and marine mammal incidental captures in New Zealand fisheries.	Complete	Baird 2001; Doonan 2000
PRO	No project number	Factors influencing bycatch of protected species	To determine the factors that influence the level of seabird and marine mammal incidental capture in New Zealand fisheries.	Complete	Baird & Bradford 2000a, 2000b
PRO	ENV98-01	Estimation of non-fish bycatch in commercial fisheries in New Zealand waters, 1997–98	To estimate the level of non-fish bycatch in New Zealand fisheries.	Complete	Baird 1999b; Baird & Bradford 1999
PRO	No project number	Annual review of bycatch in southern bluefin and related tuna longline fisheries in the New Zealand 200 n. mile Exclusive Economic Zone	Review bycatch in New Zealand's southern bluefin and related tuna longline fisheries.	Complete	Baird et al. 1998
PRO	SANF01	Report on the incidental capture of nonfish species during	To report on incidental captures of non-fish species in New Zealand fisheries.	Complete	Baird 1997

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		fishing operations in New Zealand waters			
PRO	No project number	Non-fish Species and Fisheries Interactions	To estimate the level of non-fish bycatch in New Zealand fisheries.	Complete	Baird 1996
PRO	No project number	Analyses of factors which influence seabird bycatch in the Japanese southern bluefin tuna longline fishery in New Zealand waters, 1989-93	1. To assess the influence that 15 monitored environmental and fishery related factors had on seabird bycatch rates, and to gauge the effectiveness of various mitigation measures.	Complete	Duckworth 1995
PRO	No project number	Incidental catch of Hooker's sea lion in the southern trawl fishery for squid, summer 1994	Report on the incidental catch of Hooker's sea lion in the souther trawl fishery for squid, summer 1994.	Complete	Doonan 1995
PRO	No project number	Nonfish Species and Fisheries Interactions	To estimate the level of non-fish bycatch in New Zealand fisheries.	Complete	Baird 1995
PRO	No project number	Nonfish Species and Fisheries Interactions	To estimate the level of non-fish bycatch in New Zealand fisheries.	Complete	Baird 1994
PRO	No project number	Incidental catch of fur seals in the west coast South Island hoki trawl fishery, 1989-92	To report on incidental captures of fur seals in the west coast South Island hoki trawl fishery 1989–92.	Complete	Mattlin 1993
PRO	No project number	Incidental catch of non-fish species by setnets in New Zealand waters	To report on incidental captures of non-fish species in New Zealand setnet fisheries.	Complete	Taylor 1992
PRO	No project number	Seabird bycatch by Southern Fishery longline vessels in New Zealand waters	1. To describe the tuna longline fishery in the New Zealand EEZ and how seabirds are caught by longline vessels. 2. To summarise information available on seabird population trends, and estimates the scale of the incidental capture of seabirds in the larger of two tuna longline fisheries in the EEZ. 3. To describe measures which could reduce the number of seabirds caught by tuna longlines.	Complete	Murray et al. 1992

Theme	Project code	Project title	Specific objectives	Status	Citation/s
NPB	DAE2017-01	Bycatch monitoring and quantification in deepwater fisheries (HOK/HAK/LIN)	1. To estimate the catch composition in target trawl fisheries for hoki, hake and ling. This should include the quantity of non-target fish species caught, and the target and non-target fish species discarded, using data from MPI Observers to the end of the most recent complete fishing year in a format that meets management needs. 2. To compare estimated rates, amounts, and trends of bycatch and discards over time in the hoki, hake, and ling trawl fisheries. 3. To update any relevant sections of the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.		
NPB	DAE2016-01	Total catch composition in deepwater fisheries (squid & scampi)	1. To estimate the catch composition in the target fisheries for squid and scampi. This should include the quantity of non-target fish species caught, and the target and non-target fish species discarded, using data from MPI Observers and commercial fishing returns to the end of the most recent complete fishing year in a format that meets management needs. 2. To compare estimated rates, amounts, and trends of bycatch and discards from this study with previous projects on bycatch in the squid and scampi fisheries. 3. To update any relevant sections of the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.	Contracted, in progress	
NPB	SEA2016-19	Spatial methods for development to support risk assessment (part II). Estimation of capture and retention efficiency for non-target fish species in commercial trawl fisheries	1. To implement and test a spatially-explicit two-part delta-gamma statistical model (e.g., Thorson et al. 2015) for estimating species density and capture and retention efficiency in the commercial fishing gear. 2. To estimate relative densities and fishery groups catchability (with uncertainty) in a number of non-target fish species in Chatham Rise trawl fisheries. 3. To perform a simulation self-test of the model. 4. To submit the results for publication in the primary literature.	Contracted, in progress	
NPB	ENV2015-01	Updating tools for at-sea fish identification	1. To review the level of information required by the seafood sector and other users of fish identification guides in New Zealand. 2. To evaluate the most beneficial and cost-effective methods of delivery that are practicable and consistent with MPI policy directions. 3. To review, revise and produce the appropriate information tools on fish identification.	Ongoing	
NPB	ENV2015-03	Addressing key information gaps identified by the shark	1. To collect and analyse biological information to improve estimates of risk for inshore and deepwater shark species identified as being at relatively high risk.	Complete	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		qualitative risk assessment			
NPB	No Project Code	Qualitative Shark risk assessment	To produce a qualitative risk assessment for all shark species possible within the New Zealand EEZ.	Complete	Ford et al. 2015
NPB	ENV2014-02	NPOA-sharks: age and growth of selected at-risk species	1. To estimate basic biological parameters for high risk, high uncertainty chondrichthyans.	Ongoing	
NPB	No project code	Mitigation options for shark bycatch in longline fisheries	Conduct a literature review and assess the options for improvements in the practice of fisheries to mitigate shark bycatch.	Complete	Howard 2015
NPB	SEA2013-16	Data collation for shark risk assessments	1. To assemble and collate all available information on the distribution and intensity of all fishing methods for the most recent five full fishing years that potential cause fishing-related mortality of chondrichthyans. 2. To assemble and collate all available information on the distribution, abundance, demographics and productivity of all New Zealand chondrichthyans.	Complete	Francis 2015
NPB	ENV2013-01	Development of model-based estimates of fish bycatch	1. To develop a statistical modelling approach to estimating total captures of fish and invertebrates using observer and catch-effort information from selected fisheries. 2. To compare estimates of total captures, confidence limits, and trends for selected species, species groups, and fisheries made using existing ratio-based methods and statistical models. 3. To estimate, within a simulation framework, the potential for bias in ratio-based and model-based methods, the sizes of confidence limits for estimates from the two approaches in comparable situations, and identify the factors associated with good and poor performance.	Complete	Edwards et al. 2015
NPB	DAE2010-02	Bycatch monitoring & quantification for scampi bottom trawl	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded in the specified fishery, for the fishing years since the last review, using data from Ministry of Fisheries Observers and commercial fishing returns. 2. To compare estimated rates and amounts of bycatch and discards from this study with previous projects on bycatch in the specified fishery. 3. To compare any trends apparent in bycatch rates in the specified fishery with relevant fishery independent trawl surveys. 4. To provide annual estimates of bycatch for nine Tier 1 species fisheries and incorporate into the Aquatic Environment and Biodiversity Report specified in Objective 3 for SQU, SCI, HAK, HOK, JMA, ORH, OEO, LIN, SBW.	Complete	Anderson 2012, 2013a, 2013b, 2014a, 2014b; Ballara 2015; Ballara & O'Driscoll 2015.
NPB	ENV2009-02	Bycatch and discards in oreo and orange roughy trawl fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for oreos for the fishing years 2002/03 to 2008/09 using data from Scientific Observers and commercial fishing returns. 2. To estimate the quantity of non-target fish species caught, and the target and non-target fish species	Complete	Anderson 2011

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			discarded, in the trawl fisheries for orange roughy for the fishing years 2004/05 to 2008/09 using data from Scientific Observers and commercial fishing returns.		
NPB	IDG2009-01	Finfish field identification guide	1. To complement the field identification guide under IDG2006/01 with the remaining 120 fish species caught by commercial fishers in New Zealand waters.	Complete	McMillan et al. 2011a, 2011b, 2011c
NPB	ENV2008-01	Fish and invertebrate bycatch and discards in southern blue whiting fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for southern blue whiting for the fishing years 2002/03 to 2006/07 using data from Scientific Observers and commercial fishing returns.	Complete	Anderson 2009b
NPB	ENV2008-02	Estimation of non-target fish catch and both target and non-target fish discards in hoki, hake and ling trawl fisheries	Estimates of the catch of non-target fish species, and the discards of target and non-target fish species in the hoki (*Macruronus novaezelandiae*), hake (*Merluccius australis*), and ling (*Genypterus blacodes*) trawl fisheries for the fishing years 2003–04 to 2006–07 using data from Scientific Observers and commercial fishing returns.	Complete	Ballara et al. 2010
NPB	ENV2008-04	Productivity of deepwater sharks	1. To determine the growth rate, age at maturity, longevity and natural mortality rate of shovelnose dogfish (*Deania calcea*) and leafscale gulper shark (*Centrophorus squamosus*).	Complete	Parker & Francis 2012
NPB	ENV2007-01 & ENV2007-02	Bycatch and Discards in Squid Trawl Fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for squid for the fishing years 2001/02 to 2005/06 using data from MFish Observers and commercial fishing returns.	Complete	Ballara & Anderson 2009
NPB	ENV2007-03	Productivity and Trends in Rattail Bycatch Species	1. To estimate growth, longevity, rate of natural mortality, and length at maturity of four key rattail bycatch species in New Zealand trawl fisheries. 2. To examine data from trawl surveys and other data sources for trends in catch rates or indices of relative abundance for species in Objective 1.	Complete	Stevens et al. 2010
NPB	DEE2006-03	Monitoring the abundance of deepwater sharks	1. To monitor the abundance of deepwater sharks taken by commercial trawl fisheries.	Complete	Blackwell 2010
NPB	ENV2006-01	Bycatch and discards in ling longline fisheries	To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the longline fisheries for ling for the fishing years 1998/99 to 2005/06 using data from MFish Observers and commercial fishing returns.	Complete	Anderson 2008
NPB	IDG2006-01	Finfish field identification guide	1. To produce a field guide for fish species in New Zealand. 2. To produce a field identification guide for all QMS and other fish species commonly caught in commercial and non-commercial fisheries.	Complete	McMillan et al. 2011a, 2011b, 2011c
NPB	TUN2006-02	Estimation of non-target fish catches in	1. To estimate the catches, catch rates, and discards of non-target fish in tuna longline fisheries data from the Observer Programme and commercial fishing returns for the 2005/06 fishing year.	Complete	Griggs et al. 2008

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		the tuna longline fishery	2. To describe bycatch trends in tuna longline fisheries using data from this project and the results of previous similar projects.		
NPB	ENV2005-17	Estimation of non-target fish catch and both target and non-target fish discards in jack mackerel trawl fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for jack mackerel for the fishing years 20011/2002 to 2004/05 using data from Mfish observers and commercial fishing returns.	Complete	Anderson 2007a
NPB	ENV2005-18	Estimation of non-target fish catch and both target and non-target fish discards in orange roughy trawl fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for orange roughy for the fishing years 1999/2000 to 2003/04 using data from Scientific Observers and commercial fishing returns.	Complete	Anderson 2009a
NPB	TUN2004-01	Estimation of non-target fish catches in the tuna	1. To estimate the catch rates of non-target fish in the 10ngline fisheries for tuna using data from the Observer Programme and commercial fishing returns for the 2002/03, 2003/04 and 2004/05 fishing years. 2. To estimate the quantities of non-target fish caught in the longline fisheries for tuna using data from the Observer Programme and commercial fishing returns for the 2002/03, 2003/04 and 2004/05 fishing years. 3. To estimate the discards of non-target fish caught in the longline fisheries for tuna using data from the Observer Programme and commercial fishing returns for the 2002/03, 2003/04 and 2004/05 fishing years. 4. To describe trends in the non-target fish catches in the tuna longline fisheries using data from this project and the results of previous similar projects.	Complete	Griggs et al. 2007
NPB	ENV2003-01	Estimation of non-target catches in the hoki fishery	1. To estimate the catch rates, quantity and discards of non-target fish catches and the discards of target fish catches in trawl fisheries for hoki, using data from the Observer Programme and commercial fishing returns for the 1999/00 to 2002/03 fishing years. 2. To compare and contrast the estimates from the four years of data in Specific Objective 1 above with the 1990/91 through 1998/99 series previously reported.	Complete	Anderson & Smith 2005
NPB	ENV2002-01	Estimation of non-target fish catch and both target and non-target fish discards for the tuna longline fishery	1. To estimate the catch rates, quantity and discards of non-target fish, particularly oceanic shark species, broadbill swordfish and marlin species, caught in the longline fisheries for tuna, using data from Scientific Observers and commercial fishing returns for the 2000/01 and 2001/02 fishing years.	Complete	Ayers et al. 2004
NPB	ENV2001-04	Non-target fish catch and discards in	To generate estimates of the catch of non-target fish species, and the discards of target and non-target fish species in three important New Zealand trawl fisheries: arrow squid (*Nototodarus sloani & N. gouldi*), jack	Complete	Anderson 2004

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		selected New Zealand fisheries	mackerel (*Trachurus declivis, T. novaezelandiae, & T. symmetricus murphyi*) and scampi (*Metanephrops challengeri*).		
NPB	ENV2001-05	To assess the productivity and relative abundance of deepwater sharks	1. To review the relative abundance, distribution and catch composition of the most commonly caught deepwater shark species: shovelnose dogfish (*Deania catcea*), Baxter's dogfish (*Etmopterus baxten*), Owston's dogfish (*Cenhoscymnus owstoni*), longnosed velvet dogfish (*Centroscymnus crepidater*), leafscale gulper shark (*Cenhophom squamosus*), and the seal shark (*Dalatias ticha*).	Complete	Blackwell & Stevenson 2003
NPB	ENV2001-07	Reducing bycatch in scampi trawl fisheries	1. Collate and review the international literature on methods of reducing bycatch in crustacean trawl fisheries. 2. Review and analyse the data from New Zealand studies. 3. Develop recommendations on future approaches to reducing bycatch in the New Zealand scampi fishery, including some general thoughts on the experimental design of field trials.	Complete	Hartill et al. 2006
NPB	PAT2000–01	Review of rattail and skate bycatch, and analysis of rattail standardised CPUE from the Ross Sea toothfish fishery in Subarea 88.1, from 1997-1998 to 2001–02	Report on review of rattail and skate bycatch, and analysis of rattail standardised CPUE from the Ross Sea toothfish fishery in Subarea 88.1, from 1997-1998 to 2001–02.	Complete	Fenaughty et al. 2003; Marriot et al. 2003
NPB	ENV99-02	Estimation of non-target fish catch and both target and non-target fish discards in selected New Zealand fisheries	1. To estimate the quantity of non-target fish species caught in the trawl fisheries for hoki and orange roughy for the fishing years 1990-91 to 1998-99 using data from Scientific Observers, commercial fishing returns and from research trawl surveys. 2. To estimate the quantity of target and non-target fish species discarded in the trawl fisheries for hoki and orange roughy for the fishing years 1990-91 to 1998-99 using data from Scientific Observers, commercial fishing returns and from research trawl surveys. 3. To explore the effects of various factors on the total catch of non-target fish species and the discards of target and non-target fish species in the trawl fisheries for hoki and orange roughy for the fishing years 1990-91 to 1998-99. 4. To recommend appropriate levels of observer coverage for estimation of non-target fish catch and discards of target and non-target fish species in the hoki and orange roughy fisheries.	Complete	Anderson et al. 2001
NPB	ENV99-05	To identify trends in abundance of associated or dependent species from selected commercial fisheries	To estimate trends in abundance of associated and depeadent species, including invertebrates, from deepwater and middle depth fisheries on the Chatham Rise.	Complete	Livingston et al. 2003

Theme	Project code	Project title	Specific objectives	Status	Citation/s
NPB	ENV98-02	Pelagic shark bycatch in the New Zealand tuna longline fishery	To determine pelagic shark bycatch in the New Zealand tuna longline fishery.	Complete	Francis et al. 2001
NPB	No project number	Fish bycatch in New Zealand tuna longline fisheries	To report on fish bycatch in New Zealand tuna longline fisheries.	Complete	Francis et al. 1999; 2000
NPB	ENV97-01	Estimation of nonfish bycatch in New Zealand fisheries	To estimate non-fish bycatch in New Zealand fisheries.	Complete	Doonan 1998; Baird 1999a; Baird et al. 1999
NPB	SCI97-01	Scampi stock assessment for 1998 and an analysis of the fish and invertebrate bycatch of scampi trawlers	1. To summarise catch, effort, observer, and research information for scampi fisheries in QMAs 1,2,3,4 (east and western portions), and 6A in 1998.	Complete	Cryer et al. 1999
BEN	BEN2017-01	Monitoring of deepwater trawl footprint	1. To help MPI groom data, develop summary statistics, for Tier 1 deepwater fisheries and the aggregate of all Tier 1 and Tier 2 deepwater fisheries, of the extent and frequency of fishing by year, by depth zone, by fishable area, and by predicted BOMEC habitat class, and to identify any trends or changes to meet management needs. 2. To update any relevant sections in the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.	Approved, not yet contracted	
BEN	SEA2016-08	Power Analysis - Benthic Fauna in Spirits Bay	Using previous survey results, conduct a power analysis to estimate the likelihood of a range of survey designs consistent with the monitoring programme from project ENV2005/23 detecting changes in key indicators of the state of the benthic communities in Spirits Bay and Tom Bowling Bay since the last survey.	Complete	
BEN	SEA2016-12	SEA2016-12 GLM Spat composition	Half funding of GLM spat composition study for 90 mile beach (aquaculture unit funding the other half).	Complete	
BEN	DAE2016-05	Monitoring the trawl footprint for deepwater fisheries	1. To estimate the trawl footprint and map the spatial and temporal distribution of trawling on or near the seabed throughout the EEZ between 1989/90 and the most recent completed fishing year. 2. To produce summary statistics, for Tier 1 deepwater fisheries and the aggregate of all Tier 1 and Tier 2 deepwater fisheries, of the extent and frequency of fishing by year, by depth zone, by fishable area, and by predicted BOMEC habitat class, and to identify any trends or changes to meet management needs. 3. To update any relevant sections in the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.	Ongoing	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
BEN	BEN2014-01	Risk assessment for benthic habitats, biodiversity, and production	1. To review the design and implementation of management frameworks, including objectives and targets, to manage the effects of mobile bottom fishing methods on vulnerable benthic taxa and habitats. 2. To complete spatially explicit quantitative impact assessments for benthic taxa and/or habitats affected by bottom fisheries, within spatially distinct or overlapping zones within the New Zealand EEZ, consistent with available databases and the outputs of existing projects. 3. To compile and combine impact assessments from Objective 2, to inform a spatially explicit quantitative risk assessment with reference to potential management targets for benthic taxa and/or habitats (from Objective 1) combined across all bottom fisheries in the New Zealand EEZ. 4. To conduct spatially explicit Management Strategy Evaluation to simulate and evaluate the effects of alternate fisheries management scenarios on benthic taxa and/or habitats in the EEZ.	Ongoing	
BEN	BEN2014-02	Monitoring recovery of benthic fauna on the Graveyard complex	1. To repeat the quantitative photographic survey of benthic invertebrate communities on the Graveyard complex. 2. To assess changes in benthic communities since the first survey in 2001.	Ongoing analysis	
BEN	BEN2014-03	Monitoring recovery of benthic fauna in Spirits Bay	1. Using previous survey results, conduct a power analysis to estimate the likelihood of a range of survey designs consistent with the monitoring programme from project ENV2005/23 detecting changes in key indicators of the state of the benthic communities in Spirits Bay and Tom Bowling Bay since the last survey. 2. To survey Spirits Bay and Tom Bowling Bay benthic invertebrate communities in accordance with an agreed design from Objective 1. 3. To assess changes in benthic communities inside and outside of the closed area since 1997.	Contracted, in progress	
BEN	SEA2014-09	Review of New Zealand's SPRFMO VME protocol	1. To prepare a review of the scientific basis for the 'biodiversity component' of the move-on-rule thresholds comprising the current New Zealand Vulnerable Marine Ecosystem Evidence Process.	Complete	Penney 2014
BEN	BEN2012-02	Spatial overlap of mobile bottom fishing methods and coastal benthic habitats	1. To use existing information and classifications to describe the distribution of benthic habitats throughout New Zealand's coastal zone (0–200 m depth). 2. To rank the vulnerability to fishing disturbance of habitat classes from Objective 1. 3. To describe the spatial pattern of fishing using bottom trawls, Danish seine nets, and shellfish dredges and assess overlap with each of the habitat classes developed in Objective 1.	Complete	Baird et al. 2015
BEN	DEE2010-06	Design a camera / transect study	1. To design and provide indicative costs for a programme to monitor trends in deepwater benthic habitats and communities. 2. To explore the feasibility of using existing trawl and acoustic surveys to capture data relevant to monitoring trends in deepwater benthic habitats and communities.	Complete	Bowden et al. (2015)
BEN	DAE2010-04	Monitoring the trawl footprint for deepwater fisheries	1. To estimate the 2009/10 trawl footprint and map the spatial and temporal distribution of bottom contact trawling throughout the EEZ between 1989/90 and 2009/10. 2. To produce summary statistics, for major deepwater fisheries and the aggregate of all deepwater fisheries,	Ongoing analysis	Black et al. 2013; Black & Tilney 2015.

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			of the spatial extent and frequency of fishing by year, by depth zone, by fishable area, and by habitat class, and to identify any trends or changes.		
BEN	Internally funded 1	SPRFMO	1. To develop detection criteria for measuring trawl impacts on vulnerable marine ecosystems in high sea fisheries of the South Pacific Ocean.	Complete	Parker and Bowden 2010
BEN	Internally funded 2	SPRFMO	1. To document protection measures implemented by New Zealand for vulnerable marine ecosystems in the South Pacific Ocean.	Complete	Penney et al. 2009
BEN	Internally funded 3	CCAMLR	1. An Impact Assessment Framework for Bottom Fishing Methods in the CCAMLR Convention Area.	Complete	Sharp et al. 2009
BEN	Internally funded 4	SPRFMO	1. To develop a bottom Fishery Impact Assessment: Bottom Fishing Activities by New Zealand Vessels Fishing in the High Seas in the SPRFMO Area during 2008 and 2009.	Complete	Ministry of Fisheries 2008
BEN	BEN2009-02	Monitoring recovery of benthic communities in Spirits Bay	1. To survey Spirits Bay and Tom Bowling Bay benthic invertebrate communities according to the monitoring programme designed in ENV2005/23. 2. To assess changes in benthic communities inside and outside the closed area since 1997.	Complete	Tuck & Hewitt 2013
BEN	IFA2008-04	Guide for the rapid identification of material in the process of managing Vulnerable Marine Ecosystems	To produce a guide for the rapid identification of material in the process of managing Vulnerable Marine Ecosystems.	Complete	Tracey et al. 2008
BEN	BEN2007-01	Assessing the effects of fishing on soft sediment habitat, fauna, and processes	1. To design and test sampling and analytical strategies for broad-scale assessments of habitat and faunal spatial structure and variation across a variety of seafloor habitats. 2. To design and carry out experiments to assess the effects of bottom trawling and dredging on benthic communities and ecological processes important to the sustainability of fishing at scales of relevance to fishery managers.	Complete	Tuck et al. 2016
BEN	IFA2007-02	Development of a Draft New Zealand High-Seas Bottom Trawling Benthic Assessment Standard	1. To generate data summaries and maps of New Zealand's recent historic high-seas bottom trawling catch and effort in the proposed convention area of the South Pacific Regional Fisheries Management Organization (SPRFMO). 2. To map vulnerable marine ecosystems (VMEs) in the SPRFMO area. 3. To develop a draft standard for assessment of benthic impacts of high-seas bottom trawling on VMEs in the proposed SPRFMO convention area.	Complete	Parker 2008
BEN	BEN2006-01	Mapping the spatial and temporal extent of fishing in the EEZ	1. To update maps and develop GIS layers of fishing effort from project ENV2000/05 to show the spatial and temporal distribution of mobile bottom fishing throughout the EEZ between 1989/90 and 2004/05. 2. To produce summary statistics of major fisheries and the aggregate of all bottom impacting fisheries in terms of the extent and frequency of fishing by year, by depth zone, by fishable area, and, to the extent possible, by	Complete	Baird et al. 2009, 2011; Baird & Wood 2010;

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			habitat type. 3. To identify and document any major trends or changes in fishing effort or fishing behaviour. 4. To identify, discuss the implications of, and make recommendations on data quality and other problems with current reporting systems that complicate characterisation and quantification of bottom fishing effort. 5. To integrate information on the distribution, frequency, and magnitude of fishing disturbance with habitat characteristics throughout the EEZ, using information stored in national databases, expert opinion, and the MEC.		Leathwick et al. 2010, 2012
BEN	ENV2005-15	Information for managing the Effects of Fishing on Physical Features of the Deep-sea Environment	1. To provide an updated database that identifies all known seamounts in the 'New Zealand region', encompassing the area from 24°00' – 57°30'S, 157°00'E – 167°00'W. The database will catalogue relevant data (e.g., physical, biological, location, fishing effort) for individual seamounts. 2. To identify indicators and measures suitable for the assessment of risk pertaining to the effects of fishing disturbance on the benthic biota of seamounts, and review suitable ecological risk assessment methods, that can be derived or utilise information contained within the seamount database.	Complete	Rowden et al. 2008; Clark et al. 2010b
BEN	ENV2005-16	Investigate the Effects of Fishing on Physical Features of the Deep-sea Environment	1. To monitor changes in fauna and habitats over time on selected UTFs in the Chatham Rise area that have a range of fishing histories. 2. To continue development of the risk assessment model to predict the effects of fishing, and provide options for the management of UTF ecosystems.	Complete	Clark et al. 2010a, 2010b, 2010c, 2011
BEN	ENV2005-20	Benthic invertebrate sampling and species identification in trawl fisheries	1. To produce identification guides for benthic invertebrate species encountered in the catches of commercial and research trawlers.	Complete	Tracey et al. 2007; Williams et al. 2010; Clark et al. 2009
BEN	ENV2005-23	Monitoring recovery of the benthic community between North Cape and Cape Reinga	1. To design a monitoring programme that will provide the following quantitative estimates: i) Estimates of the nature and extent of past fishing impacts on the benthic community between North Cape and Cape Reinga; ii) Estimates of change over time in areas previously fished but subsequently closed to fishing. Estimated parameters will include indices representing biodiversity, community composition, and biogenic structure; iii) Estimates of change over time in areas environmentally comparable to those assessed in (ii), above, but subject to ongoing fishing impacts; and iv) Estimates of change over time in areas comparable to those above, but not impacted by fishing (if any such areas can be found).	Complete	Tuck et al. 2010
BEN	ZBD2005-04	Information on benthic impacts in support of the Foveaux Strait Oyster Fishery Plan	1. To assess the distribution- vulnerability to disturbance- and ecological importance of habitats in Foveaux Strait- and describe the spatial distribution of the Foveaux Strait oyster fishery relative to those habitats. 2. To assemble and collate existing information on the Foveaux Strait system between the Solander Islands and Ruapuke Island or other area to be agreed with MFish. 3. To map- using best available information- substrate type- bathymetry- wave energy- and tidal flow in this area.	Complete	Michael et al. 2006

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			4. To assess the extent to which these data can be used to define useful functional categories that might serve as habitat classes. 5. To rank the vulnerability to fishing disturbance of habitat classes developed in Objective 3 using approximate regeneration times. 6. To describe the functional role and ecosystem services provided by each habitat class developed in Objective 3- including an assessment of the relative importance of each to overall ecosystem function and productivity. 7. To describe the spatial pattern and intensity of dredge fishing for Foveaux Strait oysters over the past 10 fishing years and relate this to natural disturbance regimes and habitat classes developed in Objective 3. 8. To carry out a qualitative video survey of benthic habitats in Foveaux Strait- both within the established commercial oyster fishery area and areas outside the fishery area but within OYU 5.		
BEN	ZBD2005-15	Information on benthic impacts in support of the Coromandel Scallops Fishery Plan	1. To assemble and collate existing information on the coromandel Scallop Fishery between cape Rodney and Town Point or other, wider area to be agreed with Mfish. 2. To map, using best available information, substrate type, bathymetry, wave energy, and tidal flow in this area. 3. To assess the extent to which data can be used to define useful functional categories that might serves as habitat classes. 4. To rank the vulnerability of fishing disturbance of habitat classes developed in Objective 3 using approximate regeneration times. 5. To describe the functional role and ecosystem services provided by each habitat class developed in Objective 3, including an assessment of the relative importance of each to overall ecosystem function and productivity. 6. To describe the spatial pattern and intensity of dredge and trawl fishing within the Coromandel scallop fishery over the past 15 fishing years and relate this to natural disturbance regimes and habitat classes developed in Objective 3.	Complete	Tuck et al. 2006a, 2006b
BEN	ZBD2005-16	Information on benthic impacts in support of the Southern Blue Whiting Fishery Plan	1. To assemble and collate existing information on the Southern Blue Whiting fishery in SBW6A, SBW6B, SBW6I, and SBW6R or other wider area to be agreed with MFish 2. To map, using best available information, substratum type, bathymetry, wave energy, tides, and ocean currents in these areas 3. To assess the extent to which these data can be used to define useful functional categories that might serve as habitat categories. 4. To rank the vulnerability to fishing disturbance of habitat classes developed in Objective 3 using approximate regeneration times. 5. To describe the functional role and ecosystem services provided by each habitat class developed in Objective 3, including an assessment of the relative importance of each to overall ecosystem function and productivity. 6. To describe the spatial pattern and intensity of trawl fishing within the Southern Blue Whiting fishery over the past 10 fishing years and relate this to natural disturbance regimes and habitat classes developed in Objective 3.	Complete	Cole et al. 2007

Theme	Project code	Project title	Specific objectives	Status	Citation/s
BEN	ENV2003-03	Determining the spatial extent, nature and effect of mobile bottom fishing methods	1. To determine the spatial extent, nature and time between disturbances of mobile bottom fishing methods in the Chatham Rise trawl fisheries.	Complete	Baird et al. 2006
BEN	ENV2002-04	Benthic invertebrate sampling and specific identification in trawl fisheries	1. To quantify and map the benthic invertebrate species incidental catch in commercial and research trawling throughout the New Zealand EEZ.	Complete	Tracey et al. 2005
BEN	ENV2001-09	The effects of mobile bottom fishing gear on bentho-pelagic coupling	To describe any effects of fishing that might modify bentho-pelagic coupling (a complex, interlinked suite of processes transferring energy, oxygen, carbon, and nutrients between pelagic and benthic systems), to consider the scale of such possible effects, and to put the summary in a New Zealand context.	Complete	Cryer et al. 2004
BEN	ENV2001-15	The effects of bottom impacting trawling on seamounts	1. To design a programme in New Zealand waters previously trawled and now closed to trawling to monitor the rate of regeneration of benthic communities on seamounts.	Complete	Clark & O'Driscoll 2003; Clark & Rowden 2009
BEN	OYS2001-01	Foveaux Strait oyster stock assessment	1. To carry out a survey and determine the distribution and absolute abundance of pre-recruit and recruited oysters in both non-commercial and commercial areas of Foveaux Strait. The target coefficient of variation (c.v.) of the estimate of absolute recruited abundance is 20%. 2. To estimate the sustainable yield for the areas of the commercial oyster fishery in Foveaux Strait for the year 2002 oyster season. 3. To identify and count benthic macro-biota collected during the dredge survey.	Complete	Rowden et al. 2007
BEN	ENV2000-05	Spatial extent, nature and impact of mobile bottom fishing methods in the New Zealand EEZ	1. To determine the spatial extent, nature and impact of mobile bottom fishing methods within the New Zealand EEZ.	Complete	Cryer and Hartill 2002; Baird et al. 2002
BEN	ENV2000-06	Review of technologies and practices to reduce bottom trawl bycatch and seafloor disturbance in New Zealand	To review technologies and practices to reduce bottom trawl bycatch and seafloor disturbance in New Zealand.	Complete	Booth et al. 2002; Beentjes & Baird 2004
BEN	ENV98-05	The effects of fishing on the benthic	1. To determine the effects of fishing on the benthic community structure between North Cape and 1. Cape Reinga.	Complete	Cryer et al. 2000

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		community structure between North Cape and Cape Reinga			
ECO	ANT2017-03	Antarctic Research - Ross Sea region MPA	1. To provide advice and scientific knowledge to MPI that would allow the CCAMLR Scientific Committee to advise the Commission on: (i) the degree to which the specific objectives of the MPA are being achieved. 2. To provide advice and scientific knowledge to MPI that would allow the CCAMLR Scientific Committee to advise the Commission on the degree to which the MPA objectives are still relevant in different areas of the MPA. 3. To provide advice and scientific knowledge to MPI that would allow the CCAMLR Scientific Committee to advise the Commission on what management actions may be required to improve the achievement of the objectives for this MPA.	Approved	
ECO	MDC2015-01	MDC Benthic coring	To support benthic coring in the Marlborough Sounds by the Marlborough District Council (MDC). The results of this will provide historical information to support environmental restoration and reporting goals. It will also support potential restoration using empty mussel shell disposal.	Complete	
ECO	ENV2014-09	Spatial decision support tools for multi-use and cumulative effects	To provide a customised GIS decision support tool to help assess the cumulative effects of fishing.	Ongoing	
ECO	SEA2013-01	Provision of identification guides (sea pens and black corals)	To produce identification guides for sea pens and black corals electronically as AEBR (including MPI review).	Complete	Tracey et al. 2014; Williams et al. 2014; Opresko et al. 2014; Clark et al. 2015
ECO	ENV2012-01	A literature review of Nitrogen levels and adverse ecological effects in embayments in temperate regions.	1. To complete a literature review of Nitrogen levels and adverse ecological impacts from temperate embayments in order to assist aquaculture consenting authorities in determining at what concentration of Nitrogen adverse effects may be expected.	Complete	Hartstein & Oldman 2015
ECO	SEA2012-17	NPOA Sharks extension work	NPOA Sharks extension work.	Complete	Clarke et al. 2013
ECO	ZBD2012-02	Tier 1 statistic: Ocean	1. To identify candidate oceanographic variables for potential development as part of the proposed Tier 1 Statistic, Atmospheric and Ocean Climate Change.	Complete	Pinkerton et al. 2015a
ECO	DAE2010-01	Taxonomic identification of benthic specimens	1. To identify benthic invertebrates in samples taken during research trawls and by Observers on fishing vessels. 2. To update relevant databases recording the catch of invertebrates in research trawls and commercial fishing.	Complete	Mills et al. 2013

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ECO	DEE2010-04	Development of a methodology for Environmental Risk Assessments for deepwater fisheries	To review approaches to Ecological Risk Assessments (ERA) and methods available for deepwater fisheries both QMS and non-QMS. 2. To develop and recommend a generic, cost effective, method for ERA in deepwater fisheries by using or modifying methods identified in Objective 1.	Complete	Clark et al. submitted; Mormede & Dunn 2013
ECO	DEE2010-05	Development of a suite of environmental indicators for deepwater fisheries	1. To review the literature and hold a workshop to recommend a suite of ecosystem and environmental indicators that will contribute to assessing the performance of deepwater fisheries within an environmental context. 2. To examine available data and design a data collection programme to enable future calculation of the indicators identified in Specific Objective 1.	Complete	Tuck et al. 2014
ECO	ENV2010-03	Habitats of particular significance for inshore finfish fisheries management	1. To review the literature to determine the most important juvenile or reproductive (spawning, pupping or egg-laying) areas for inshore finfish target species. 2. To use a gap analysis to prioritise areas for future research concerning the important juvenile or reproductive (spawning, pupping or egg-laying) areas for target inshore finfish fisheries.	Complete	Morrison et al. 2014b
ECO	ENV2010-05A&B and SEA2010-15	Habitats of particular significance for fisheries management: shark nursery areas	1. Identify, from the literature, important nursery grounds for rig in estuaries around mainland New Zealand. 2. Design and carry out a survey of selected estuaries and harbours around New Zealand to quantify the relative importance of nursery ground areas. 3. Identify threats to these nursery ground areas and recommend mitigation measures.	Complete	Francis et al. 2012; Jones et al. 2016
ECO	ZBD2010-42	Development of a National Marine Environment Monitoring Programme	1. To design a Marine Evnvironment Monitoring Programme (MEMP) to track the physical, chemical and biological changes taking place across New Zealand's marine environment over the long term. 2. To prepare an online inventory (metadatabase) of repeated (time series) biological and abiotic marine observations/datasets in New Zealand. 3. To review, evaluate fitness for purpose, and identify gaps in the utility and interoperability of these datasets for inclusion in MEMP from both science and policy perspectives. 4. To design a MEMP that includes relevant existing data collection and proposed new time series.	Complete	Hewitt et al. 2014
ECO	ENV2009-04	Trends in relative mesopelagic biomass using time series of acoustic backscatter data from trawl surveys	1. To evaluate relative changes in abundance of mesopelagic fish and other biological components from acoustic records collected during Chatham Rise and Sub-Antarctic trawl surveys. 2. To explore links between trends in mesopelagic biomass and climate variables and variations, and condition indices of commercial species in the Chatham Rise and Sub-Antarctic areas.	Complete	O'Driscoll et al. 2011
ECO	ENV2009-07	Habitats of particular significance for fisheries	1. Collate and review information on the role and spatial distribution of habitats in the Kaipara Harbour that support fisheries production. 2. Assess historical, current, and potential anthropogenic threats to these habitats that could affect fisheries values, including fishing and land-based threats.	Complete	Morrison et al. 2014d

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		management: kaipara harbour	3. Design and implement cost-effective habitat mapping and monitoring surveys of habitats of particular significance for fisheries management in the Kaipara Harbour.		
ECO	GMU2009-01	Spatial Mixing of GMU1 using Otolith Microchemistry	1. To determine the level of spatial mixing and connectivity of grey mullet (*Mugil cephalus*) populations using otolith microchemistry. 2. To collect and analyse the chemical composition of grey mullet otoliths. 3. To analyse the otoliths collected under Objective 1 to determine if the samples can be spatially separated.	Complete	Morrison et al. (2016)
ECO	IPA2009-11	Trophic studies publication of review	1. To publish the comprehensive review of New Zealand-wide trophic studies completed in 2000 that was prepared by NIWA.	Complete	Stevens et al. 2011
ECO	FLA2009-01	Assess the feasibility of using juvenile netting surveys to predict adult yellow-belly & sand flounder	1. Assess the feasibility of using juvenile netting surveys to predict adult yellow-belly and sand founder abundance in the Manukau Harbour and Firth of Thames (this also examined correlations between juvenile catch and environmental factors).	Complete	McKenzie et al. 2013
ECO	AQE2008-02	Review of ecological effects of farming shellfish and other species	1. To collate and review information on the ecological effects of farming mussels (*Perna canaliculus*), including offshore mussel farming and spat catching, in the New Zealand marine environment. 2. To collate and review information on the ecological effects of farming oysters in the New Zealand marine environment. 3. To collate and review information on the ecological effects of farming species other than mussels (*Perna canaliculus*), oysters, and finfish, in the New Zealand marine environment.	Complete	Keeley et al. 2009
ECO	IFA2008-08	Inputs to the Ross Sea bioregionalisation	1. To produce one or more benthic invertebrate classifications of the Ross Sea region. 2. To use fishery catch data to examine spatial distributions of major demersal fish species. 3. To prepare other biological or environmental spatial data layers for use in the Ross Sea workshop.	Complete	Pinkerton et al. 2009a
ECO	TOH2007-03	Toheroa Abundance	1. To investigate variations in the abundance of toheroa. 2. To investigate sources of mortality of toheroa and factors affecting the recruitment of toheroa	Complete	Williams et al. 2013
ECO	BEN2007-05	Risk assessment framework for assessing fishing &other anthropogenic effects on coastal fisheries	1. To collate existing information on the distribution, intensity, and frequency of anthropogenic disturbances in the coastal zone that could be used in a risk assessment model to estimate their likely aggregate effect on ecosystem function across habitats and over different scales of ecosystem functioning and biological organisation. 2. To develop a risk assessment framework in conjunction with a variety of stakeholders and environmental scientists.	Complete	MacDiarmid et al. 2012
ECO	ENH2007-01	Stock enhancement of blackfoot paua	1. To assess the survival rate of enhanced paua from introduction into the wild through to harvest. 2. To assess the genetic diversity of hatchery spawned juvenile paua bred for enhancement purposes. 3. To assess interactions between introduced and wild paua populations and to recommend research and monitoring to quantify those impacts that are potentially adverse.	Complete	McCowan 2013

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ECO	ENV2007-04	Climate and Oceanographic Trends Relevant to New Zealand Fisheries	1. To summarise, for fisheries managers, climatic and oceanographic fluctuations and cycles that affect productivity, fish distribution and fish abundance in New Zealand.	Complete	Hurst et al. 2012
ECO	ENV2007-06	Trophic Relationships of Commercial Middle Depth Species on the Chatham Rise	1. To quantify the inter-annual variability in the diets of hoki, hake and ling on the Chatham Rise 1992–2007. 2. To quantify seasonal dietary cycles for hoki, hake and ling that have been collected from the commercial fleet throughout the year.	Complete	Horn & Dunn 2010
ECO	HAB2007-01	Biogenic habitats as areas of particular significance for fisheries management	1. To collate and review available information on the location, value, functioning, threats to, and past and current status of biogenic habitats that may be important for fisheries production in the New Zealand marine environment. 2. To identify information gaps, in the New Zealand context, and recommend measures to address those important to an ecosystem approach to fisheries management.	Complete	Morrison et al. 2014a
ECO	IPA2007-07	Land Based Effects on Costal Fisheries	1. To review and collate scientific knowledge and research on the impacts of land-based activities on coastal fisheries and biodiversity.	Complete	Morrison et al. 2009
ECO	ENV2006-04	Ecosystem indicators for New Zealand fisheries	1. To carry out a literature review of potential fish-based ecosystem indicators and identify a suite of indicators to be tested in Objective 2. 2. To test a suite of fish-based ecosystem indicators (identified by Objective 1) on existing trawl survey time series in New Zealand. The utility of these indicators for monitoring the effects of fishing in New Zealand should also be evaluated.	Complete	Tuck et al. 2009
ECO	GBD2006-01	DNA database for commercial marine fish and invertebrates	1. To collect DNA sequences for vouchered specimens of commercially important marine fishes and submit the DNA data to the international Barcode of Life Database (BOLD). 2. To collect DNA sequences for vouchered specimens of commercially important marine invertebrates and submit the DNA data to the international Barcode of Life Database (BOLD). Note: The funding was limited to $60 000 for this Objective. Therefore MFish agreed to omit the invertebrate species (Objective 2) from this project and reduce the number of fish species sequenced from 100 to 80 (up to 5 specimens per species). During the course of the project MFish staff asked NIWA to identify smoked eel product, suspect shark fillets, and possible paua slime with DNA markers, consequently the project was modified to accommodate these requests.	Complete	No reports specified as required output
ECO	IPA2006-08	Review of the Ecological Effects of Marine Finfish aquaculture: Final Report	1. Summarise and review existing information on ecological effects of finfish farming on the marine environment in New Zealand and overseas.	Complete	Forrest et al. 2007
ECO	SAP2006-06	West coast south island review	1. To publish a review document summarising oceanic and environmental research information particularly relevant to hoki- but also other fisheries- that spawn off Westland in winter.	Complete	Bradford-Grieve & Livingston 2011

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			2. Update the draft chapters prepared in 2004 by oceanographers- modellers and scientists towards the overall objective. 3. Incorporate a section on other west coast spawning fisheries.		
ECO	ENV2005-08	Experimental design of a programme of indicators	1. To assess the utility/feasibility of using demographic information to assess the effects of fishing on seabird populations. 2. To identify population indicators and to provide sampling protocols and experimental design for selected high to medium priority seabird populations. 3. To recommend experimental protocols for sampling of selected seabird populations in New Zealand influenced by fisheries mortality, employing robust-design methodology and including recommendations for inclusions of data into Ministry of Fisheries databases.	Complete	MacKenzie & Fletcher 2010
ECO	IPA2005-02 and MOF2003-03A	A guide to common offshore crabs in New Zealand Waters	1. Develop a guide to common offshore crabs in new Zealand waters	Complete	Naylor et al. 2005
ECO	SAM2005-02	Effects of climate on commercial fish abundance	To examine the possible effects of climate on fishery yields and abundance indices for commercial fisheries around New Zealand.	Complete	Dunn et al. 2009a
ECO	HOK2004-01	Hoki Population modelling and stock assessment	2. To investigate the prediction of year class strength from environmental variables.	Complete	Francis et al. 2005
ECO	AQE2003-01	Effects of aquaculture and enhancement stock sources on wild fisheries resources and the marine environment.	1. To identify, discuss the effects and qualitatively assess the risks of aquaculture and enhancement stocks improved by hatchery technology on New Zealand's wild fisheries resources and the marine environment. 2. To identify, discuss the effects and qualitatively assess the risks associated with the translocation of aquaculture and enhancement stocks on New Zealand's wild fisheries resources and the marine environment. 3. To make recommendations on priority issues, risks, or research to be undertaken, as a result of information discussed and evaluated in Objectives 1–2.	Complete	Speed 2005
ECO	EEL2003-01	Non-fishing mortality of freshwater eels	1. To undertake a feasibility study on establishing an estimate of the mortality of eels caused by hydroelectric turbines and other point sources of mortality caused by human activity.	Complete	Beentjes et al. 2005
ECO	MOF2003-01	The implications of marine reserves for fisheries resources and management in the New Zealand context	Investigations of the implications of marine resources for fisheries resources and management in the New Zealand context.	Complete	Speed et al. 2006
ECO	ENV2002–03	Beach cast seaweed review	1. To collate existing information on the role of beach-cast seaweed in coastal ecosystems to assess the nature and extent of the impacts that the removal of beach cast seaweed may have on the marine environment.	Complete	Zemke-White et al. 2005

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			2. On the basis of the review in Specific Objective 1 above, to identify key research gaps related to any marine environment impacts that the removal of beach cast seaweed may have.		
ECO	ENV2002-07	Energetics and trophic relationships of important fish and invertebrate species	1. To quantify food webs supporting important fish and invertebrate species.	Complete	Livingston 2004
ECO	CRA2000–01	Rock lobster stock assessment	Objective 11: To conduct a desktop study to identifi and explore data needs associated with managing the effects of rock lobsterfishing on the environment.	Complete	Breen 2005
ECO	ENV2000-04	Identification of areas of habitat of particular significance for fisheries management within the New Zealand EEZ	1. To review literature and existing data for all significant fish species, including all QMS species, encountered from the 200 1500 m contour within the New Zealand EEZ to: a) determine areas of important juvenile fish habitat; b) determine areas of importance to spawning fish populations; and c) determine areas of importance for shark populations for pupping or egg laying. 2. To review literature and existing data for all significant pelagic fish species (excluding highly migratory species) encountered within the New Zealand EEZ to: a) determine areas of important juvenile fish habitat; b) determine areas of importance to spawning fish populations; and c) determine areas of importance for shark populations for pupping or egg laying 3. To review literature and existing data for all significant marine invertebrate species encountered within the New Zealand EEZ to: a) determine areas of important juvenile habitat; and b) determine areas of importance to spawning populations.	Complete	O'Driscoll et al. 2003
ECO	MOF2000-02A	Future research requirements for the Ross Sea Antarctic toothfish (*Dissostichus mawsoni*) fishery.	To recommend future research requirements for the Ross Sea Antarctic toothfish (*Dissostichus mawsoni*) fishery.	Complete	Hanchet 2000
ECO	ENV99-03	Identification of areas of habitat of particular significance for fisheries management within the NZ EEZ.	1. To determine areas of habitat of importance to fisheries management within the New Zealand EEZ for selected fish species in selected areas.	Complete	Hurst et al. 2000
ECO	ENV99-04	A framework for evaluating spatial closures as a fisheries management tool	To design a framework for evaluating spatial closures as a fisheries management tool.	Complete	Bentley et al. 2004

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ECO	No project number	The fishery for freshwater eels (*Anguilla spp*.) in New Zealand	To review the fishery for freshwater eels (*Anguilla spp*.) in New Zealand.	Complete	Jellyman 1994
ZBD	ZBD2017-02	Linking primary and secondary production in the sea	1. Investigate the role that the p-ratio and the z-ratio play in modifying the relationship between primary and secondary productivity at fishery-relevant scales in New Zealand 2. Improve and refine the methodology for the projection of climate change scenarios on primary and secondary productivity.	Withdrawn	
ZBD	ZBD2017-04	Implications of ocean acidification on the capacity of carbonates in sediments to buffer eutrophication effects	1. Determine how the carbonate content of coastal benthic sediments affects sediment biogeochemistry and processes such as nitrogen recycling and removal. 2. Improve understanding of what the loss of carbonate materials from sediments in an acidified world (through dissolution) will mean for critical marine ecosystem functions.	Withdrawn	
ZBD	ZBD2016-07	Multiple stressors on coastal ecosystems – in situ	To assess the effects of global warming and ocean acidification on coastal productivity processes in New Zealand	Ongoing	
ZBD	ZBD2016-11	Quantifying benthic biodiversity	1. Collect quantitative data about seabed habitats and fauna by undertaking a survey of unsampled areas on Chatham Rise. 2. Process and compile seabed habitat and fauna data from the survey and merge these with comparable data from previous quantitative surveys on Chatham Rise. 3. Use merged data to assess the utility of existing community and species distribution models for Chatham Rise. 4. Use merged data to build new community and species distribution models for Chatham Rise.	Contracted	
ZBD	DAE2015-05	Taxonomic ID of benthic samples	To identify benthic invertebrates in samples taken during research trawls and by observers on fishing vessels.	Complete	Tracey & Mills in prep
ZBD	ZBD2014-01	Live corals: Age and growth study of deepsea coral in aquaria.	Ocean acidification and temperature manipulation are now underway to look at the physiological responses (e.g., growth) of deepsea corals to future predicted environmental conditions.	Complete	Tracey et al. 2016
ZBD	ZBD2014-03	Sublethal effects of environment change on fish populations	Co-funded by MBIE, this project explores the effects of ocean acidification on the behaviour of young snapper with a view to scaling up these effects to model long-term effects on the snapper population.	Ongoing	
ZBD	ZBD2014-04	Isoscapes for Trophic and Animal Studies	1. To generate a new tool for fisheries management and conservation. Specifically, to produce a validated, modelled, south-west Pacific and Southern Ocean carbon and nitrogen isotopic map, referred to as an 'isoscape', which will improve our understanding of trophic interactions and its relationship with marine animals.	Ongoing	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ZBD	ZBD2014-05	Modelling the effects of ocean acidification.	1. Determine how much the aragonite and calcite saturation horizons (ASH and CSH) have changed over the industrial era for the southwest Pacific, including New Zealand's EEZ.	Complete	Mikaloff-Fletcher et al. in prep
ZBD	ZBD2014-06	Macroalgae mapping and potential as national scale indicators	Many countries use seaweeds to monitor the state of the marine environment, however this approach has not been explored in New Zealand. In this project, seaweeds will be selected according to their mapped distribution and availability, and assessed for their indicator potential.	Ongoing	
ZBD	ZBD2014-07	Southern coralline algae shellfish habitat	Coralline algae are a structurally important component of coastal habitats, and play an important role in ecosystem processes. They produce chemicals which promote the settlement of the larvae of certain herbivorous invertebrates, particularly paua. Coralline algae appear to enhance larval metamorphosis and the survival of larvae through the critical settlement period. The first objective is to document critical baseline information on the diversity of coralline algae in southern New Zealand using morphological and molecular identification.	Ongoing	
ZBD	ZBD2014-09	Climate change risks and opportunities	Objective 1. To prepare a technical report that explains the most up to date issues and hypotheses with regard to observed and predicted changes to the physical, chemical and biological properties of New Zealand coastal and offshore waters. Objective 2. To prepare a synthesis of the Technical Report that will be informative and provide guidance on what can be done, for stakeholders, policymakers and resource managers.	Ongoing	
ZBD	ZBD2014-10	BPA Biodiversity	Obj 1. To update the inventory of benthic samples and biodiversity data available within BPA and Seamount Closure areas. Obj 2. To process and identify undescribed samples and material in selected BPAs and for selected taxonomic groups. Obj 3. To identify gaps in sample coverage, evaluate priority areas and design a sampling programme to collect appropriate data. Obj 4 To undertake an objective spatial management planning exercise to assess the effectiveness of the current BPAs to protect biodiversity.	Ongoing	
ZBD	ZBD2013-02	VME Genetic Connectivity	This project addresses the critical lack of data concerning deep sea genetic connectivity of VME indicator taxa, and will clarify the spatial relationships and distribution of biodiversity of several protected invertebrate VME species within New Zealand's EEZ and beyond.	Ongoing	Rowden et al. 2015
ZBD	ZBD2013-03	Continuous Plankton Recorder - Phase 2	The overall objective of the CPR programme is to map changes in the quantitative distribution of epipelagic plankton, including phytoplankton, zooplankton and euphausiid (krill) life stages, in New Zealand's EEZ and transit to the Ross Sea, Antarctica. To enable trend analysis, the Contractor will continue the annual time series for a further 5 year period (years 6–10).	Ongoing	
ZBD	ZBD2013-08	NZ-Ross sea connectivity Humpback whales	1. To determine the migration path and Antarctic feeding grounds for New Zealand humpback whales.	Complete	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ZBD	ZBD2013-06	Shell generation and maintenance of aquaculture species	Shells of individuals of NZ paua, flat oysters and cockles will undergo detailed analysis to determine how the decreased pH/increased temperature modified their shell (i) thickness, (ii) mineralogy and (iii) construction.	Complete	Cummings et al. 2013
ZBD	ZBD2013-07	Interactive keys for easy identification keys of amphipods	Generate interactive identification keys for marine Amphipoda families Synopiidae and Epimeriidae for easy and free use online.	Complete	
ZBD	ZBD2012-01	Tier 1 Stat. Marine Biodiversity	To perform a preliminary investigation of the utility and feasibility of developing the variables published by Costello et al. (2010) as a Tier 1 statistic.	Complete	Lundquist et al. 2015
ZBD	ZBD2012-03	Chatham Rise Benthos - Ocean Survey	1. In relation to the Fishing Intensity Effects Survey, determine whether there are quantifiable effects of variations in seabed trawling intensity on benthic communities. 2. In relation to the Crest Survey, conduct seabed mapping and photographic surveys in previously un-sampled areas on the central crest of the Chatham Rise.	Complete	Pinkerton et al. 2016
ZBD	SRP2011-02	IDG 2009-01 MPI fish ID field guide	1. IDG 2009-01 field guide.	Complete	McMillan 2011a, 2011b, 2011c
ZBD	ZBD2011-01	Evaluation of ecotrophic and environmental factors affecting the distribution and abundance of highly migratory species in NZ waters	Evaluation of ecotrophic and environmental factors affecting the distribution and abundance of highly migratory species in New Zealand waters.	Complete	Horn et al. 2013; McGregor & Horn 2015
ZBD	ZBD2010-39	Improved benthic invertebrate species identification in trawl fisheries	1. To revise and update the document 'A guide to common deepsea invertebrates in New Zealand waters (second edition)' to allow a third edition of this guide to be printed.	Complete	Tracey et al. 2011a
ZBD	ZBD2010-40	Predictive modelling of the distribution of vulnerable marine ecosystems in the	1. To develop and test spatial habitat modelling approaches for predicting distribution patterns of vulnerable marine ecosystems in the convention Area of the South Pacific Regional Fisheries Management Organisation with agreed international partners.	Complete	Rowden et al. 2013b

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		South Pacific Ocean region.	2. To collate datasets and evaluate modelling approaches which are likely to be useful to predict the distribtuion of vulnerable marine ecosystmes in the South pacific Ocean region.		
ZBD	ZBD2010-41	Ocean acidification in fisheries habitat	1. To assess the risks of ocean acidification to deep sea corals and deepwater fishery habitat. 2. To determine the carbonate mineralogy of selected deep sea corals found in the New Zealand region. 3. To assess the distribution of deep sea coral species in the New Zealand region relative to improved knowledge of current and predicted aragonite and calcite saturation horizons, assessment of potential locations vulnerable to deep water upwelling. 4. Through a literature search and analysis, determine the most appropriate tools to age and measure the effects of ocean acidification on deep sea habitat-forming corals, and recommend the best approach for future assessments of the direct effects.	Complete	Tracey et al. 2011b
ZBD	ZBD2009-25	Predicting impacts of increasing rates of disturbance on functional diversity in marine benthic ecosystems	1. Further develop the landscape ecological model of disturbance/recovery dynamics in marine benthic communities, incorporating habitat connectivity, based on existing model by Lundquist, Thrush, and Hewitt. 2. Predict impacts of increasing rates of disturbance on rare species abundance, functional diversity, relative importance of biogenic habitat structure, and ecosystem productivity. 3. Use literature and expert knowledge to quantify rare species abundance, biomass, functional diversity, habitat structure, and productivity of various successional community types in the model. 4. Field test predictions of the model in appropriate marine benthic communities where historical rates of disturbance are known, and benthic communities have been sampled.	Complete	Lundquist et al. 2010, 2013
ZBD	IPA2009-14	Bryozoan identificaiton guides	1. For each of ~50 species of common bryozoans, provide photos and text to allow for identification. Provide information on distribution and habitat (as far as is known) and further references for each species and on bryozoans as a whole. 2. Submit these data for publication in the Ministry of Fisheries series New Zealand Aquatic Environment and Biodiversity Research.	Complete	Smith & Gordon 2011
ZBD	ZBD2009-03	To evaluate the vulnerability of New Zealand rhodolith species to environmental stressors and to characterise diversity of rhodolith beds.	1. To characterise the distribution and physical characteristics of two New Zealand rhodolith beds and characterise the associated biodiversity. 2. To measure the growth rates and evaluate the vulnerability of New Zealand species of rhodoliths to environmental stressors.	Complete	Nelson et al. 2012
ZBD	ZBD2009–10	Multi-species analysis of coastal marine connectivity	1. Determine overall patterns of regional connectivity in a broad range of NZ coastal marine organisms to define the geographic units of genetic diversity for protection and the dispersal processes that maintain this diversity. 2. Review previous studies of marine connectivity and population genetics in NZ coastal organisms to determine the preliminary range of patterns observed and the principal gaps (taxonomic geographic and	Complete	Gardner et al. 2010

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			ecological) in our understanding. 3. In a range of invertebrate and vertebrate marine organisms determine geographic patterns of genetic variation using standardised sampling and molecular techniques. 4. Analyse data across past and present studies to reveal both common and unique patterns of connectivity around the NZ coastline and the locations of common barriers to dispersal.		
ZBD	ZBD2009-13	Ocean acidification impact on key NZ molluscs	1. Controlled laboratory experiments will be used to determine the effect of pCO2 levels that are predicted to occur in NZ waters over the next few decades on appropriate life history stages of at least two key NZ mollusc species. A number of response variables will be assessed. 2. Implications of these responses to the local and broader ecosystems will be assessed.	Complete	Cummings 2011; Cummings et al. 2011, 2013
ZBD	ZBD2008-01	Biogenic large–habitat–former hotspots in the near-shore coastal zone (50–250 m); quantifying their location, identity, function, threats and protection	1. To collect and integrate existing knowledge on biogenic habitat-formers in the <5–150 m depth zone of New Zealand's continental shelf, from sources including structured fisher interviews, primary and grey literature, and other sources as available. 2. Using the findings of Objective 1, design and deploy a series of sampling voyages to selected locations, to map and characterise locations of significant biogenic structure (either still existing, or historical), and collect relevant biological samples (both through visual census, and physical collection). 3. Process and analyse the samples collected in Objective 2, to provide a hierarchical, quantitative description of the biogenic habitats and associated species encountered. 4. Using the findings from Objective 1–3, assess the present status, likely extent, ecological role, and threats to, biogenic habitat formers in the <5–150 m depth zone. This should include a spatial modelling and risk assessment framework. Integrate (as appropriate) with other information sources and/or approaches that may exist by the year 2010/11.	Complete	Jones et al. 2016, submitted
ZBD	ZBD2008-05	Macroalgal diversity associated with soft sediment habitats	1. Conduct a targeted collection programme across diverse soft sediment environments to develop a permanent reference collection of representative macroalgae. 2. Examine algal distribution in soft sediment habitats in relation to selected environmental variables. 3. Prepare an annotated checklist of macroalgae found in soft sediment environments in the New Zealand region.	Complete	Neill et al. 2012
ZBD	ZBD2008-07	Carbonate sediments: the positive and negative effects of land-coast interactions on functional diversity	1. To quantify shifts in community structure and functional diversity in mollusc dominated habitats along gradients associated with an estuary-coast interface in two locations. 2. To characterise the influence of estuary-derived food sources across these gradients for key species. 3. To measure changes in growth of key species in relation to changes in food supply and land-derived sediment impacts. 4. To quantify carbon and nitrogen uptake and tissue turnover rates of key species in laboratory experiments.	Complete	Thrush et al. 2011; Savage et al. 2012
ZBD	ZBD2008-11	Predicting changes in plankton biodiversity and productivity of the EEZ in response to	1. To document the spatial and inter-annual variability of coccolithophore abundance and biomass- and assess in terms of the phytoplankton abundance- biomass and community composition in sub-tropical and sub-Antarctic water.	Complete	Law et al. 2012; Boyd & Law 2011

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		climate change induced ocean acidification	2. To document the seasonal and inter-annual variability of foraminifera and pteropod abundance and biomass at fixed locations in sub-tropical and sub-Antarctic water by analysis of sediment trap material from time-series data collection. 3. To document the spatial and seasonal distribution of the key coccolithophore species- *Emiliana huxleyi*- using both archived and ongoing ingestion of satellite images of Ocean Colour- and ground-truth the reflectance. 4. To determine the sensitivity and response of *E. huxleyi* and other EEZ coccolithophores to pH under a range of realistic atmospheric CO2 concentrations in perturbation experiments- using monocultures and mixed populations from in situ sampling. 5. To document the spatial variability of diazotrophs (nitrogen-fixing organisms) and associated nitrogen fixation rate- and assess in terms of phytoplankton abundance- biomass and community composition in sub-tropical waters north of the STF. 7. To determine the sensitivity of- and response of *Trichodesmium* spp. and other diazotrophs to pH under a range of realistic atmospheric CO2 concentrations in perturbation experiments using monocultures		
ZBD	ZBD2008-14	What and where should we monitor to detect long-term marine biodiversity and environmental changes-remote sensing, biota, context, inshore offshore workshop	1. Identify the key questions to be addressed by long-term monitoring of marine biodiversity and environment. 2. Identify appropriate monitoring indices, how they should be spatially distributed and their sampling frequency. 3. Identify relevant existing monitoring programmes across the range of New Zealand agencies and science providers and identify gaps. 4. Provide those agencies setting environmental goals/ standards or research needs (MoRST, FRST, MFish, DOC, MfE, Commissioner for the Environment) with a thorough situational analysis, including a list of priority monitoring projects/plans.	Complete	Livingston 2009
ZBD	ZBD2008-15	Continuous plankton recorder project: implementation and identification	1. To set up a time series of annual CPR data collection by deployment from a toothfish vessel on the annual summer transit between New Zealand and the Ross Sea. 2. To identify phytoplankton and zooplankton according to strict observation protocols determined by the SAHFOS[1] CPR Survey and SO-CPR[2]. 3. To enter species data, frequency and location along the transect into a spreadsheet that will allow spatial mapping of the plankton density and distribution. 4. To analyse the full dataset after 5 years of data collection to: (a) determine trends in the dataset and (b) compare results with Australian datasets available through SO-CPR. 5. To evaluate the continuation of the programme.	Complete	Robinson et al. 2014
ZBD	ZBD2008-20	Ross sea benthic ecosystem function: predicting consequences of shifts in food supply	1. To increase understanding of Ross Sea coastal benthic ecosystem function. 2. Conduct in situ investigations into responses to and utilisation of primary food sources by key species, at two contrasting coastal Ross Sea locations.	Complete	Cummings & Lohrer 2011; Cummings et al. 2011; Lohrer et al. 2013

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ZBD	ZBD2008-22	Acidification and ecosystem impacts in NZ and southern ocean waters (data collected during IPY).	1. To assess the response of cocolithophorids, and their replacement by non-calcifying organisms during incubation under a range of dissolved CO2 concentrations. 2. To describe and characterise changes in abundance and biodiversity of microbial components of the samples incubated at sea under a range of dissolved CO2 concentrations. 3. To predict the likely impacts of higher acidity on foodwebs and on carbon fixation under scenarios to be encountered in the Southern Ocean under forecasted trends associated with climate change.	Complete	Maas et al. 2010b
ZBD	ZBD2008-23	Macroalgae diversty and benthic community structure at the Balleny Islands	1. To describe and characterise macroalgae diversity from the Balleny Islands and the Western Ross Sea. 2. To describe and quantify benthic community structure from one location at the Balleny Islands 3. To complete anatomical and morphological investigations & molecular sequencing required for the identification of macroalgae samples from the Balleny Islands & western Ross Sea coastline to describe & characterise macroalgae diversity in Balleny Isds 4. To process and analyse samples collected at the Balleny Islands- to analyse them using ICECUBE methodology and compare results with those from other ICECUBE sampling locations along the Ross Sea coastline	Complete	Nelson et al. 2010
ZBD	ZBD2008-27	Scoping investigation into New Zealand abyss and trench biodiversity	1. Review what is already known of abyssal, canyon and trench faunas in NZ. 2. Review what is already known of abyssal, canyon and trench faunas around the world. 3. Prioritise science questions and locations for exploration. 4. Assess NZ capacity to sample at the required depths; identify sampling equipment needs. 5. Design a suitable vessel-based sampling programme.	Complete	Lörz et al. 2012b
ZBD	ZBD2008-50	OS2020 Chatham Rise Biodiversity Hotspots	1. To improve understanding of the effects of trawl fishing in New Zealand on the biodiversity of seamounts-knolls and hills. 2. To describe differences in benthic biodiversity between northwestern and eastern regions of the Chatham Rise. 3. To continue the time series of observations in the NW Chatham Rise to demonstrate recovery in terms of biodiversity. 4. To extend the observations on fished-unfished contrasts and recovery of fauna on protected seamounts to an oceanographically distinct location.	Complete	Clark et al. 2009
ZBD	IPY2007-01	International polar year census of antarctic marine life post-voyage analysis: Ross Sea - Southern Ocean Biodiversity	1. To measure seabed depth and rugosity using the multibeam system to identify topographic features such as bottom type, iceberg scouring, seamounts etc and to determine areas for targeted benthic faunal sampling. 2. To continue the analysis of opportunistic seabird and marine mammal distribution observations from this and previous BioRoss voyages and published records, and in relation to environmental variables. 3. To identify and determine near-surface spatial distribution, diversity and abundance of phytoplankton, and zooplankton, based on Continuous Plankton Recorder samples collected during transit to and from the Ross Sea. 4. To collect & analyse data collected both underway, & at stations for salinity, temperature nutrient and chlorophyll a data, spot optical measurements with the SeaWiFS. 5. To identify and determine the spatial distribution, abundance (biomass), diversity, and size structure of	Complete	Allcock et al. 2009, 2010, submitted; Alvaro et al. 2011; Baird & Mormede 2014; Bowden et al. 2011a, in prep; Clark et al. 2010a; Dettai et al. 2011; Eakin et al. 2009; Eléaume et al.

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			epipelagic, mesopelagic (and possibly bathypelagic) species using acoustics and net sampling. 6. To identify and measure diversity, distribution & densities of mesozooplankton, macrozooplankton & meroplankton (as collected by all plankton sampling methods except transit CPR samples). 7. To determine diversity, distribution & densities of viral, bacterial, phytoplankton & microzooplankton species in the water column. 8. To determine the spatial distribution, abundance (biomass), diversity, and size structure of shelf and slope demersal fish species and associated invertebrate species using a demersal survey. 9. To determine the diversity, abundance/density, spatial distribution, and physical habitat associations of benthic assemblages across a body size spectrum from megafauna to bacteria, for shelf, slope, seamounts, and abyssal sites in Ross Sea. 10. To describe trophic/ecosystem relationships in the Ross Sea ecosystem (pelagic and benthic, fish and invertebrates). 11. Assess molecular taxonomy and population genetics of selected Antarctic fauna and flora to estimate evolutionary divergence within and among ocean basins in circumpolar species. Provide DNA barcoding.		2011, in prep; Ghiglione et al. 2012; Gordon 2000; Grotti et al. 2008; Hanchet et al. 2008a, 2008b, 2008c, 2008d, 2013 ; Hanchet 2009, 2010; Heimeier et al. 2010; Hemery et al. in prep; Koubbi et al. 2011; Leduc et al. 2012a, 2012b, 2012c, 2013, 2014; Linse et al. 2007; Lörz 2009, 2010a, 2010b; Lörz & Coleman 2009; Lörz et al. 2007, 2009, 2012a, 2012b, 2012c, in prep; Maas et al. 2010a; McMillan et al. 2012; Mitchell 2008; Nielsen et al. 2009; Norkko et al. 2005; O'Driscoll 2009; O'Driscoll et al. 2009, 2010, 2012; O'Loughlin et al. 2011; Pakhomov et al. 2011; Pinkerton et al. 2007a, 2009b, 2010, 2013; Schiaparelli

Theme	Project code	Project title	Specific objectives	Status	Citation/s
					et al. 2006, 2008, 2010; Smith et al. 2011a, 2011b; Stein 2012; Strugnell et al. 2012
ZBD	IPY2007-02	International polar year census of antarctic marine life post-voyage analysis: Ross Sea - Southern Ocean Biodiversity	1. To measure and describe key elements of species distribution- abundance (density or biomass) & biodiversity for the Ross Sea and Southern Ocean for main habitats and key functional ecosystem roles- for major groups- viruses- bacteria- archaea. 2. To report on the diversity of Antarctic Cephalopoda (Octopus and Squid)- including a complete inventory of taxa- & reports on ontogenetic & sexual variation in species- their systematics- diversity- distribution- life histories- & trophic importance. 3. To Beak/Biomass Regression Equations. 4. Life cycle determination.	Complete	Garcia 2010
ZBD	ZBD2007-01	Chatham-Challenger Oceans 20/20 Post-Voyage	1. To quantify in an ecological manner- the biological composition and function of the seabed at varying scales of resolution- on the Chatham Rise and Challenger Plateau. 2. To elucidate the relative importance of environmental drivers- including fishing- in determining sea bed community composition and structure. 3. To determine if remote-sensed data (e.g., acoustic) and environmentally derived classification schemes (e.g., marine environmental classification system) can be utilised to predict bottom community composition-function and diversity. 4. To count- measure- and identify to species-level (where possible- otherwise to genus) all macro invertebrates (> 2 mm) and fish collected during Oceans 20/20 voyages. 5. To count- measure and identify to species-level (where possible- otherwise to genus or family) all meiofauna (> 2 mm) from multicore samples collected during the Oceans 20/20 voyages. 6. To count- measure and identify to species- level (where possible- otherwise to genus or family) all fauna collected by hyper-benthic sled during the Oceans 20/20 voyages. 7. To count- measure- and identify to species-level all macrofauna observed on DTIS images collected during the Oceans 20/20 voyages. The number of biogenic features (burrows/mounds) and habitat (spatial) complexity should also be estimated. 8. To count- measure- and identify to species-level (where possible- otherwise to genus or family) all macrofauna observed on DTIS video footage collected during the Oceans 20/20 voyages. 9. To calculate and compare the performance of a suite of diversity measures (species and taxonomic-based) at varying levels of resolution. 10. To estimate particle size composition and organic content of sediment samples. Sediment samples should be aggregated over the top 5 cm of sediment. 11. To measure the bacterial biomass (top 2 cm) of the sediment and in the sediment surface water samples-	Complete	Bowden 2011; Bowden et al. 2011b, 2014; Bowden & Hewitt 2012; Coleman and Lörz 2010; Compton et al. 2012; Floerl et al. 2012; Hewitt et al. 2011a, 2011b; Lörz 2011a, 2011b; Nodder et al. 2012

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			collected during the Oceans 20/20 voyages. 12. To elucidate the relationships- patterns and contrasts in species composition- assemblages- habitats- biodiversity and biomass (abundance) both within and between stations- strata and areas. 13. To define habitats (biotic) encountered during the survey and assess their relative sensitivity to modification by physical disturbance- their recoverability and their importance to ecosystem function / production. 14. To quantify the productivity- energy flow (trophic networks) and the energetic coupling (bentho pelagic or otherwise) of the area surveyed areas at various levels of resolution. 15. To assess the extent to which patterns of species distributions and communities can be predicted using environmental data (including fishing) collected during the Ocean 20/20 voyages or held in other databases. 16. To provide an interactive- high resolution mapping facility for displaying & plotting all data collected & derived indices. Includes environmental data- the abundance of species- indices of biomass or diversity- and statistically derived groupings. 17. To assess the extent to which acoustic, environmental or other remote-sensed data can provide cost-effective reliable means of assessing biodiversity at the scale of the Oceans 20/20 surveys. 18. To assess the extent to which the 2005 MEC and subsequent variants can provide cost-effective reliable means of assessing biodiversity at the scale of the Oceans 20/20 surveys. 19. Collating all information and analysis from all objectives- devise a series of statistically supported recommendations for surveying marine biodiversity in the future. Including – but may not be limited to – statistical analyses and modelling.		
ZBD	ZBD2006-02	Ongoing NABIS development	As part of NABIS, users will be able to identify spatial information relating to the annual distribution (average distribution over the period of a year) of particular species within the waters around New Zealand and in the terrestrial environment (including off shore islands) of New Zealand. Users will also be able to interrogate metadata and attribute data related to the information layers presented. Users will employ NABIS to identify where a particular species is found, to identify what species are found within an area of interest, and be able to compare the spatial distribution of a particular species with other information layers. 2. Some species may have notable changes in their spatial distribution throughout a year. For such species, users of NABIS will be able to view spatial information relating to the seasonal distribution of particular species within the waters around New Zealand and in the terrestrial environment (including offshore islands) of New Zealand. Users will also be able to interrogate metadata and attribute data related to the information layers presented. For species with a seasonal component to their biological distribution, users will employ NABIS to identify where a particular species is found within the waters around New Zealand and in the terrestrial environment (including off shore islands) of New Zealand at a particular time of the year, to identify what species are found within an area of interest at a particular time of year, or be able to compare the distribution of a particular species at a particular time of year, with other information layers. 3. To provide analysis of the data used in determining the hotspot distribution.	Complete	Anderson 2007

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ZBD	ZBD2006-03	Antarctic coastal marine systems	1. Quantify patterns in benthic community structure and function at two coastal Ross Sea locations (Terra Nova Bay and Cape Evans). 2. Quantify benthic community structure and function at selected locations in Terra Nova Bay and Cape Evans.	Complete	Cummings et al. 2003, 2006b, 2008; Thrush & Cummings 2011; Thrush et al. 2010
ZBD	ZBD2006-04	Chatham/challenger oceans 20/20	1. To collect seabed fauna, sediment samples and photographic images along transects in the Chatham Rise and the Challenger Plateau, as determined by the sampling protocol described in the Voyage Programmes for Voyages 2 and 3 of the project. Multibeam data should be collected opportunistically as time allows. 2. To describe the distribution of broad macro epifauna groups (I.D. level to be determined at sea during Surveys 2 & 3), their relative abundance, the substrate and habitat types, including representative photographic images of each sea-bed habitat and associated fauna along transects in the survey areas. 3. To provide a description of the observed evidence of fishing along transects. 4. To provide indicative measures of alpha biodiversity (richness, number of taxonomic groups) at appropriate scales within and between transects, and between the Chatham Rise and the Challenger Plateau. 5. To determine broad scale variability in sea-bed habitats and associated biodiversity within and between MEC classes at 20 class level. 6. To process and archive biological samples and data into databases and collections for future analysis in meeting the Overall Objectives above.	Complete	Nodder 2008; Nodder et al. 2011
ZBD	ZBD2005-01	Balleny Islands Ecology Research, Tiama Voyage (2006)	1. To characterise shallow benthic communities across a range of habitat settings around the Balleny Islands, utilising a range of data collection methodologies (including SCUBA-based rock-wall suspension feeder photo quadrats, SCUBA-based linear video transects, and drop camera photography), and to analyse community patterns with reference to possible physical/oceanographic, biological, and/or biogeographic influences on community structure. 2. To characterise aspects of the marine food web of the Balleny Islands area, using stable isotope analysis of specimens from important functional groups, and to make inferences about factors affecting ecosystem-scale trophodynamics in the Balleny Islands area and potential implications for the function of the wider ecosystem. 3. To characterise the spatial and temporal distributions of higher-level consumer species (birds, seals and whales) and of dominant pelagic prey (i.e., krill swarms) by opportunistically recording all at-sea sightings, and by systematic observation of landbased top predators (birds and seals) while sailing along the coast of the islands. 4. To collect and photograph and/or retain fish specimens from shallow benthic environments using a range of fishing methods, including food-baited fish traps, lightbaited fish traps, rotenone sampling, and/or baited lines. 5. To continuously collect bathymetric data and water-column acoustic data (i.e., mesopelagic acoustic marks) throughout the voyage, using an acoustic sounder. 6. To opportunistically collect a variety of data/materials during shore-based landings, including wherever possible: i) breast feathers from living penguins; ii) tissue samples/feathers/bones from dead seals/penguins/other sea birds; iii) seal scats; iv) visual estimates of adult and juvenile penguin numbers; v)	Terminated	Smith 2006

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			visual assessments of penguin colony status; vi) photographs of penguin colonies; vii) sediment excavations of occupied and abandoned colonies. (Where appropriate these data will contribute to Objective 2).		
ZBD	ZBD2005-02	Marine Environment Classification Project	1. Co-fund the Marine Environment Classification Project (being done by NIWA) with the Department of Conservation.	Complete	Snelder et al. 2005, 2006; Leathwick et al. 2006a, 2006b, 2006c
ZBD	ZBD2005-03	Tangaroa ross seaRoss Sea voyage	1. To test the feasibility of obtaining estimates of demersal fish relative abundance using cameras with and without flood lights in areas of high importance for the Ross Sea toothfish fishery (principally 800–1200 m). 2. To utilise deepwater camera transects, supported by other direct sampling methods, to characterise the relative abundance, distribution, and diversity of demersal fish species (assuming Objective 1 yields satisfactory results) and of benthic macro-invertebrates, and to examine relationships between demersal fishes and benthic habitats/communities. Camera transects will be deployed opportunistically, with focus on the following high-priority areas (in order of high to low priority) wherever possible: i) Areas of the continental shelf break at depths of high importance for the toothfish fishery (principally 800-1200 m but also 600-800m & 1200-1500 m if time permits), ii) Shallow (50-200 m) water in the immediate vicinity of the Balleny Islands; iii) Deeper water in the vicinity of the Balleny Islands; iv) seamounts around and between Scott Island and the Balleny Islands; and v) at other locations (< 600 m) as opportunity arises (e.g., around Scott Island, western Ross Sea, south-eastern Ross Sea). 3. To collect specimens/tissues of selected benthic and pelagic organisms with priority in the vicinity of the Balleny Islands (and to the east/southeast, for pelagic specimens especially Antarctic krill species) and deliver specimens to other projects for stable isotope analysis in order to contribute to understanding of trophic relationships. 4. To acquire a continuous acoustic survey of the water column, opportunistically undertake species verification of acoustic marks, integrate the acoustic marks and produce a GIS map of verified and unverified distributions of functionally important mesopelagic species (e.g., krill, Antarctic silverfish). 5. To undertake routine identification and abundance estimates of marine mammal and seabird species and deliver raw and GIS summarised data to other related projects in order to generate spatially and temporally explicit population biomass and foraging distribution estimates for top air-breathing predators in the Ross Sea. 6. To undertake automated water sampling in order to monitor the identities and spatial and temporal distributions of plankton in the Ross Sea region and to allow ground-truthing of data collection from satellites (e.g., surface seawater temperature, and chlorophyll-a concentration).	Complete	MacDiarmid & Stewart 2015; Mitchell & MacDiarmid 2006
ZBD	ZBD2005-05	Long-term effects of climate variation and human impacts on the structure and functioning of New	1. To estimate changes in marine productivity via fluctuations in ocean climate and terrestrial nutrient input over the last 1000 years. 2. To assess and collate existing archaeological, historical and contemporary data (including catch records and stock assessments) on relevant components of the marine ecosystem to provide a detailed description of change in the shelf marine ecosystem in two areas of contrasting human occupation over last 1000 years.	Complete	Carroll et al. 2015; Lalas et al. 2014; Lalas & MacDiarmid 2014; Lorrey et al. 2013;

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		Zealand shelf ecosystems	3. To collect additional oral histories from Maori and non-Maori fishers and shellfish gathers regarding the distribution, sizes and relative abundance (compared to present availability) of key fish and invertebrate stocks in both regions during the first half of the 20th century before the start of widespread modern industrial fishing. 4. To build mass-balance ecosystem models (e.g., Ecopath) of the coastal and shelf ecosystem in each area for five critical time periods: now, 60 years BP (before modern industrial fishing), 250 years BP (before European whaling and sealing), 600 y BP (early Maori phase) and 1000 years BP (before human settlement). 5. To use qualitative modelling techniques to determine the critical interactions amongst species and other ecosystem components in order to identify those that should be a priority for future research.		MacDiarmid et al. 2016a, 2016b, 2018, submitted; Maxwell & MacDiarmid 2016; McKenzie & MacDiarmid submitted; Neil et al. 2012; Parsons et al. 2011; Paul 2012, 2014; Pinkerton et al. 2015b; Smith 2011
ZBD	ZBD2005-09	Rocky reef ecosystems - how do they function? Integrating the roles of primary and secondary production, biodiversity and connectivity across coastal habitats	1. To develop a qualitative numerical model of how New Zealand's rocky reef systems are functionally structured. 2. To quantify the effects of human predation, and environmental degradation across reef gradients – top-down, or bottom-up functioning? 3. To advance our understanding of how subtidal reef systems are fuelled through primary and secondary production (from a range of sources), the role that biodiversity plays, and how this varies across different reef settings. 4. To quantify how subtidal reef systems are linked with other habitats and ecosystems at broader spatial scales, including the connectivity of MPAs with other habitats and areas.	Complete	Beaumont et al. 2011
ZBD	ZBD2004-01	Baseline information on the diversity and function of marine ecosystems	1. To quantify, and compare, the macro-invertebrate assemblage composition of a number of seamounts at the southernmost end of the Kermadec volcanic arc. 2. To compare the macro-invertebrate diversity of the southernmost end of the Kermadec volcanic arc with that of seamounts already sampled and reported on.	Complete	Rowden & Clark 2010; Smith et al. 2008
ZBD	ZBD2004-02	Ecosystem-scale trophic relationships: diet composition and guild structure of middle-depth fish on the chatham rise	1. To quantitatively characterise the diets of abundant middle-depth fish species on the Chatham Rise, by analysis of fish stomach contents collected from the January 2005, January 2006 and January 2007 Chatham Rise middle-depths trawl surveys. 2. To quantitatively characterise Chatham Rise fish diets throughout the year, for a period of 24 months, by analysis of fish stomach contents collected opportunistically aboard industry vessels. 3. To describe and examine patterns of diet variation within each fish species as a function of spatial, temporal, and environmental variables, and of fish size. 4. To define and characterise trophic guilds for abundant fish species on the Chatham Rise, using multivariate analysis of fish diet data, and to analyse the nature and relative strength of potential trophic interactions between guilds.	Complete	Connell et al. 2010; Dunn 2009; Dunn et al. 2009b, 2010a, 2010b, 2010c; Forman & Dunn 2010; Horn et al. 2010; Stevens & Dunn 2010

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			5. To create and populate a diets database to store all of the dietary information collected under Objectives 1 and 2, and for use in subsequent dietary studies.		
ZBD	ZBD2004–05	Assessment and definition of the biodiversity of coralline algae of northern New Zealand	1. To assess and define the biodiversity of coralline algae in northern New Zealand. 2. To develop rapid identification tools for coralline algae using molecular sequencing data. 3. To contribute representative material to the national Coralline Algal Collections. 4. To produce ID guides to common coralline algae of northern New Zealand.	Complete	Farr et al. 2009
ZBD	ZBD2004-08	Sea-grass meadows as biodiversity and connectivity hotspots	1. Quantify the biodiversity values and functioning of New Zealand sea-grass assemblages. 2. Complete national bio-geographic assessment of sea-grass associated biodiversity. 3. Quantify sea-grass connectivity with surrounding marine landscapes through nursery functions and detritus export. 4. Quantify sea-grass replication connectivity mechanisms. 5. Develop a risk assessment and appraisal model for sea-grass systems.	Complete	Morrison et al. 2014c
ZBD	ZBD2004-10	Development of bioindicators in coastal ecosystems	1. Investigate linkages between land use patterns in catchments and nitrogen loading to recipient estuaries and coastal ecosystems. 2. Characterise isotopic signatures of selected bioindicator organisms in relation to different terrestrial nutrient loads. 3. Validate the use of bioindicators using controlled laboratory and field experiments.	Complete	Savage 2009
ZBD	ZBD2004-19	Ecological function and critical trophic linkages in New Zealand soft-sediment habitats	1. Define the interactive effects of two functionally important benthic species in maintaining critical trophic linkages in soft-sediment systems from a series of integrated field experiments. 2. Quantify effects of heart urchins (*Echinocardium australe*) on sediment properties- benthic primary production- and macrofaunal diversity through manipulative field experiments in Mahurangi Harbour. 3. Test for interactions between pinnid bivalves (*Atrina zelandica*) and heart urchins (*Echinocardium australe*) in field experiments- and measure their respective and combined contributions to sediment properties- benthic primary production- and macrofau na 4. Determine the dependence of results from objectives 1 and 2 (functional contributions of *Echinocardium* and *Atrina*) in an environmental context by conducting experiments along an estuarine-coastal gradient.	Complete	Lohrer et al. 2010
ZBD	ZBD2003-02	Biodiversity of Coastal Benthic Communities of the North Western Ross Sea.	1. Quantify patterns in biodiversity and community structure in the coastal Ross Sea region. 2. Quantify biodiversity in benthic communities at selected locations in the Ross sea north of Terra Nova Bay. 3. Describe ecosystem function at selected locations in the Ross Sea north of Terra Nova Bay.	Complete	Cummings et al. 2003, 2006a, 2010; De Domenico et al. 2006; Guidetti et al. 2006; Norkko et al. 2004
ZBD	ZBD2003-03	Biodiversity of deepwater invertebrates and fish	1. To describe, and quantify the diversity of, the benthic macroinvertebrates and fish assemblages of the Balleny Islands and adjacent seamounts, and to determine the importance of certain environmental variables influencing assemblage composition.	Complete	Rowden et al. 2012a, 2013a;

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		communities of the north western Ross Sea			Mitchell & Clark 2004
ZBD	ZBD2003–04	Fiordland Biodiversity Research Cruise	1. How can ecotone boundaries be defined? 2. If you have an ecotone boundary defining the edge of a commercial exclusion zone how wide is the transition zone across the boundary? 3. If you have an area delineated as a marine protected area or a commercial exclusion zone, does it adequately represent the different habitats or biodiversity of the whole region?	Complete	Wing 2005
ZBD	ZBD2003-09	Macquarie Ridge Complex Research Review	To review and summarise both biological and physical research carried out on or around the section of the Macquarie Ridge Complex that lies between New Zealand and Macquarie Island.	Complete	Grayling 2004
ZBD	ZBD2002-01	Ecology of Coastal Benthic Communities in Antarctica	To research the ecology of coastal benthic Communities in Antarctica.	Complete	Cummings et al. 2003; Schwarz et al. 2003, 2005; Sharp et al. 2010; Sutherland 2008; Thrush et al. 2006; Thrush & Cummings 2011
ZBD	ZBD2002-02	Whose larvae is that? Molecular identification of planktonic larvae of the Ross Sea.	1. To use molecular sequencing tools in the taxonomic identification of cryptic/invasive marine 2. To provide a molecular description and characterisation of gobies that are introduced (*Arenigobius bifrenatus* and *Acentrogobius pflaumii*) cryptogenic (*Parioglossus marginalis*) or native (eg.*Favonigobius lentiginosus* and *F. expuisitus*). 3. To describe the molecular diversity of the above species throughout their native and introduced distributions- and characterise a range of the greatest potential invasive gobioid and blennioid species from the Australasian region. 4. To develop molecular criteria to rapidly identify invasive or cryptogenic gobioid and blennioid fish	Complete	Sewell 2005, 2006; Sewell et al. 2006
ZBD	ZBD2002-06A	Impacts of terrestrial run-off on the biodiversity of rocky reefs	1. Conduct field and laboratory experiments to determine relationships between sediment loading, epifaunal assemblages, and mortality of filter feeding invertebrates. 2. Conduct field and laboratory experiments to identify the influence of sediment on early life stages of key grazers. 3. Determine photosynthetic characteristics and survival of large brown seaweeds and understorey algal species in relation to a sediment gradient.	Complete	Schwarz et al. 2006
ZBD	ZBD2002-12	Molecular identification of cryptogenic/invasive	1. To use molecular sequencing tools in the taxonomic identification of cryptic/invasive marine species 2. To provide a molecular description and characterisation of gobies that are introduced (*Arenigobius bifrenatus* and *Acentrogobius pflaumii*) cryptogenic (*Parioglossus marginalis*) or native (eg.*Favonigobius*	Complete	Lavery et al. 2006

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		marine species – gobies.	*lentiginosus* and *F. expuisitus*). 3. To describe the molecular diversity of the above species throughout their native and introduced distributions- and characterise a range of the greatest potential invasive gobioid and blennioid species from the Australasian region. 4. To develop molecular criteria to rapidly identify invasive or cryptogenic gobioid and blennioid fish.		
ZBD	ZBD2002-16	Joint New Zealand and Australian Norfolk Ridge	1. To describe the marine biodiversity of the Norfolk Ridge and Lord Howe Rise seamount communities. 2. To survey, sample and document the marine biodiversity and environmental data from seamounts on the Norfolk Ridge and Lord Howe Rise to a depth of at least 1000 m depth. 3. To preserve samples of fishes and invertebrates and hold these in accessible curated museum collections to support biosystematic research projects. 4. To provide specimens to support projects which research the identity, diversity, relationships, distributions, and assess uniqueness and conservation value of the marine life. 5. To correlate observed distribution patterns, especially areas of high diversity and areas of endemism, with measured biological and physical parameters.	Complete	Clark & Roberts 2008
ZBD	ZBD2002-18	Quantitative survey of the intertidal benthos of Farewell Spit Golden Bay	1. To undertake a baseline survey of intertidal macrobenthic organisms at Farewell Spit Nature Reserve and adjacent flats. 2. To undertake an initial field survey of Zostera distribution at Farewell Spit Nature Reserve and adjacent intertidal flats. 3. To undertake a preliminary survey of sediment characteristics of the intertidal flats at Farewell Spit Nature Reserve and adjacent flats.	Complete	Battley et al. 2005
ZBD	ZBD2001–02	Documentation of New Zealand Seaweed	1. To publish a regional algal flora of Fiordland based on voucher herbarium specimens. 2. To assemble a database of references and to review the current state of knowledge about New Zealand macroalgae.	Complete	Nelson et al. 2002
ZBD	ZBD2001-03	Ecology and biodiversity of coastal benthic communities in Antarctica.	1. To develop sampling protocols for estimating the relative abundance of algae and benthic invertebrates. 2. To quantify patterns in biodiversity and benthic community structure at two locations in McMurdo Sound. 3. To analyse Ross Island Sea-Level data.	Complete	Norkko et al. 2002
ZBD	ZBD2001-04	'Deep Sea New Zealand'	To help publish the book 'Deep Sea New Zealand'.	Complete	Batson 2003
ZBD	ZBD2001-05	Crustose coralline algae of New Zealand	1. To assess the biodiversity of crustose coralline algae in NZ using modern taxonomic methods and molecular sequence tools. 2. To establish the NZ National Coralline Algal Collection. 3. To produce identification guides to NZ species.	Complete	Harvey et al. 2005; Farr et al. 2009; Broom et al. 2008
ZBD	ZBD2001-06	Biodiversity of New Zealand's soft-	1. To review the current knowledge of the biodiversity of macroinvertebrates and macrophytes living in and on soft-sediment substrates in New Zealand's harbours- estuaries- beaches and to 1000 m water depth. 2. To review existing published and unpublished sources of information on soft-sediment marine assemblages	Complete	Rowden et al. 2012b

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		sediment communities	around New Zealand. 3. Using the results of Objective 1, identify gaps in the knowledge, hotspots of biodiversity, areas of particular vulnerability, and make recommendations on areas or assemblages that could be the subject of directed research in future years.		
ZBD	ZBD2001-10	Additional Research on Biodiversity of Seamounts	1. To determine the macro-invertebrate assemblage composition on Cavalii seamount, and adjacent seamount W1, by photographic transects and epibenthic sled sampling. 2. To determine the distniution of macro-invertebrate assemblages on the seamounts. 3. To compare the macro-invertebrate species diversity of neighbouring seamounts. 4. To evaluate and collect samples fiom suitable macro-invertebrate species for genetic analysis. 5. To map bathymetry and habitat characteristics of the seamounts. 6. To compare macro-invertebrate assemblage composition of the seamounts with nearby hard bottom low relief (under 100 m) on the slope, if suitable areas can be located.	Complete	Rowden et. al 2004
ZBD	MOF2000–01	Bryozoan thickets off Otago Peninsula	To research the bryozoan thickets off the Otago Penninsula.	Complete	Batson & Probert 2000
ZBD	ZBD2000–01	A review of current knowledge describing the biodiversity of the Ross Sea region	1. To review and document existing published and unpublished information describing the biodiversity of the Ross Sea region. 2. To identify and document Ross Sea region marine communities that are under high pressure or likely to come under high pressure from human activities in the near future.	Complete	Bradford-Grieve & Fenwick 2001a, 2001b, 2002; Fenwick & Bradford-Grieve 2002a, 2002b; Varian 2005
ZBD	ZBD2000-02	Exploration and description of the biodiversity, in particular the benthic macrofauna, of the western Ross Sea	1. To utilise sampling opportunities provided by the presence of RV Tangaroa in the western Ross Sea in February / March 2001 to make collections of (primarily) benthic organisms as a contribution to the understanding of biodiversity in the region. 2. To identify and document the organisms collected and provide for their proper storage in national collections. 3. To describe the logistic constraints of working in the Ross Sea region, and make recommendations for future research to improve understanding of biodiversity in the Ross Sea.	Complete	Page et al. 2001
ZBD	ZBD2000-03	The spatial extent and nature of the bryozoan communities at Separation Point, Tasman Bay	1. To assess the present state and extent of bryozoan communities around Separation Point. 2. To characterise the bryozoan communities around Separation Point.	Complete	Grange et al. 2003

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ZBD	ZBD2000-04	Supplementary Research on Biodiversity of Seamounts	1. To determine the biodiversity of seamounts of the southern Kermadec volcanic arc (Rumble V, Rumble 111, Brothers). 2. To describe the distribution of fauna, with an emphasis on mapping the nature and extent, of biodiversity associated with hydrothermal vents. 3. To compare the biodiversity of the three seamounts, and adjacent slope. 4. To collect samples from near the vent sources (if possible, as these are thought to be very localised) to measure chemical and thermal aspects of the environment.	Complete	Rowden et al. 2002, 2003; Clark & O'Driscoll 2003
ZBD	ZBD2000-06	'The Living Reef: The Ecology of New Zealand's Rocky Reefs'	1. Funding to support the publication of this book.	Complete	Andrew & Francis 2003
ZBD	ZBD2000-08	A review of current knowledge describing New Zealand's Deepwater Benthic Biodiversity	1. To review and document existing published and unpublished reports and data describing New Zealand's deepwater benthic biodiversity. 2. To make recommendations on representative communities and potentially impacted communities that could be the subject of directed research.	Complete	Key 2002
ZBD	ZBD2000-09	Antarctic fish taxonomy	1. Ross Sea fishes processing and identification.	Complete	Roberts & Stewart 2001

19.09 APPENDICES FROM CHAPTER 9 FISH AND INVERTEBRATE BYCATCH

Appendix 19.09.1: Bycatch trends for seven deepwater trawl fisheries and one longline fishery (1990–91 to 2013–14). Regression slopes for each species/species group and fishery. Slopes indicating a decline in bycatch over time are highlighted in red, and slopes indicating an increase in bycatch over time are highlighted in green. Species/species groups are ordered alphabetically; blank cells = not estimated; LLL = ling longline fishery; HHL = hoki/hake/ling fishery. NB: These linear regression slopes should be considered only a simple indicator of general changes as relationships may be non-linear; some trends may be strongly influenced by changes in observer recording of species over time. The main purpose of the highlighted cells is to draw attention to species for which closer examination of trends may be warranted. [Continued on next pages]

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
ACS		0.01	0.22			0.06		0.19	Actinostolidae
ADT			0.02						*Aphrodita* spp.
AER			0.02						*Aeneator recens*
AFO			0.04						*Aristaeomorpha foliacea*
AGR								-0.18	*Agrostichthys parkeri*
AIR			-0.01						*Argyripnus iridescens*
ALB		0.01			0.45			0.00	*Thunnus alalunga*
ALL			0.07						*Alcithoe larochei*
ANC									*Engraulis australis*
ANT		0.00	-0.09	-0.02		0.03	-0.01	0.11	Anthozoa
ANZ		0.02							*Ecionemia novaezelandiae*
API		-0.03	0.04					0.00	*Alertichthys blacki*
APR		0.01	0.06	-0.02		0.02	0.02	0.09	*Apristurus* spp.
ARE			0.02						*Apatopygus recens*
ASR	0.01	0.11	0.08	-0.04		0.01	-0.02	0.16	Asteroid
AST			-0.02	-0.02					Astronesthinae
ATT					0.49				*Arripis trutta*
AWI			0.05						*Alcithoe wilsonae*
BAC						-0.03			*Bathygadus cottoides*
BAM			0.04						*Bathyplotes* spp.
BAR	0.00	-0.01	-0.01		-0.04			-0.11	*Thyrsites atun*
BAS		-0.01	-0.20	-0.10				0.06	*Polyprion americanus*
BAT		-0.01				0.00	-0.01		*Rouleina* spp.
BBE	-0.02	0.03	-0.03			-0.04	0.03	0.03	*Centriscops humerosus*
BCA		0.00						-0.09	*Magnisudis prionosa*
BCD		0.19	-0.01	-0.12				-0.01	*Paranotothenia magellanica*
BCO	-0.03	0.11	-0.01	-0.04				0.00	*Parapercis colias*
BCR			-0.01					-0.03	*Brotulotaenia crassa*
BDA								-0.01	*Sphyraena novaehollandiae*
BEE			0.00			-0.09	0.11	0.04	*Diastobranchus capensis*
BEL		0.06	-0.01			-0.01		0.13	*Centriscops* spp.
BEN								0.20	*Benthodesmus* spp.
BER			-0.06					0.00	*Typhlonarke* spp.
BES			0.02					0.03	*Benthopecten* spp.
BFE						0.00			*Bathysaurus ferox*
BFI						0.00		0.00	*Bathophilus filifer*
BFL		0.01							*Rhombosolea retiaria*
BGZ		0.11							*Kathetostoma binigrasella*
BIG		0.01						-0.02	*Thunnus obesus*
BJA						-0.02	0.03		*Mesobius antipodum*
BKM								-0.04	*Makaira indica*
BNE								-0.01	*Benthodesmus elongatus*
BNS		-0.07	-0.28	-0.31		-0.19	0.01	-0.10	*Hyperoglyphe antarctica*
BNT								-0.01	*Benthodesmus tenuis*
BOA	-0.03		0.02					-0.01	*Paristiopterus labiosus*
BOC		0.01	0.11						*Bolocera* spp.
BOE						-0.20		0.05	*Allocyttus niger*
BOO						0.01			*Keratoisis* spp.

Appendix 19.09.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
BOT		0.00						-0.01	Bothidae
BPE			-0.02					-0.01	*Caesioperca lepidoptera*
BPI								0.02	*Benthopecten pikei*
BRA		-0.01						0.01	*Dasyatis brevicaudata*
BRC			-0.08	0.01		-0.02		0.01	*Pseudophycis breviuscula*
BRG						0.08			Brisingida
BRS	-0.01							-0.01	*Echinorhinus brucus*
BRZ			0.04						*Xenocephalus armatus*
BSH	-0.01	-0.06	-0.14	-0.11		-0.11	-0.03	-0.01	*Dalatias licha*
BSK		0.16				-0.02		-0.16	*Cetorhinus maximus*
BSL						-0.12	0.03	0.11	*Xenodermichthys* spp.
BSP				-0.02				0.02	*Taractichthys longipinnis*
BSQ	-0.02					-0.03		-0.08	*Sepioteuthis australis*
BTA			0.09					0.07	*Brochiraja asperula*
BTH	-0.03	0.01	0.05	0.01		0.04	0.01	0.04	*Notoraja* spp.
BTS			-0.02					0.10	*Brochiraja spinifera*
BWH			0.10			0.00			*Carcharhinus brachyurus*
BWS		0.03		-0.08				-0.06	*Prionace glauca*
BYD								0.13	*Beryx decadactylus*
BYS		0.00	0.06	0.01		0.08		0.19	*Beryx splendens*
BYX		0.01	-0.22	-0.04		-0.25		-0.10	*Beryx splendens & B. decadactylus*
CAL			0.07						*Caenopedina porphyrogigas*
CAM			0.10						*Camplyonotus rathbunae*
CAR		0.24	0.12	0.02		-0.02		0.14	*Cephaloscyllium isabellum*
CAS		0.06	0.05					-0.04	*Coelorinchus aspercephalus*
CAY							0.01		*Caryophyllia* spp.
CBB		0.02	0.02			0.08			Coral rubble
CBD		0.08				0.01	0.02		Coral rubble - dead
CBE		0.04	-0.03					0.03	*Notopogon lilliei*
CBI							0.00	-0.02	*Coelorinchus biclinozonalis*
CBO	-0.04	0.00	-0.02			0.00		-0.02	*Coelorinchus bollonsi*
CBX								-0.01	*Cubiceps baxteri*
CCA								0.00	*Cubiceps caeruleus*
CCO			0.01					0.02	*Coelorinchus cookianus*
CCR		0.00							*Cetonurus crassiceps*
CCX								0.07	*Coelorinchus parvifasciatus*
CDL						-0.20	-0.01	0.02	Epigonidae
CDO		0.05	0.05		-0.41			0.18	*Capromimus abbreviatus*
CDX			0.13					-0.01	*Coelorinchus maurofasciatus*
CDY			0.01						*Cosmasterias dyscrita*
CEN				-0.01		-0.04			Squalidae
CFA			0.03					0.02	*Coelorinchus fasciatus*
CHA								0.01	*Chauliodus sloani*
CHC		0.02							*Chaceon bicolor*
CHG				0.01		0.02	0.09	0.05	*Chimaera lignaria*
CHI			-0.03	-0.05		0.04	0.00	-0.06	*Chimaera* spp.
CHM								-0.01	Chiasmodontidae
CHP				-0.04		0.01	0.04	-0.01	*Chimaera* sp.
CHQ		0.06						0.02	Cranchiidae
CHR							0.03		*Chrysogorgia* spp.
CHX			-0.04			0.00		0.01	*Chaunax pictus*
CJA			0.09					0.12	*Crossaster multispinus*
CMA								0.02	*Coelorinchus matamua*
CMT		0.02							Comatulida
CMU							0.01	-0.02	*Coryphaenoides murrayi*
COB						0.01			Antipatharia

Appendix 19.09.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
COD						0.01	-0.01	-0.02	Cod
COF		0.01						0.01	*Flabellum* spp.
COL			0.02			-0.01		0.13	*Coelorinchus oliverianus*
CON	-0.02	0.07	0.00	-0.31		-0.04	0.00	0.11	*Conger* spp.
COR			-0.01			0.00	0.00		Stylasteridae
COU		-0.01	-0.01			-0.04	-0.05	0.01	Alcyonacea, Scleractinia, Antipatharia, Stylasteridae
CPA			0.10					0.06	*Ceramaster patagonicus*
CPD								-0.03	Centrolophidae
CRA		-0.02						-0.02	*Jasus edwardsii*
CRB		-0.16	-0.12	-0.01		-0.03	0.00	0.02	Crab
CRM		0.08						0.02	*Callyspongia* cf *ramosa*
CRN		0.02							Sea lily, stalked crinoid
CRS						-0.01			*Callyspongia ramosa*
CRU		-0.04	-0.07					-0.01	Crustacea
CSH		0.04	0.00	0.01		-0.04	-0.01	0.13	Catshark
CSP		-0.01							*Coelorinchus spathulatus*
CSQ		-0.01	0.02	0.08		0.11	0.04	0.10	*Centrophorus squamosus*
CST								-0.01	*Caristius* sp.
CSU						0.02			*Coryphaenoides subserrulatus*
CTU			-0.01					-0.01	*Cookia sulcata*
CUB							-0.01	-0.01	*Cubiceps* spp.
CUC		-0.02	-0.07					0.00	*Paraulopus nigripinnis*
CVI			0.02						*Pycnoplax victoriensis*
CYL						0.14		0.15	*Centroscymnus coelolepis*
CYO				-0.03		0.14		0.11	*Centroscymnus owstoni*
CYP			0.01	-0.02		0.16	0.13	0.13	*Centroscymnus crepidater*
DAP			0.15						*Dagnaudus petterdi*
DAS			0.01						*Pteroplatytrygon violacea*
DCO			0.02						*Notophycis marginata*
DCS			-0.03	0.00		-0.02		-0.04	*Bythaelurus dawsoni*
DDI			0.06			0.02	0.01		*Desmophyllum dianthus*
DEA	0.00							-0.12	*Trachipterus trachypterus*
DEQ						-0.02		-0.02	*Deania quadrispinosum*
DHO			0.01			0.02		0.01	*Dermechinus horridus*
DIR			0.07						*Diacanthurus rubricatus*
DIS						0.00			*Diretmus argenteus*
DMG			0.11					0.09	*Dipsacaster magnificus*
DPO								-0.02	*Desmodema polystictum*
DSK		0.01	-0.13	0.00		0.01	-0.04	0.10	*Amblyraja hyperborea*
DSP	-0.02	0.03							*Congiopodus coriaceus*
DSS						0.00		-0.01	*Bathylagus* spp.
DWE			-0.04	-0.04		-0.04	-0.01	0.14	Whelks
DWO								0.18	*Graneledone spp.*
ECH			-0.05	-0.01		-0.01	0.00	-0.04	Echinodermata
ECN			0.01	-0.01		-0.01	-0.02	0.01	Echinoid
EEL			-0.16	-0.01		0.00	-0.01	-0.09	Eels
EEX		0.03							*Enypniastes eximia*
EGA			0.02						*Euciroa galatheae*
EGR					0.20				*Myliobatis tenuicaudatus*
ELE									*Callorhinchus milii*
ELT									*Electrona spp.*
EMA	0.00	0.03			-0.04			-0.20	*Scomber australasicus*
EMO			-0.02			-0.01		0.01	*Etmopterus molleri*
EPD								0.02	*Epigonus denticulatus*
EPL		0.03	0.01			-0.11	-0.03	0.21	*Epigonus lenimen*

Appendix 19.09.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
EPO			-0.02						*Melanostigma gelatinosum*
EPR			0.02			0.06		0.13	*Epigonus robustus*
ERA		0.02	-0.01		0.24			0.04	*Torpedo fairchildi*
ERO						0.04			*Enallopsammia rostrata*
ETB	-0.02	0.05	0.06	0.03		0.08	0.26	0.24	*Etmopterus baxteri*
ETL		0.03	0.08	-0.18		-0.14	0.04	0.06	*Etmopterus lucifer*
ETM		-0.03	-0.04	-0.09		-0.11	0.01	-0.24	*Etmopterus* sp.
ETP						-0.04	-0.01	-0.01	*Etmopterus pusillus*
EUC			0.05			-0.02		0.13	*Euclichthys polynemus*
EZE		0.07	0.07						*Enteroctopus zealandicus*
FAN								-0.01	*Pterycombus petersii*
FHD		0.03	0.03					0.08	*Hoplichthys haswelli*
FLA		0.13	-0.03					-0.03	Flatfish
FLO		0.01						-0.02	Flounder
FMA		0.01	0.18					0.19	*Fusitriton magellanicus*
FOR								-0.02	*Forsterygion* spp.
FRO	0.00	0.08	-0.05		-0.01	-0.03		-0.10	*Lepidopus caudatus*
FRS						-0.05		-0.02	*Chlamydoselachus anguineus*
FRX								-0.01	Trichiuridae
FTU		0.01							*Auxis thazard*
GAO						0.00			*Gadomus aoteanus*
GAS			0.19					0.05	Gastropoda
GAT			0.03						*Gastroptychus* spp.
GDU			0.02			0.16	0.09		*Goniocorella dumosa*
GFL		0.14							*Rhombosolea tapiri*
GIZ	0.00	0.07	-0.08	-0.01	0.16	-0.03		0.00	*Kathetostoma giganteum*
GLS	0.01					0.03		0.12	Hexactinellida
GMC		0.04	0.23					0.02	*Leptomithrax garricki*
GMU		-0.01							*Mugil cephalus*
GOB							-0.01		*Mitsukurina owstoni*
GON		0.25						0.08	*Gonorynchus forsteri & G. greyi*
GOR								0.05	*Gorgonocephalus* spp.
GOU			0.02			0.03			*Goniocidaris umbraculum*
GPA			0.07						*Goniocidaris parasol*
GRC						0.01	0.04	-0.01	*Tripterophycis gilchristi*
GRM							0.02	0.04	*Gracilechinus multidentatus*
GSA								-0.01	*Hoplostethus gigas*
GSC		0.38	0.13	-0.03		0.00		0.09	*Jacquinotia edwardsii*
GSH	-0.09	0.10	0.03	-0.28	0.25	-0.18	-0.18	-0.09	*Hydrolagus novaezealandiae*
GSP	0.13	0.14	0.14	0.07		0.09	0.17	0.16	*Hydrolagus bemisi*
GSQ	0.00	0.00				0.01		0.02	*Architeuthis* spp.
GUR		-0.01	0.00		-0.07			0.03	*Chelidonichthys kumu*
GVO			0.09					0.01	*Provocator mirabilis*
HAG			-0.08	-0.03				0.18	*Eptatretus cirrhatus*
HAK	-0.05	0.06	-0.06	-0.06		-0.03	-0.02		*Merluccius australis*
HAL								0.01	*Halosauropsis macrochir*
HAP		0.04	-0.05	-0.18				-0.03	*Polyprion oxygeneios*
HAT									Sternoptychidae
HCO		0.02	0.01	0.09		-0.01		-0.01	*Bassanago hirsutus*
HEC			0.02						*Henricia compacta*
HEP			-0.04					0.06	*Heptranchias perlo*
HEX		0.05	-0.06	0.01				0.15	*Hexanchus griseus*
HGB						0.01		0.00	*Hydrolagus* sp. D
HIS			0.02						*Histocidaris* spp.
HJO			0.00			0.05	0.12	0.01	*Halargyreus johnsonii*

Appendix 19.09.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
HMT			0.21					0.07	Hormathiidae
HOK	-0.12	0.04	-0.06	0.01	-0.19	-0.09	0.13		*Macruronus novaezelandiae*
HOL								0.01	*Holtbyrnia* sp.
HOR								-0.01	*Atrina zelandica*
HPB		-0.08	-0.20	-0.18				-0.16	*Polyprion oxygeneios & P americanus*
HSI			0.13						*Haliporoides sibogae*
HTH		-0.02	0.04			0.07	0.02	0.05	Holothurian unidentified
HTR			0.07					0.08	*Hippasteria phrygiana*
HYA	0.03	0.03	0.07			0.02		0.31	*Hyalascus* sp.
HYB				0.00					*Hydrolagus homonycteris*
HYD						0.00	0.01	-0.01	*Hydrolagus* sp.
HYM			0.07						*Hymenocephalus* spp.
HYP						0.00			*Hydrolagus trolli*
IBR						0.05	0.02	0.00	*Isistius brasiliensis*
ISI							0.01		Isididae
JAV	0.06	0.20	-0.01	-0.03	0.18	0.00	0.08	0.04	*Lepidorhynchus denticulatus*
JDO					0.01			-0.02	*Zeus faber*
JFI		0.00	-0.06		0.04	0.01	0.01	0.05	Jellyfish
JGU		-0.01	-0.03					0.00	*Pterygotrigla picta*
JMA	0.00	-0.16	-0.14			-0.03		-0.25	*Trachurus declivis, T. murphyi, T. novaezelandiae*
KIC			-0.02			0.04	-0.02	0.04	*Lithodes murrayi, Neolithodes brodiei*
KIN			-0.02		0.12			0.01	*Seriola lalandi*
KWH			0.01					0.01	*Austrofucus glans*
LAE			0.00			-0.03	-0.01		*Laemonema* spp.
LAG			0.09						*Laetmogone* spp.
LAN		0.14	0.00			0.00	0.01	0.07	Myctophidae
LCH	0.02		0.01			-0.02	0.03	0.03	*Harriotta raleighana*
LDO	-0.01	0.06	-0.04			-0.05	0.00	0.00	*Cyttus traversi*
LEA		-0.01			-0.19				*Meuschenia scaber*
LEG						-0.06	0.04	0.00	*Lepidion schmidti & Lepidion inosimae*
LHE								-0.02	*Lampanyctodes hectoris*
LHO			0.08					0.02	*Lipkius holthuisi*
LIN	-0.04	0.05	-0.12			-0.08	-0.06		*Genypterus blacodes*
LLC		0.07	0.02					0.02	*Leptomithrax longipes*
LMI			0.03						*Leptomithrax* spp.
LMU						0.01		0.03	*Lithodes murrayi*
LNV								0.05	*Lithosoma novaezelandiae*
LPI							0.02		*Lepidion inosimae*
LPS						0.02	-0.01		*Lepidion schmidti*
LSK		0.01	0.08			-0.02		0.10	*Arhynchobatis asperrimus*
LSO		-0.02	0.04					0.00	*Pelotretis flavilatus*
LUC			-0.04			-0.02		-0.02	*Luciosudus* sp.
MAK	0.03	0.03	-0.01	0.00	0.36	-0.05		-0.06	*Isurus oxyrinchus*
MAN	-0.04	-0.02				0.02		-0.07	*Neoachiropsetta milfordi*
MCA						0.11	0.26	0.00	*Macrourus carinatus*
MDO		0.02	0.02			-0.03		0.03	*Zenopsis nebulosa*
MIC	-0.02								*Microstoma microstoma*
MIQ	-0.05		-0.07			-0.09	-0.01	0.07	*Onykia ingens*
MNI			0.13						*Munida* spp.
MOC						0.06	0.02		*Madrepora oculata*
MOD			0.00			0.15	0.16	0.16	Moridae
MOK		-0.02	-0.01			-0.02		-0.09	*Latridopsis ciliaris*
MOL			-0.02					-0.02	Molluscs
MOO	-0.14	0.01						-0.18	*Lampris guttatus*
MOR						0.00	0.00	-0.01	Muraenidae

Appendix 19.09.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
MRL								0.00	Muraenolepididae
MRQ								0.04	*Onykia robsoni*
MSL			0.05						*Mediaster sladeni*
MST						0.03		0.02	*Melanostomiidae*
MUR						-0.02			*Muraenolepis marmoratus*
MUU		0.00							Mullet
NCA		0.03							*Nectocarcinus antarcticus*
NCB		0.47						0.01	*Nectocarcinus bennetti*
NEB						0.07		0.01	*Neolithodes brodiei*
NEX								0.00	Nemichthyidae
NMP		0.12	-0.06		0.20			-0.08	*Nemadactylus macropterus*
NOC								0.01	*Notacanthus chemnitzi*
NOR							0.01		*Normichthys yahganorum*
NOT		-0.06	-0.03	-0.26		0.00			Nototheniidae
NSD		0.01	0.00	-0.02		0.00		0.21	*Squalus griffini*
NTO		0.01							*Notomithrax* spp.
NTU								-0.02	*Thunnus thynnus*
NUD			0.04						Nudibranchia
OAR								-0.08	*Regalecus glesne*
OCO			0.02						*Octopus* spp.
OCP			0.01					-0.02	Octopod
OCT	0.00	0.05	-0.04			0.02	-0.01	-0.05	*Pinnoctopus cordiformis*
ODO			0.01					-0.01	*Odontaspis ferox*
OEO						-0.13		-0.09	*P. maculatus, A. niger, & N. rhomboidalis*
OFH			-0.05			0.01		0.00	*Ruvettus pretiosus*
OLY			0.02						*Ophiomusium lymani*
ONG	-0.03	0.15	0.10	0.00		0.07	-0.01	0.06	Porifera
OPA	-0.02	0.15	0.06					0.02	*Hemerocoetes* spp.
OPE		0.17	-0.02		0.20	-0.01		-0.04	*Lepidoperca aurantia*
OPH						-0.02			Ophiuroid
OPI			0.11					0.26	*Opisthoteuthis* spp.
OPL		0.01							Opheliidae
ORH			-0.02				-0.01	-0.08	*Hoplostethus atlanticus*
OSE								0.00	*Ophisurus serpens*
OSK			0.20			0.04		0.18	Rajidae
OSP							0.01	0.00	*Crassostrea gigas*
PAB						0.03	0.07		*Paragorgia arborea*
PAD		-0.30							*Ovalipes catharus*
PAG			0.04						Paguroidea
PAH	0.23							0.00	*Lampris immaculatus*
PAL								-0.01	*Paralepididae*
PAM			0.05						*Pannychia moseleyi*
PAO			0.02					0.01	*Pillsburiaster aoteanus*
PCH			0.05						*Penion chathamensis*
PCO			-0.04						*Auchenoceros punctatus*
PDG		0.06	0.00			-0.06		0.05	*Oxynotus bruniensis*
PDO			0.00						*Paphies donacina*
PDS								0.02	*Paradiplospinus gracilis*
PED			-0.03						*Aristaeopsis edwardsiana*
PFL			0.02						*Pseudechinus flemingi*
PHO		0.03				-0.01		0.02	*Phosichthys argenteus*
PHW		0.02							*Psammocinia cf hawere*
PIG	-0.08	0.20	0.03					0.05	*Congiopodus leucopaecilus*
PIL					0.00				*Sardinops sagax*
PIN						-0.01		0.01	*Idiolophorhynchus andriashevi*

Appendix 19.09.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
PKN								0.08	*Plutonaster knoxi*
PLS		0.02	0.00	0.05		0.04	0.01	0.04	*Proscymnodon plunketi*
PLT			0.01					0.02	*Plutonaster spp.*
PLY			0.02						*Polycheles* spp.
PLZ			-0.05						*Pleuroscopus pseudodorsalis*
PMO			0.03					0.02	*Pseudostichopus mollis*
PMU			0.08						*Paramaretia peloria*
PNE			0.09						*Proserpister neozelanicus*
PNN			0.02						*Pennatula spp.*
PNO			0.03						*Pteropeltarion novaezelandiae*
POM	0.01								Bramidae
POP					0.02				*Allomycterus jaculiferus*
POR	-0.02	-0.04						-0.23	*Nemadactylus douglasii*
POS	0.02	0.01	0.02	-0.11				-0.08	*Lamna nasus*
PRA			0.08					0.00	Prawn
PRK			0.19						*Ibacus alticrenatus*
PRU			0.04					0.01	*Pseudechinaster rubens*
PSE			0.00				-0.01		*Pseudechinus* spp.
PSI			0.22					0.14	*Psilaster acuminatus*
PSK		0.01	0.06	0.00		0.07	-0.01	0.15	*Bathyraja shuntovi*
PSL						-0.01	0.01		*Paralomis dosleini*
PSO								-0.02	*Psolus* spp.
PSP								0.01	*Psenes pellucidus*
PSQ						0.02		0.09	*Pholidoteuthis massyae*
PSY			-0.04			0.02	0.00	-0.02	*Psychrolutes microporos*
PTO				-0.02			0.00		*Dissostichus eleginoides*
PZE								0.01	*Paralomis zealandica*
QSC		0.15							*Psychrochlamys delicatula subantactica*
RAG						0.03	0.01	-0.09	*Pseudoicichthys australis*
RAT	-0.07	0.07	-0.02	-0.12	0.32	-0.03	0.08	0.02	Macrouridae
RAY			-0.06				-0.02	0.02	Torpedinidae, Dasyatidae, Myliobatidae, Mobulidae
RBM	0.08	-0.11		-0.24	0.09	-0.01		-0.05	*Brama brama*
RBT	0.00	0.01	0.01		-0.05			0.05	*Emmelichthys nitidus*
RBY		0.01	-0.08					-0.18	*Plagiogeneion rubiginosum*
RCH						0.05		0.05	*Rhinochimaera pacifica*
RCK			0.00						Acanthoclinidae
RCO	0.04	0.05	-0.07	-0.24	0.38			-0.07	*Pseudophycis bachus*
RDO		0.11	-0.01		0.07			0.06	*Cyttopsis roseus*
RHY			0.15		0.02	0.06		0.18	*Paratrachichthys trailli*
RIB		0.04	-0.20	-0.27		-0.06	-0.03	0.00	*Mora moro*
RIS								0.05	*Bathyraja richardsoni*
RMU								-0.02	*Upeneichthys lineatus*
ROC		0.01	-0.02			0.02	0.02	0.00	*Lotella rhacinus*
RPE			-0.03						Red perch
RPI									*Bodianus vulpinus*
RSC						0.00			*Scorpaena papillosa*
RSK	0.02	0.23	0.14	0.11		0.01		0.11	*Zearaja nasuta*
RSN			-0.01					-0.02	*Centroberyx affinis*
RSO		-0.11	-0.09	-0.01	0.11			-0.01	*Rexea solandri*
RSQ		0.02				-0.07		0.00	*Ommastrephes bartrami*
RUD			-0.07			-0.06	-0.03	-0.02	*Centrolophus niger*
SNA		-0.05	-0.02		0.07	-0.04		-0.09	*Pagrus auratus*
SAF						0.01			*Synaphobranchus affinis*
SAI								0.01	*Istiophorus platypterus*
SAR						0.00			*Squilla armata*

Appendix 19.09.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
SAW								-0.02	*Serrivomer* spp.
SBI	0.02					-0.11	-0.02	-0.03	*Alepocephalus australis*
SBK			-0.04			-0.03		0.05	*Notacanthus sexspinis*
SBO	-0.04	-0.01	0.03			0.00		0.06	*Pseudopentaceros richardsoni*
SBR		0.02	-0.06			-0.03	0.01	0.08	*Pseudophycis barbata*
SBW		0.13	-0.01				0.02	0.20	*Micromesistius australis*
SCA		0.02							*Pecten novaezelandiae*
SCD		0.11						0.01	*Notothenia microlepidota*
SCG			-0.06		0.20			0.01	*Lepidotrigla brachyoptera*
SCH		0.11	-0.05	-0.23	0.07	0.03		0.04	*Galeorhinus galeus*
SCI								0.10	*Metanephrops challengeri*
SCM		0.00	0.00	-0.04		0.03	0.01	0.08	*Centroscymnus macracanthus*
SCO			0.02	-0.02		0.01		0.13	*Bassanago bulbiceps*
SDE							0.02	-0.02	*Cryptopsaras couesii*
SDF			0.04					0.02	*Azygopus pinnifasciatus*
SDL						0.01			*Scorpaena cardilis*
SDM			0.14					0.01	*Sympagurus dimorphus*
SDO		0.38	-0.02		-0.18			0.12	*Cyttus novaezealandiae*
SDR						-0.01		0.01	*Solegnathus spinosissimus*
SEE			-0.03	0.04				0.07	*Gnathophis habetus*
SER			0.02						*Sergestes* spp.
SEV		0.04	0.03	-0.02				0.11	*Notorynchus cepedianus*
SFL		0.05							*Rhombosolea plebeia*
SHA	0.00	0.06	-0.12	-0.11		-0.15	-0.12	-0.04	Shark
SHE						-0.01		-0.04	*Scymnodalatias sherwoodi*
SHL			-0.06						*Scyllarus* sp.
SHR			0.00						*Aplysiomorpha*
SIA						0.12	0.06		Scleractinia
SKA	-0.05	-0.08	-0.38	-0.37	0.00	-0.08	-0.04	-0.33	Rajidae & Arhynchobatidae
SKJ		0.01			0.12				*Katsuwonus pelamis*
SLB								0.03	*Scymnodalatias albicauda*
SLC						-0.02			*Slosarczykovia circumantarctica*
SLG			-0.04			0.00			*Scutus breviculus*
SLK						0.01	0.15	0.14	Alepocephalidae
SLR			-0.04			0.00			*Optivus elongatus*
SLS		0.00							*Peltorhamphus tenuis*
SMA		0.01							*Stigmatophora macropterygia*
SMC			0.02			-0.04	0.06	-0.04	*Lepidion microcephalus*
SMI		0.02				0.00		0.07	*Somniosus microcephalus*
SMK		0.02	0.23						*Teratomaia richardsoni*
SMO		0.05							*Sclerasterias mollis*
SMT			0.04						*Spatangus mathesoni*
SND		0.03	-0.11	-0.01		0.03	0.06	-0.01	*Deania calcea*
SNE								0.02	*Simenchelys parasitica*
SNI		-0.03	-0.01					0.01	*Macroramphosus scolopax*
SNO						0.01		0.02	*Sio nordenskjoldii*
SNR				0.01		-0.03	0.03	0.00	*Deania histricosa*
SOL			0.02						Sole
SOM						0.02			*Somniosus rostratus*
SOP	0.02					-0.01		-0.03	*Somniosus pacificus*
SOR						-0.09		-0.01	*Neocyttus rhomboidalis*
SOT			0.02					0.03	*Solaster torulatus*
SPD	0.02	0.03	0.09	-0.18	0.02	-0.18	-0.03	-0.01	*Squalus acanthias*
SPE		0.06	-0.01	-0.25	0.20	-0.09		0.00	*Helicolenus* spp.
SPF								-0.01	*Pseudolabrus miles*

Appendix 19.09.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
SPI	0.01	-0.06	-0.13	0.00		-0.03	-0.01	-0.08	Spider crab
SPK			-0.01					-0.01	*Macrorhamphosodes uradoi*
SPL								0.00	*Scopelosaurus* sp.
SPO		0.03	-0.09	-0.04				-0.08	*Mustelus lenticulatus*
SPP								0.00	*Callanthias* spp.
SPR								0.00	*Sprattus antipodum, S. muelleri*
SPT			0.19					0.01	*Spatangus multispinus*
SPZ			-0.01					-0.05	*Genyagnus monopterygius*
SQA				0.01		0.02	0.03	0.03	*Squalus spp.*
SQI		-0.03							*Pristilepis oligolepis*
SQU	-0.01		0.03		-0.08	-0.11	-0.03	0.01	*Nototodarus sloanii & N. gouldi*
SQX	0.02		-0.06			0.00	-0.03	0.11	Squid
SRB	0.02							0.02	*Brama australis*
SRH			0.02			-0.01		0.13	*Hoplostethus mediterraneus*
SRI						0.01		0.03	*Scymnodon ringens*
SSC		-0.13	-0.04					0.01	*Leptomithrax australis*
SSH		0.03		-0.03				0.17	*Gollum attenuatus*
SSI	0.00	0.19	0.04		0.24	0.01	-0.04	0.04	*Argentina elongata*
SSK	0.00	-0.01	0.03	-0.15		-0.01	0.06	0.04	*Dipturus innominatus*
SSM						0.00	0.06	-0.01	*Alepocephalus antipodianus*
SSO						-0.19		0.00	*Pseudocyttus maculatus*
SSP								-0.02	*Pecten novaezelandiae*
STG		-0.01	0.03					-0.11	Stargazer
STM								0.02	*Tetrapturus audax*
STN		0.03	-0.02					0.06	*Thunnus maccoyii*
STO								0.01	*Stomias* spp.
STR		0.01	-0.05					-0.01	Stingray
STU	-0.02	-0.06			0.52			-0.10	*Allothunnus fallai*
SUH								-0.01	*Schedophilus huttoni*
SUN		-0.01	0.01		0.20	0.01		0.01	*Mola mola*
SUR			-0.06			0.00		-0.03	*Evechinus chloroticus*
SVA						0.05	0.08		*Solenosmilia variabilis*
SWA	0.06	0.05	-0.18		0.07	-0.03		-0.05	*Seriolella punctata*
SWO						-0.04		0.00	*Xiphias gladius*
SWR						-0.02		0.00	*Coris sandageri*
SYD								0.01	*Systellaspis debilis*
SYN			0.00			-0.03		0.01	Synaphobranchidae
TAM			0.06			0.04	0.08	0.23	*Echinothuriidae & Phormosomatidae*
TAY			0.10					0.04	*Typhlorke aysoni*
TDQ								0.04	*Taningia danae*
TFA			0.19						*Trichopeltarion fantasticum*
THR		-0.09			-0.02			-0.12	*Alopias vulpinus*
TLD								0.03	*Tetilla leptoderma*
TLO			0.01						*Telesto* spp.
TOA		0.10	-0.03	-0.03		0.05	0.00	0.08	*Neophrynichthys* sp.
TOD		0.05	0.03					0.06	*Neophrynichthys latus*
TOP	-0.02		0.03			0.00		0.13	*Ambophthalmos angustus*
TOR		0.06						0.14	*Thunnus orientalis*
TRA								-0.01	Trachichthyidae
TRE					0.06				*Pseudocaranx georgianus*
TRS						-0.02			*Trachyscorpia eschmeyeri*
TRU		0.00		0.00				-0.02	*Latris lineata*
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
TSQ						0.03		0.13	*Todarodes filippovae*
TTA			0.03						*Typhlonarke tarakea*

Appendix 19.09.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
TUR		0.02							*Colistium nudipinnis*
TVI								0.02	*Trachonurus villosus*
UFISH	-0.07	-0.22	-0.40	0.02		-0.31	-0.08	-0.37	Unidentified fish
URP		0.02	0.02						*Uroptychus* spp.
VCO						0.04	0.09	0.00	*Antimora rostrata*
VIT							-0.01		*Vitjazmaia latidactyla*
VNI								0.02	*Lucigadus nigromaculatus*
VOL			0.00					0.01	Volutidae
VSQ						0.03		0.19	*Histioteuthis* spp.
WAR		-0.01			-0.13	0.00		-0.18	*Seriolella brama*
WHE			0.01					0.02	Witch
WHR						-0.05		-0.05	*Trachyrincus longirostris*
WHX			0.01			0.07		0.18	*Trachyrincus aphyodes*
WIT	-0.01	0.11	0.10			0.04		0.12	*Arnoglossus scapha*
WOE						-0.05	-0.04		*Allocyttus verrucosus*
WPS		0.05				0.01		0.01	*Carcharodon carcharias*
WRA								0.03	*Dasyatis thetidis*
WSE									Labridae
WSQ	-0.02	0.09	0.02			0.03	0.15	-0.03	*Onykia* spp.
WWA	-0.04	0.05	-0.05	-0.04		0.00	0.03	0.06	*Seriolella caerulea*
YBF								0.02	*Rhombosolea leporina*
YBO			0.14					0.12	*Pentaceros decacanthus*
YCO		0.08							*Parapercis gilliesi*
YEM		-0.02							*Aldrichetta forsteri*
YFN		0.00						0.00	*Thunnus albacares*
YSG			0.01						*Pterygotrigla pauli*
YSP			0.02						*Yaldwynopsis spinima*
ZAS						0.03			*Zameus squamulosus*
ZOR			0.14					0.09	*Zoroaster* spp.

Appendix 19.09.2: BYCATCH: Total annual bycatch by fishery area for seven deepwater trawl fisheries and one longline fishery (1990–91 to 2016–17). Where data have not yet been updated for recent years for a fishery, figures from the last available year have been assumed, in order for annual totals to be calculated. LLL = ling longline fishery; HHL = hoki/hake/ling fishery. [Continued on next pages]

									AUCKLAND ISLANDS
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	12	642	24	66	7	0	0	0	750
1992	1 318	300	24	60	0	0	0	0	1 701
1993	1 009	422	24	38	0	24	48	0	1 564
1994	882	497	0	158	0	9	585	0	2 132
1995	866	189	21	127	0	64	2 079	0	3 346
1996	853	5 445	36	166	57	17	996	0	7 570
1997	1 954	1 641	51	198	17	11	503	0	4 374
1998	893	209	15	550	0	18	554	0	2 240
1999	1 040	276	21	255	0	33	1 042	0	2 667
2000	1 084	416	65	130	0	61	290	0	2 046
2001	751	353	64	304	0	12	277	0	1 762
2002	1 191	362	2	546	0	15	62	0	2 177
2003	1 225	1 326	0	501	0	23	20	0	3 095
2004	1 063	2 877	27	262	0	10	47	0	4 286
2005	1 035	2 010	1	187	0	0	16	0	3 249
2006	576	3 283	0	36	0	3	0	0	3 898
2007	745	1 105	0	39	0	3	0	0	1 892
2008	597	1 349	0	130	0	13	20	0	2 109

2009	876	1 914	0	109	0	18	20	0	2 937
2010	631	1 790	24	29	0	11	53	0	2 538
2011	705	2 574	111	147	0	40	17	0	3 594
2012	604	1 675	10	43	0	7	3	0	2 343
2013	586	1 688	10	56	0	0	0	0	2 340
2014	609	751	10	485	0	54	0	0	1 909
2015	329	713	10	514	0	13	50	0	1 629
2016	519	989	10	207	0	13	50	0	1 788
2017	519	989	10	342	0	13	50	0	1 923

									CHATHAM RISE
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	344	1 919	1 831	8 349	1 111	1 430	15 909	0	30 893
1992	4 343	4 000	1 831	8 993	72	356	15 677	0	35 272
1993	5 603	1 781	1 831	6 467	549	592	6 877	0	23 700
1994	2 644	5 273	1 795	3 974	1 019	499	15 043	0	30 247
1995	4 422	2 954	1 870	7 551	1 855	145	12 057	0	30 854
1996	3 865	5 328	1 970	11 340	272	781	8 664	0	32 221
1997	1 191	4 350	1 881	15 042	1 472	444	7 808	0	32 188
1998	585	3 537	2 392	16 757	1 515	1 137	5 685	0	31 609
1999	777	4 819	1 020	17 707	1 944	1 744	2 913	0	30 925
2000	1 548	3 798	1 277	19 485	2 603	414	2 763	0	31 887
2001	1 987	10 277	1 240	16 745	106	236	6 016	0	36 606
2002	1 798	8 014	2 781	16 872	698	153	2 917	0	33 232
2003	1 581	6 876	1 430	20 002	1 549	111	3 265	0	34 814
2004	1 006	2 096	1 239	19 822	168	348	3 873	0	28 552
2005	2 658	4 642	1 518	13 797	73	441	3 134	0	26 263
2006	1 111	6 886	1 576	12 470	5 771	203	2 870	0	30 887
2007	2 458	4 863	849	9 970	1 476	515	2 944	0	23 075
2008	1 635	2 241	1 364	11 551	1 859	199	2 908	0	21 757
2009	1 209	511	1 366	8 766	685	312	3 204	0	16 053
2010	1 572	984	1 587	14 552	677	560	2 854	0	22 786
2011	1 150	2 282	1 751	8 130	528	134	458	0	14 433
2012	1 538	979	1 041	8 772	1 139	316	615	0	14 400
2013	1 953	221	1 041	9 879	1 885	260	314	0	15 553
2014	2 737	72	1 041	15 554	2 378	141	604	0	22 528
2015	2 261	778	1 041	24 659	2 378	340	553	0	32 010
2016	1 300	2 133	1 041	17 227	2 378	340	553	0	24 972
2017	1 300	2 133	1 041	18 530	2 378	340	553	0	26 275

									COOK STRAIT
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	0	0	9	1 501	4	9	1 059	0	2 582
1992	0	0	9	1 068	9	7	1 089	0	2 182
1993	0	58	9	1 090	0	2	469	0	1 628
1994	0	0	5	2 621	0	12	878	0	3 517
1995	33	1	9	2 400	0	0	1 060	0	3 504
1996	0	19	8	3 274	0	3	373	0	3 677
1997	0	0	3	4 525	0	8	396	0	4 933
1998	0	1	20	2 249	113	9	221	0	2 612
1999	2	90	56	2 035	423	3	181	0	2 791
2000	2	2	67	3 244	155	12	132	0	3 614
2001	0	47	221	1 561	312	12	72	0	2 225
2002	53	0	40	882	26	10	0	0	1 012
2003	7	2	22	3 197	0	6	0	0	3 234
2004	0	0	54	1 396	0	1	0	0	1 452
2005	36	0	103	1 825	0	2	0	0	1 965

2006	0	0	128	507	0	4	0	0	639
2007	1	2	126	1 070	0	1	0	0	1 200
2008	4	0	201	790	0	0	30	0	1 025
2009	0	0	29	660	0	1	0	0	690
2010	0	0	21	700	0	0	0	0	721
2011	0	0	28	325	0	0	20	0	373
2012	0	0	11	532	0	0	0	0	543
2013	0	0	11	645	0	0	0	0	656
2014	0	0	11	706	0	0	0	0	717
2015	0	0	11	650	0	0	23	0	684
2016	0	2	11	239	0	0	23	0	275
2017	0	2	11	231	0	0	23	0	267

									EAST COAST NORTH ISLAND
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	2 390	0	69	725	12	22	1 438	0	4 655
1992	916	0	69	540	9	22	992	0	2 548
1993	1 466	0	69	561	16	15	509	0	2 636
1994	1 233	0	196	1 563	10	12	732	0	3 747
1995	1 255	0	96	1 137	0	1	1 619	0	4 109
1996	1 508	0	111	2 192	0	34	446	0	4 292
1997	720	0	108	3 402	4	73	1 092	0	5 400
1998	883	0	160	2 960	0	105	3 514	0	7 623
1999	1 043	0	95	1 387	0	31	804	0	3 359
2000	956	2	161	1 146	0	73	1 560	0	3 898
2001	1 365	0	398	1 313	45	45	535	0	3 700
2002	3 733	0	204	736	53	26	65	0	4 816
2003	1 282	10	83	1 035	0	50	182	0	2 642
2004	795	0	247	1 031	0	11	0	0	2 084
2005	845	0	252	680	0	10	168	0	1 956
2006	336	0	399	386	0	8	0	0	1 129
2007	591	0	159	543	0	2	0	0	1 294
2008	391	0	84	401	0	1	142	0	1 019
2009	236	0	283	442	0	1	142	0	1 105
2010	619	0	258	421	0	1	285	0	1 584
2011	441	0	269	730	0	1	169	0	1 610
2012	251	0	229	458	0	0	93	0	1 031
2013	235	0	229	507	0	0	82	0	1 052
2014	321	0	229	998	0	0	0	0	1 548
2015	332	0	229	758	0	0	54	0	1 373
2016	234	0	229	570	0	0	54	0	1 087
2017	234	0	229	697	0	0	54	0	1 214

									NORTHLAND
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	1 422	0	75	24	0	0	70	0	1 591
1992	919	36	75	120	4	0	12	0	1 166
1993	667	13	75	104	22	0	19	0	901
1994	456	0	152	83	6	0	50	0	747
1995	1 208	0	81	303	0	0	42	0	1 635
1996	803	0	58	393	0	0	450	0	1 705
1997	584	8	67	1 205	0	0	3 240	0	5 104
1998	392	0	109	959	0	0	569	0	2 030
1999	230	0	29	359	8	0	93	0	718
2000	482	0	24	318	0	0	132	0	956
2001	603	6	70	200	179	0	30	0	1 087

2002	1 085	0	14	270	0	0	14	0	1 383
2003	811	12	29	189	0	0	41	0	1 082
2004	746	8	18	243	0	0	15	0	1 030
2005	948	1	64	155	0	0	23	0	1 191
2006	347	11	37	275	0	0	19	0	689
2007	580	11	36	232	0	0	13	0	872
2008	445	2	46	216	0	0	15	0	724
2009	255	5	38	183	0	0	26	0	507
2010	576	0	38	396	0	0	5	0	1 015
2011	390	0	63	277	0	0	40	0	770
2012	394	0	44	244	0	0	36	0	718
2013	560	9	44	383	0	0	16	0	1 012
2014	566	0	44	216	0	0	26	0	852
2015	479	0	44	302	0	0	16	0	841
2016	318	8	44	404	0	0	16	0	790
2017	318	8	44	395	0	0	16	0	781

									PUYSEGUR
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	0	6	30	318	0	82	429	0	865
1992	4	118	30	418	143	53	1 689	0	2 456
1993	35	145	30	144	10	78	909	0	1 352
1994	1	12	60	246	14	59	2 343	0	2 736
1995	8	29	39	414	5	26	444	0	964
1996	0	32	59	153	0	107	921	0	1 272
1997	28	125	103	515	4	35	493	0	1 302
1998	37	38	219	535	0	417	333	0	1 580
1999	212	58	123	658	8	27	53	0	1 138
2000	40	74	93	1 050	0	200	27	0	1 484
2001	148	543	217	2 887	0	134	503	0	4 432
2002	0	887	99	660	0	124	193	0	1 964
2003	165	2 708	81	637	0	252	27	0	3 870
2004	21	487	62	328	0	45	0	0	943
2005	8	497	77	368	0	39	112	0	1 101
2006	0	538	87	971	0	26	131	0	1 753
2007	0	33	71	236	0	7	0	0	347
2008	0	32	190	206	0	64	0	0	492
2009	0	13	25	160	0	5	10	0	214
2010	0	63	5	120	0	1	14	0	204
2011	0	152	95	275	0	7	0	0	529
2012	0	47	21	302	0	3	15	0	388
2013	0	66	21	513	0	11	0	0	611
2014	0	127	21	1 609	0	75	42	0	1 874
2015	0	90	21	1 170	0	3	68	0	1 352
2016	0	107	21	1 702	0	3	68	0	1 901
2017	0	107	21	1 269	0	3	68	0	1 468

									STEWART-SNARES SHELF
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	0	14 215	28	2 108	65	55	68	0	16 539
1992	1	11 222	28	2 592	536	40	122	0	14 542
1993	11	14 924	28	1 707	114	14	102	0	16 901
1994	0	4 669	85	859	159	16	126	0	5 914
1995	5	1 309	205	1 250	298	8	10	0	3 085
1996	2	5 029	37	1 752	204	49	231	0	7 304
1997	10	7 993	143	1 195	257	124	880	0	10 600
1998	19	5 719	219	3 464	506	114	692	0	10 733

1999	69	14 713	508	2 499	1 559	288	154	0	19 790
2000	8	7 685	198	4 860	4 621	184	90	0	17 646
2001	3	8 536	165	6 069	877	66	165	0	15 881
2002	0	15 158	164	7 148	1 567	248	2	0	24 288
2003	156	15 326	75	2 336	2 063	158	0	0	20 115
2004	40	19 606	166	2 833	723	34	0	0	23 401
2005	6	20 599	112	1 876	527	48	0	0	23 168
2006	0	28 634	48	2 067	1 369	62	0	0	32 180
2007	0	10 859	343	3 883	360	8	0	0	15 454
2008	0	10 356	179	1 866	183	9	5	0	12 598
2009	0	9 670	91	1 673	996	15	0	0	12 445
2010	0	9 100	54	2 211	1 144	27	0	0	12 536
2011	0	13 547	60	2 092	1 081	23	0	0	16 803
2012	0	10 295	514	1 495	1 622	42	0	0	13 968
2013	0	9 000	514	2 269	935	64	0	0	12 782
2014	0	8 482	514	9030	1 135	31	0	0	19 192
2015	0	7 126	514	9369	1 135	1	0	0	18 145
2016	0	5 625	514	4196	1 135	1	0	0	11 471
2017	0	5 625	514	7029	1 135	1	0	0	14 304

									SUBANTARCTIC
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	1	38	1 078	433	0	3	33	533	2 120
1992	14	0	1 078	226	0	2	40	1 479	2 839
1993	95	5	1 078	98	0	4	35	206	1 521
1994	85	6	705	139	0	12	31	382	1 360
1995	3	0	549	95	0	26	76	178	928
1996	8	20	641	106	0	49	538	63	1 426
1997	154	0	637	72	0	44	2 986	203	4 096
1998	66	0	132	306	0	152	1 762	296	2 714
1999	27	0	759	166	0	124	231	283	1 590
2000	9	15	905	442	0	312	147	283	2 113
2001	4	21	912	415	0	59	104	223	1 739
2002	53	16	537	1 223	0	63	0	364	2 256
2003	41	486	609	2 003	0	596	133	230	4 098
2004	58	548	882	1 901	0	69	74	390	3 922
2005	0	103	212	449	0	101	75	250	1 190
2006	0	110	139	86	0	301	172	190	997
2007	0	194	109	197	0	57	66	40	663
2008	0	13	575	820	0	85	49	40	1 582
2009	0	2	474	514	0	126	52	40	1 208
2010	0	9	274	83	0	145	61	40	612
2011	0	18	149	335	0	54	32	40	628
2012	0	5	139	95	0	37	5	40	321
2013	0	24	139	133	0	0	0	40	336
2014	18	4	139	1 214	0	0	0	40	1 415
2015	35	7	139	723	0	0	0	40	944
2016	5	4	139	238	0	0	0	40	426
2017	5	4	139	730	0	0	0	40	918

									WEST COAST NORTH ISLAND
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	0	2	2	27	1 839	0	15	0	1 885
1992	4	4	2	28	6 978	0	160	0	7 175
1993	64	9	2	15	8 761	0	474	0	9 324
1994	0	23	2	14	4 924	2	2 397	0	7 362
1995	3	2	0	21	1 791	1	116	0	1 933

1996	0	0	18	100	3 621	0	33	0	3 771
1997	1	2	79	33	1 814	3	200	0	2 132
1998	13	0	1	45	5 784	0	268	0	6 111
1999	11	168	36	24	2 010	0	210	0	2 460
2000	5	553	4	61	360	0	158	0	1 140
2001	0	1 155	16	89	1 826	2	489	0	3 577
2002	10	836	5	48	5 693	0	40	0	6 632
2003	0	89	0	110	7 366	0	72	0	7 637
2004	0	6	1	43	13 310	0	61	0	13 421
2005	8	0	1	20	7 292	0	43	0	7 364
2006	0	0	5	61	10 312	0	51	0	10 429
2007	0	1	36	33	11 015	0	71	0	11 156
2008	0	0	29	94	8 975	0	41	0	9 139
2009	0	0	36	77	6 891	0	39	0	7 043
2010	0	0	25	24	10 760	0	29	0	10 838
2011	0	0	56	26	4 484	0	23	0	4 589
2012	0	0	14	39	5 674	0	60	0	5 788
2013	4	2	14	51	5 696	0	49	0	5 816
2014	0	0	14	80	5 281	0	64	0	5 440
2015	0	0	14	70	5 281	0	30	0	5 396
2016	0	0	14	42	5 281	0	30	0	5 368
2017	0	0	14	99	5 281	0	30	0	5 425

									WEST COAST SOUTH ISLAND
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	1	36	164	8 589	2 206	0	516	0	11 513
1992	0	23	164	6 335	2 864	0	1 082	0	10 468
1993	95	2	164	3 431	800	0	147	0	4 640
1994	2	5	320	9 426	2 129	9	213	0	12 103
1995	68	1	223	13 187	3 253	0	509	0	17 240
1996	12	0	401	12 211	3 050	2	370	0	16 045
1997	7	2	480	11 341	3 535	5	419	0	15 789
1998	26	2	406	10 964	4 376	24	276	0	16 074
1999	38	5	292	7 028	9 536	8	426	0	17 333
2000	15	0	281	7 168	3 906	14	654	0	12 037
2001	0	38	463	8 222	9 191	2	161	0	18 076
2002	15	5	221	7 245	3 903	0	0	0	11 389
2003	8	1 590	282	6 077	1 902	0	0	0	9 859
2004	0	369	158	6 830	706	0	0	0	8 063
2005	8	764	310	4 554	452	0	7	0	6 095
2006	0	185	255	4 051	1 271	0	0	0	5 762
2007	0	418	264	2 557	3 304	0	0	0	6 543
2008	0	0	364	3 083	1 540	0	0	0	4 987
2009	0	37	370	2 750	1 287	0	18	0	4 462
2010	0	41	386	1 696	506	0	4	0	2 633
2011	0	0	332	3 439	524	0	14	0	4 308
2012	14	0	428	2 755	588	0	23	0	3 808
2013	21	0	428	3 827	889	0	41	0	5 206
2014	31	0	428	5 864	1 003	0	11	0	7 337
2015	34	450	428	7 567	1 003	0	148	0	9 630
2016	39	0	428	6 303	1 003	0	148	0	7 921
2017	39	0	428	5 410	1 003	0	148	0	7 028

Appendix 19.09.3: DISCARDS: Total annual discards by fishery area for seven deepwater trawl fisheries and one longline fishery (1990–91 to 2015–16). Where data have not yet been updated for recent years for a fishery figures from the last available year have been assumed, in order for annual totals to be calculated. LLL = ling longline fishery; HHL = hoki/hake/ling fishery. [Continued on next pages]

									AUCKLAND ISLANDS
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	4	74	17	5	0	0	0	0	100
1992	90	18	17	5	0	0	0	0	130
1993	278	68	17	9	0	9	2	0	383
1994	539	37	0	12	0	1	12	0	601
1995	707	32	12	10	0	47	20	0	829
1996	716	55	18	13	6	5	17	0	830
1997	1 383	183	34	15	2	6	10	0	1 634
1998	517	332	8	42	0	4	14	0	917
1999	645	101	13	19	0	27	15	0	820
2000	790	55	52	47	0	18	8	0	970
2001	421	115	58	69	0	5	6	0	674
2002	610	191	1	41	0	4	14	0	861
2003	832	1 162	0	205	0	7	7	0	2 046
2004	270	1 741	12	71	0	3	5	0	2 052
2005	508	1 234	1	57	0	0	1	0	1 758
2006	414	2 574	0	29	0	0	0	0	2 991
2007	448	812	0	30	0	0	0	0	1 263
2008	316	538	0	3	0	0	1	0	858
2009	412	1 331	0	3	0	1	0	0	1 744
2010	520	1 079	12	3	0	0	0	0	1 614
2011	545	1 386	66	3	0	0	1	0	1 999
2012	453	662	6	3	0	0	0	0	1 124
2013	439	804	6	4	0	0	0	0	1 253
2014	464	259	6	9	0	1	2	0	740
2015	250	389	6	20	0	0	2	0	667
2016	421	1 130	6	6	0	0	2	0	1 565
2017	421	1 130	6	22	0	0	2	0	1 581

									CHATHAM RISE
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	154	241	605	3 239	861	454	2 419	0	7 973
1992	1 364	303	605	6 387	100	201	1 053	0	10 015
1993	2 314	610	605	2 028	360	240	1 186	0	7 344
1994	1 402	1 718	862	1 650	48	302	937	0	6 919
1995	809	453	1 019	2 817	278	184	946	0	6 505
1996	4 126	110	1 107	3 737	24	423	671	0	10 199
1997	783	889	765	3 066	68	372	720	0	6 662
1998	657	1 436	1661	8 244	1 845	364	840	0	15 046
1999	361	990	587	5 314	110	272	1 020	0	8 654
2000	271	825	690	9 829	125	297	787	0	12 825
2001	1 366	2 721	874	6 981	18	151	1 012	0	13 123
2002	1 303	1 698	1 709	9 545	77	48	471	0	14 850
2003	1 056	7 324	1 027	14 762	80	68	478	0	18 598
2004	218	902	806	9 658	5	99	538	0	5 750
2005	2 117	2 156	1 080	3 348	5	95	520	0	6 843
2006	898	2 681	955	4 042	271	93	1 399	0	7 260
2007	2 249	2 392	546	3 089	52	46	491	0	7 590
2008	1 417	864	1 016	7 160	47	28	129	0	5 064
2009	1 041	327	875	12 681	11	26	195	0	5 085
2010	1 418	419	950	3 848	3	27	76	0	9 442
2011	1 002	1 083	879	5 611	3	25	29	0	5 706

2012	1 138	395	517	3 039	4	61	115	0	3 522
2013	1 743	105	517	1 718	32	9	71	0	5 105
2014	2 245	29	517	1 801	3	30	18	0	4 643
2015	1 718	478	517	2 776	3	7	32	0	5 532
2016	1 262	1 859	517	984	3	7	32	0	4 665
2017	1 262	1 859	517	924	3	7	32	0	4 605

									COOK STRAIT
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	0	0	3	1 081	0	3	76	0	1 164
1992	0	0	3	769	0	1	84	0	857
1993	0	4	3	786	0	2	56	0	852
1994	0	0	2	917	0	1	71	0	991
1995	9	0	3	1 729	0	2	84	0	1 826
1996	0	0	2	3 896	0	1	24	0	3 923
1997	0	0	1	3 259	0	6	32	0	3 299
1998	0	0	6	2 033	0	3	15	0	2 057
1999	3	9	16	1 871	31	4	35	0	1 968
2000	1	0	21	2 724	16	3	16	0	2 780
2001	0	7	65	1 312	18	6	6	0	1 414
2002	36	0	13	718	0	5	5	0	778
2003	3	2	4	1 982	0	3	3	0	2 585
2004	0	0	15	1 330	0	0	2	0	994
2005	28	0	31	651	0	0	3	0	656
2006	1	0	36	659	0	2	7	0	253
2007	1	0	48	652	0	0	3	0	903
2008	2	0	35	838	0	0	3	0	617
2009	0	0	8	943	0	0	4	0	522
2010	0	0	7	333	0	0	2	0	376
2011	0	0	8	690	0	0	3	0	758
2012	0	0	5	643	0	0	2	0	347
2013	0	0	5	346	0	0	1	0	184
2014	0	0	5	550	0	0	2	0	557
2015	0	0	5	213	0	0	4	0	222
2016	0	2	5	366	0	0	4	0	377
2017	0	2	5	303	0	0	4	0	314

									EAST COAST NORTH ISLAND
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	1 535	0	24	178	0	9	664	0	2 411
1992	696	0	24	131	0	4	692	0	1 548
1993	579	0	24	130	0	7	555	0	1 295
1994	540	0	41	351	3	1	689	0	1 625
1995	418	0	27	316	0	13	893	0	1 666
1996	1 083	0	30	582	0	13	969	0	2 677
1997	405	0	31	726	0	57	873	0	2 092
1998	624	0	42	3	0	32	669	0	1 369
1999	1 240	0	24	218	0	36	627	0	2 145
2000	348	0	43	181	0	20	275	0	866
2001	702	0	114	228	3	23	1 093	0	2 163
2002	2 284	0	59	174	6	13	31	0	2 567
2003	699	11	23	71	0	23	0	0	979
2004	161	0	58	57	0	4	5	0	344
2005	585	0	64	26	0	4	7	0	700
2006	247	0	96	24	0	3	56	0	453
2007	470	0	66	32	0	1	59	0	655
2008	289	0	84	23	0	0	59	0	470

2009	186	0	78	40	0	0	83	0	396
2010	601	0	81	33	0	0	100	0	870
2011	423	0	73	55	0	0	22	0	665
2012	209	0	79	39	0	0	32	0	376
2013	238	0	79	37	0	0	57	0	478
2014	301	0	79	71	0	0	37	0	488
2015	285	0	79	11	0	0	13	0	388
2016	255	0	79	70	0	0	13	0	417
2017	255	0	79	104	0	0	13	0	451

									NORTHLAND
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	615	0	23	13	0	0	5	0	657
1992	405	0	23	91	0	0	1	0	520
1993	303	1	23	127	0	0	2	0	457
1994	231	0	44	53	0	0	4	0	332
1995	386	0	23	191	0	0	3	0	604
1996	732	0	18	215	0	0	8	0	974
1997	431	1	22	520	0	0	40	0	1 015
1998	214	0	33	2	0	0	37	0	286
1999	137	0	7	144	0	1	118	0	406
2000	262	0	8	117	0	0	16	0	404
2001	383	1	17	70	9	0	2	0	482
2002	832	0	4	124	0	0	0	0	961
2003	367	35	9	70	0	0	4	0	496
2004	235	2	5	72	0	0	1	0	292
2005	746	0	12	32	0	0	1	0	774
2006	281	4	13	86	0	0	0	0	358
2007	507	5	12	63	0	0	1	0	568
2008	355	1	24	55	0	0	1	0	412
2009	231	2	11	67	0	0	4	0	282
2010	505	0	13	71	0	0	0	0	659
2011	335	0	19	93	0	0	1	0	448
2012	268	0	23	98	0	0	6	0	359
2013	505	3	23	116	0	0	9	0	674
2014	454	0	23	104	0	0	7	0	588
2015	342	0	23	75	0	0	3	0	444
2016	321	6	23	138	0	0	3	0	492
2017	321	6	23	158	0	0	3	0	512

									PUYSEGUR
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	0	1	21	225	0	26	34	0	306
1992	1	17	21	512	0	34	139	0	724
1993	13	31	21	208	0	115	118	0	505
1994	0	5	27	248	3	32	268	0	582
1995	2	5	22	227	0	63	38	0	358
1996	0	1	34	567	0	75	117	0	794
1997	21	25	60	429	0	100	34	0	669
1998	27	24	122	293	0	90	8	0	564
1999	196	16	66	270	0	130	5	0	683
2000	15	12	42	307	0	109	3	0	488
2001	84	162	106	253	0	167	76	0	848
2002	0	282	60	297	0	65	32	0	735
2003	32	3 204	45	784	0	172	18	0	3 667
2004	4	122	34	276	0	15	9	0	222
2005	5	196	39	681	0	15	4	0	554

2006	0	147	54	2 074	0	11	1	0	401
2007	0	12	45	484	0	3	0	0	189
2008	0	10	116	622	0	11	1	0	252
2009	0	4	14	152	0	2	2	0	110
2010	0	19	4	267	0	1	2	0	92
2011	0	49	57	199	0	3	3	0	263
2012	0	12	16	417	0	0	2	0	163
2013	0	8	16	378	0	0	0	0	241
2014	0	37	16	249	0	2	5	0	309
2015	0	34	16	275	0	0	0	0	326
2016	0	71	16	515	0	0	0	0	603
2017	0	71	16	460	0	0	0	0	548

									STEWART-SNARES SHELF
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	0	144	15	1 471	0	14	1	0	1 645
1992	0	83	15	1 531	1 405	21	2	0	3 056
1993	3	1 344	15	1 942	105	11	2	0	3 423
1994	0	99	39	585	13	7	3	0	745
1995	1	79	171	211	64	16	0	0	542
1996	3	4	15	459	502	28	4	0	1 014
1997	6	1 659	55	1 017	51	91	57	0	2 936
1998	14	1 826	96	1 167	0	50	10	0	3 163
1999	64	1 335	132	1 430	40	174	6	0	3 180
2000	2	162	55	1 114	612	140	2	0	2 088
2001	2	928	58	1 852	39	69	3	0	2 950
2002	0	3 145	46	2 221	277	146	1	0	5 836
2003	44	3 383	34	2 853	25	141	0	0	4 661
2004	9	2 858	58	2 528	4	24	0	0	3 452
2005	3	3 277	70	2 179	7	23	0	0	3 768
2006	0	4 713	18	2 008	10	37	1	0	6 077
2007	0	1 892	218	4 774	3	5	0	0	2 553
2008	0	1 096	74	594	1	5	0	0	1 258
2009	0	1 816	34	1 384	7	2	0	0	1 995
2010	0	1 656	23	512	2	4	0	0	1 962
2011	0	2 690	23	724	2	4	0	0	3 526
2012	0	1 513	252	1 019	3	4	0	0	2 362
2013	0	1 544	252	1 288	32	6	0	0	2 449
2014	0	1 152	252	1 525	17	6	0	0	2 952
2015	0	961	252	1 987	17	0	0	0	3 218
2016	0	1 284	252	876	17	0	0	0	2 430
2017	0	1 284	252	2 006	17	0	0	0	3 560

									SUBANTARCTIC
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	1	0	317	364	0	1	3	746	1 431
1992	2	0	317	114	0	1	3	1 218	1 656
1993	47	0	317	40	0	2	5	537	948
1994	34	1	235	33	0	4	3	483	793
1995	1	0	318	22	0	38	7	303	689
1996	16	0	346	26	0	20	40	406	854
1997	125	0	427	17	0	22	97	270	958
1998	38	0	10	73	0	55	34	392	602
1999	19	0	471	54	0	46	25	471	1 086
2000	2	0	537	278	0	155	10	471	1 454
2001	2	2	557	147	0	230	10	137	1 085
2002	37	3	296	655	0	38	7	159	1 195

2003	19	497	267	1 002	0	91	14	250	1 661
2004	13	129	329	773	0	58	19	90	847
2005	0	43	136	765	0	78	5	160	530
2006	0	42	66	221	0	121	27	240	518
2007	0	96	53	949	0	51	4	120	371
2008	0	2	62	115	0	32	4	120	343
2009	0	1	151	222	0	33	6	120	322
2010	0	5	115	47	0	36	3	120	299
2011	0	6	20	23	0	12	2	120	175
2012	0	2	86	90	0	9	0	120	218
2013	0	6	86	67	0	0	0	120	224
2014	14	1	86	121	0	0	0	120	342
2015	25	0	86	102	0	0	0	120	333
2016	4	1	86	9	0	0	0	120	220
2017	4	1	86	223	0	0	0	120	434

									WEST COAST NORTH ISLAND
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	0	0	0	14	0	0	5	0	19
1992	1	0	0	22	100	0	27	0	150
1993	24	1	0	18	915	0	333	0	1 291
1994	0	4	1	9	1 285	0	141	0	1 439
1995	1	0	0	14	317	4	161	0	497
1996	0	0	10	55	109	0	45	0	220
1997	0	1	32	14	0	2	106	0	155
1998	8	0	1	0	0	0	43	0	53
1999	10	18	17	9	837	0	62	0	953
2000	1	26	2	22	16	0	129	0	196
2001	0	148	6	31	51	1	246	0	483
2002	7	181	3	22	230	0	7	0	450
2003	0	140	0	13	183	0	6	0	376
2004	0	2	0	10	133	0	7	0	151
2005	5	0	1	3	98	0	15	0	121
2006	0	0	2	30	77	0	11	0	103
2007	0	1	19	19	165	0	2	0	193
2008	0	0	15	32	166	0	13	0	207
2009	0	0	17	32	97	0	3	0	131
2010	0	0	11	4	264	0	6	0	290
2011	0	0	23	8	76	0	4	0	112
2012	0	0	9	10	140	0	26	0	184
2013	3	1	9	18	113	0	21	0	164
2014	0	0	9	14	165	0	60	0	247
2015	0	0	9	8	165	0	10	0	192
2016	0	0	9	12	165	0	10	0	196
2017	0	0	9	26	165	0	10	0	210

									WEST COAST SOUTH ISLAND
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	1	0	115	5 904	76	0	195	0	6 291
1992	0	0	115	3 382	201	0	223	0	3 921
1993	36	0	115	4 889	495	0	192	0	5 727
1994	1	1	139	6 671	118	1	422	0	7 353
1995	18	0	127	8 327	461	4	147	0	9 084
1996	12	0	220	7 314	363	1	102	0	8 013
1997	5	1	270	7 520	964	3	181	0	8 946
1998	19	1	219	3 587	0	7	145	0	3 979

1999	35	1	223	3 177	520	8	290	0	4 254
2000	6	0	155	2 035	16	4	301	0	2 517
2001	0	6	231	2 421	482	1	71	0	3 211
2002	9	1	128	2 926	100	0	2	0	3 166
2003	3	212	172	3 250	7	0	1	0	3 158
2004	0	9	107	2 882	1	0	1	0	1 773
2005	6	29	165	1 509	3	0	0	0	668
2006	0	5	153	2 676	29	0	7	0	1 608
2007	0	14	157	3 740	84	0	1	0	780
2008	0	0	236	2 167	7	0	0	0	869
2009	0	1	204	2 402	5	0	0	0	689
2010	0	1	234	659	4	0	0	0	719
2011	0	0	183	2 302	1	0	2	0	1 231
2012	9	0	209	3 606	5	0	3	0	1 256
2013	14	0	209	2 727	7	0	6	0	2 319
2014	19	0	209	2 491	9	0	10	0	2 739
2015	21	20	209	2 681	9	0	48	0	2 989
2016	28	0	209	2 106	9	0	48	0	2 401
2017	28	0	209	2 802	9	0	48	0	3 097

19.10 APPENDICES FROM CHAPTER 10 CHONDRICHTHYANS

Appendix 19.10.1: List of New Zealand chondrichthyans, with details of their fisheries management classification, and IUCN and Department of Conservation threat classes. IUCN threat classes: EN, Endangered; VU, Vulnerable; NT, Near Threatened; LC, Least Concern; DD, Data Deficient. DOC threat classes: DD, Data Deficient; NE, Nationally Endangered; NV, Threatened – Nationally Vulnerable; ARNU, At Risk – Naturally Uncommon; MI, Migrant; VA, Vagrant; NOT, Not Threatened. DOC qualifiers: CD, Conservation Dependent; DP, Data Poor; Inc, Increasing; SO, Secure Overseas; S?O, Uncertain Whether Secure Overseas; TO, Threatened Overseas; T?O, Uncertain Whether Threatened Overseas. Sources: IUCN Redlist classes as at July 2013 (L. Harrison, Shark Specialist Group IUCN, pers. comm.); DOC threat classes 2018 (Duffy et al. 2018). [Continued on next page]. NB: IUCN Redlist classes for New Zealand chondrichthyans were reviewed in 2017 and the classifications were changed for some species; the updated classifications are expected to be published in late 2018.

Group	Family	Species	Common name	Code	Manage-ment class	IUCN redlist class	DoC threat class	DoC qualifier
Chimaera	Callorhinchidae	*Callorhinchus milii* Bory de St Vincent, 1823	Elephantfish	ELE	QMS	LC	NOT	CD,Inc
Chimaera	Rhinochimaeridae	*Harriotta haeckeli* Karrer, 1972	Smallspine spookfish	HHA	Non-QMS	LC	NOT	
Chimaera	Rhinochimaeridae	*Harriotta raleighana* Goode & Bean, 1895	Longnose spookfish	LCH	Non-QMS	LC	NOT	
Chimaera	Rhinochimaeridae	*Rhinochimaera pacifica* (Mitsukuri, 1895)	Pacific spookfish	RCH	Non-QMS	LC	NOT	DP
Chimaera	Chimaeridae	*Chimaera carophila* Kemper, Ebert, Naylor & Didier 2014	Brown chimaera, longspine chimaera	CHP	Non-QMS		NOT	
Chimaera	Chimaeridae	*Chimaera lignaria* Didier, 2002	Purple chimaera, giant chimaera	CHG	Non-QMS	LC	NOT	SO
Chimaera	Chimaeridae	*Chimaera panthera* Didier, 1998	Leopard chimaera	CPN	Non-QMS	DD	NOT	DP
Chimaera	Chimaeridae	*Hydrolagus bemisi* Didier, 2002	Pale ghost shark	GSP	QMS	LC	NOT	CD
Chimaera	Chimaeridae	*Hydrolagus homonycteris* Didier 2008	Black ghost shark	HYB	Non-QMS	LC	NOT	SO
Chimaera	Chimaeridae	*Hydrolagus novaezealandiae* (Fowler, 1911)	Dark ghost shark	GSH	QMS	LC	NOT	
Chimaera	Chimaeridae	*Hydrolagus trolli* Didier and Seret, 2002	Pointynose blue ghost shark	HYP	Non-QMS	LC	NOT	SO
Chimaera	Chimaeridae	*Hydrolagus* cf *affinis* (de Brito Capello 1868)	Giant black ghost shark	HGB	Non-QMS		DD	CD
Shark	Chlamydoselachidae	*Chlamydoselachus anguineus* Garman, 1884	Frill shark	FRS	Non-QMS	LC	ARNU	DP,SO
Shark	Hexanchidae	*Heptranchias perlo* (Bonnaterre, 1788)	Sharpnose sevengill shark	HEP	Non-target	NT	ARNU	DP,SO
Shark	Hexanchidae	*Hexanchus griseus* (Bonnaterre, 1788)	Sixgill shark	HEX	Non-QMS	NT	NOT	DP,SO
Shark	Hexanchidae	*Notorynchus cepedianus* (Peron, 1807)	Broadnose sevengill shark	SEV	Non-QMS	DD	NOT	DP,SO
Shark	Echinorhinidae	*Echinorhinus brucus* (Bonnaterre, 1788)	Bramble shark	BRS	Non-QMS	DD	ARNU	DP,SO
Shark	Echinorhinidae	*Echinorhinus cookei* Pietschmann, 1928	Prickly shark	ECO	Non-QMS	NT	ARNU	DP,SO
Shark	Squalidae	*Cirrhigaleus australis* White, Last & Stevens, 2007	Southern mandarin dogfish	MSH	Non-QMS	DD	ARNU	DP,TO
Shark	Squalidae	*Squalus acanthias* Linnaeus, 1758	Spiny dogfish	SPD	QMS	VU	NOT	SO
Shark	Squalidae	*Squalus griffini* Phillipps, 1931	Northern spiny dogfish	NSD	Non-QMS	LC	NOT	SO
Shark	Squalidae	*Squalus raoulensis* Duffy & Last, 2007	Kermadec spiny dogfish		Non-QMS	LC	DD	
Shark	Squalidae	*Squalus* sp.	Shortspine dogfish		Non-QMS		DD	
Shark	Centrophoridae	*Centrophorus harrissoni* McCulloch, 1915	Harrisson's dogfish		Non-QMS	EN	DD	TO
Shark	Centrophoridae	*Centrophorus squamosus* (Bonnaterre, 1788)	Leafscale gulper shark	CSQ	Non-QMS	VU	NOT	SO
Shark	Centrophoridae	*Deania calcea* (Lowe, 1839)	Shovelnose dogfish	SND	Non-QMS	LC	NOT	
Shark	Centrophoridae	*Deania hystricosa* (Garman, 1906)	Rough longnose dogfish	SNR	Non-QMS	DD	DD	
Shark	Centrophoridae	*Deania quadrispinosa* (McCulloch, 1915)	Longsnout dogfish	DEQ	Non-QMS	NT	DD	SO
Shark	Etmopteridae	*Centroscyllium kamoharai* Abe 1966	Fragile dogfish		Non-QMS	DD	DD	
Shark	Etmopteridae	*Etmopterus granulosus* (Günther, 1880)	Baxter's dogfish	ETB	Non-QMS	LC	NOT	SO
Shark	Etmopteridae	*Etmopterus lucifer* Jordan & Snyder, 1902	Lucifer dogfish	ETL	Non-QMS	LC	NOT	DP,SO
Shark	Etmopteridae	*Etmopterus molleri* (Whitley, 1939)	Moller's lantern shark	EMO	Non-QMS	DD	DD	S?O
Shark	Etmopteridae	*Etmopterus pusillus* (Lowe, 1839)	Smooth lantern shark	ETP	Non-QMS	LC	ARNU	DP,SO
Shark	Etmopteridae	*Etmopterus unicolor* (Engelhardt 1912)	Bristled lantern shark	ETU	Non-QMS	DD	NOT	SO
Shark	Etmopteridae	*Etmopterus viator* Straube 2011	Blue-eye lantern shark	EVI	Non-QMS		DD	
Shark	Somniosidae	*Centroscymnus coelolepis* Bocage & Capello, 1864	Portuguese dogfish	CYL	Non-QMS	NT	NOT	DP
Shark	Somniosidae	*Centroscymnus owstonii* Garman, 1906	Owston's dogfish	CYO	Non-QMS	LC	NOT	
Shark	Somniosidae	*Centroselachus crepidater* (Bocage & Capello, 1864)	Longnose velvet dogfish	CYP	Non-QMS	LC	NOT	SO
Shark	Somniosidae	*Scymnodalatias albicauda* Taniuchi & Garrick, 1986	Whitetail dogfish	SLB	Non-QMS	DD	DD	S?O
Shark	Somniosidae	*Scymnodalatias sherwoodi* (Archey, 1921)	Sherwood's dogfish	SHE	Non-QMS	DD	DD	S?O
Shark	Somniosidae	*Centroscymnus macracanthus* Regan 1906	Plunket's shark	PLS	Non-QMS	NT	NOT	T?O
Shark	Somniosidae	*Scymnodon ringens* Bocage & Capello, 1864	Knifetooth dogfish	SRI	Non-QMS	DD	DD	S?O
Shark	Somniosidae	*Somniosus antarcticus* Whitley, 1939	Southern sleeper shark	SSS	Non-QMS	DD	NOT	DP,S?O
Shark	Somniosidae	*Somniosus longus* (Tanaka, 1912)	Little sleeper shark	SOM	Non-QMS	DD	DD	S?O
Shark	Somniosidae	*Zameus squamulosus* (Günther, 1877)	Velvet dogfish	ZAS	Non-QMS	DD	DD	S?O
Shark	Oxynotidae	*Oxynotus bruniensis* (Ogilby, 1893)	Prickly dogfish	PDG	Non-QMS	DD	NOT	DP,SO
Shark	Dalatiidae	*Dalatias licha* (Bonnaterre, 1788)	Seal shark	BSH	Non-QMS	NT	NOT	SO
Shark	Dalatiidae	*Euprotomicrus bispinatus* (Quoy & Gaimard, 1824)	Pygmy shark	EBI	Non-QMS	LC	NOT	SO
Shark	Dalatiidae	*Isistius brasiliensis* (Quoy & Gaimard, 1824)	Cookie cutter shark	IBR	Non-QMS	LC	NOT	SO
Shark	Heterodontidae	*Heterodontus portusjacksoni* (Meyer, 1793)	Port Jackson shark	PJS	Non-QMS	LC	VA	SO
Shark	Rhincodontidae	*Rhincodon typus* Smith, 1828	Whale shark	WSH	Protected	EN	MI	SO
Shark	Odontaspidae	*Odontaspis ferox* (Risso, 1810)	Deepwater (smalltooth) sand tiger shark	ODO	Protected	VU	ARNU	TO
Shark	Pseudocarchariidae	*Pseudocarcharias kamoharai* (Matsubara, 1936)	Crocodile shark.	CRC	Non-QMS	NT	DD	SO
Shark	Mitsukurinidae	*Mitsukurina owstoni* Jordan, 1898	Goblin shark	GOB	Non-QMS	LC	ARNU	DP,SO

Appendix 19.10.1 [Continued]:

Group	Family	Species	Common name	Code	Manage-ment class	IUCN redlist class	DoC threat class	DoC qualifier
Shark	Alopiidae	*Alopias superciliosus* Lowe 1841	Bigeye thresher	BET	Non-QMS	VU	NOT	TO
Shark	Alopiidae	*Alopias vulpinus* (Bonnaterre, 1788)	Thresher shark	THR	Non-QMS	VU	NOT	DP,TO
Shark	Cetorhinidae	*Cetorhinus maximus* (Gunnerus, 1765)	Basking shark	BSK	Protected	VU	NV	
Shark	Lamnidae	*Carcharodon carcharias* (Linnaeus, 1758)	White shark, white pointer	WPS	Protected	VU	NE	DP,TO
Shark	Lamnidae	*Isurus oxyrinchus* Rafinesque, 1810	Mako shark, shortfin mako	MAK	QMS	VU	NOT	S?O
Shark	Lamnidae	*Lamna nasus* (Bonnaterre, 1788)	Porbeagle shark	POS	QMS	VU	NOT	TO
Shark	Pentanchidae	*Apristurus albisoma* Nakaya & Seret 1999	Grey roundfin catshark		Non-QMS	LC	DD	
Shark	Pentanchidae	*Apristurus ampliceps* Sasahara, Sato & Nakaya 2008	Roughskin cat shark	AAM	Non-QMS	LC	DD	
Shark	Pentanchidae	*Apristurus exsanguis* Sato, Nakaya and Stewart 1999	Pale catshark	AEX	Non-QMS	LC	DD	
Shark	Pentanchidae	*Apristurus garricki* Sato, Stewart & Nakaya 2013	Garrick's catshark	AGK	Non-QMS		DD	
Shark	Pentanchidae	*Apristurus melanoasper* Iglésias, Nakaya & Stehmann 2004	Fleshynose cat shark	AML	Non-QMS	LC	DD	
Shark	Pentanchidae	*Apristurus pinguis* Deng, Xiong & Zhan 1983	Bulldog catshark	APN	Non-QMS	LC	DD	
Shark	Pentanchidae	*Apristurus* cf *sinensis* Chu & Hu 1981	Freckled cat shark	ASI	Non-QMS		DD	
Shark	Scyliorhinidae	*Bythaelurus dawsoni* (Springer, 1971)	Dawson's cat shark	DCS	Non-QMS	DD	NOT	DP
Shark	Scyliorhinidae	*Cephaloscyllium isabella* (Bonnaterre, 1788)	Carpet shark	CAR	Non-QMS	LC	NOT	
Shark	Scyliorhinidae	*Cephaloscyllium* cf *variegatum* Last & White 2008	Swellshark		Non-QMS		DD	
Shark	Pentanchidae	*Parmaturus macmillani* Hardy, 1985	McMillan's cat shark	PCS	Non-QMS	DD	DD	S?O
Shark	Pentanchidae	*Parmaturus* sp.	Rough-backed cat shark		Non-QMS		DD	
Shark	Pseudotriakidae	*Gollum attenuatus* (Garrick, 1954)	Slender smooth hound	SSH	Non-QMS	LC	NOT	SO
Shark	Pseudotriakidae	*Pseudotriakis microdon* de Brito Capello, 1868	False cat shark	PMI	Non-QMS	LC	DD	SO
Shark	Triakidae	*Galeorhinus galeus* (Linnaeus, 1758)	School shark	SCH	QMS	VU	NOT	CD,TO
Shark	Triakidae	*Mustelus lenticulatus* Phillipps, 1932	Rig	SPO	QMS	LC	NOT	CD
Shark	Triakidae	*Mustelus* sp.	Kermadec rig		Non-QMS		NOT	
Shark	Carcharhinidae	*Carcharhinus brachyurus* (Günther, 1870)	Bronze whaler	BWH	Non-QMS	NT	NOT	CD,DP,SO
Shark	Carcharhinidae	*Carcharhinus galapagensis* (Snodgrass & Heller, 1905)	Galapagos shark	CGA	Non-QMS	NT	NOT	CD,SO
Shark	Carcharhinidae	*Carcharhinus longimanus* (Poey, 1861)	Oceanic whitetip shark	OWS	Protected	VU	MI	SO
Shark	Carcharhinidae	*Carcharhinus obscurus* (Le Sueur, 1818)	Dusky shark	DSH	Non-QMS	VU	MI	SO
Shark	Carcharhinidae	*Carcharhinus plumbeus* (Nardo 1827)	Sandbar shark		Non-QMS	VU	DD	
Shark	Carcharhinidae	*Galeocerdo cuvier* (Peron & LeSueur, 1822)	Tiger shark	TIS	Non-QMS	NT	MI	SO
Shark	Carcharhinidae	*Prionace glauca* (Linnaeus, 1758)	Blue shark	BWS	QMS	NT	NOT	SO
Shark	Carcharhinidae	*Triaenodon obesus* (Rüppell 1837)	Whitetip reef shark	TRB	Non-QMS	NT	VA	
Shark	Sphyrnidae	*Sphyrna zygaena* (Linnaeus, 1758)	Hammerhead shark, smooth hammerhead	HHS	Non-target	VU	NOT	SO
Batoid	Narkidae	*Typhlonarke aysoni* (Hamilton, 1902)	Blind electric ray	TAY	Non-QMS	DD	NOT	DP
Batoid	Torpedinidae	*Tetronarce nobiliana* (Bonaparte, 1835)	Electric ray	ERA	Non-QMS	DD	DD	
Batoid	Torpedinidae	*Tetronarce* cf *tokionis* (Tanaka 1908)	Slender electric ray		Non-QMS		DD	
Batoid	Arhynchobatidae	*Arhynchobatis asperrimus* Waite, 1909	Longtail skate	LSK	Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Bathyraja* cf. *eatonii*	Antarctic allometric skate	BEA	Non-QMS			
Batoid	Arhynchobatidae	*Bathyraja maccaini* Springer 1971	MacCain's skate	MCS	Non-QMS	NT		
Batoid	Arhynchobatidae	*Bathyraja pacifica* Last, Stewart & Seret 2016	Pacific blonde skate		Non-QMS		NOT	DP
Batoid	Arhynchobatidae	*Bathyraja richardsoni* (Garrick, 1961)	Richardson's skate	RIS	Non-QMS	LC	NOT	DP
Batoid	Arhynchobatidae	*Bathyraja shuntovi* Dolganov, 1985	Longnose deepsea skate	PSK	Non-QMS	DD	NOT	
Batoid	Arhynchobatidae	*Bathyraja* sp.	Antarctic dwarf skate	BHY	Non-QMS			
Batoid	Arhynchobatidae	*Brochiraja albilabiata* Last & McEachran, 2006	Whitemouth skate		Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Brochiraja asperula* (Garrick & Paul, 1974)	Smooth deepsea skate	BTA	Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Brochiraja heuresa* Last & Seret 2012	Eureka skate		Non-QMS		DD	
Batoid	Arhynchobatidae	*Brochiraja leviveneta* Last & McEachran, 2006	Blue skate	BRL	Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Brochiraja microspinifera* Last & McEachran, 2006	Dwarf skate	BMI	Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Brochiraja spinifera* (Garrick & Paul, 1974)	Prickly deepsea skate	BTS	Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Brochiraja vittacauda* Last & Seret 2012	Ribbontail skate		Non-QMS		DD	
Batoid	Arhynchobatidae	*Notoraja alisae* Seret & Last 2012	Velcro skate	NAL	Non-QMS		DD	
Batoid	Arhynchobatidae	*Notoraja sapphira* Seret & Last 2009	Sapphire skate		Non-QMS	DD	DD	
Batoid	Rajidae	*Amblyraja georgiana* (Norman 1938)	Antarctic starry skate	SRR	Non-QMS	DD		
Batoid	Rajidae	*Amblyraja hyperborea* (Collett, 1879)	Deepwater spiny skate	DSK	Non-QMS	LC	NOT	
Batoid	Rajidae	*Dipturus innominatus* (Garrick & Paul, 1974)	Smooth skate	SSK	QMS	NT	NOT	CD
Batoid	Rajidae	*Zearaja nasuta* (Müller & Henle, 1841)	Rough skate	RSK	QMS	LC	NOT	CD
Batoid	Dasyatidae	*Bathytoshia brevicaudata* (Hutton, 1875)	Shorttail stingray	BRA	Non-QMS	LC	NOT	SO
Batoid	Dasyatidae	*Bathytoshia lata* (Garman 1880)	Longtail stingray	WRA	Non-QMS	LC	NOT	SO
Batoid	Dasyatidae	*Pteroplatytrygon violacea* (Bonaparte, 1832)	Pelagic stingray	DAS	Non-QMS	LC	NOT	SO
Batoid	Myliobatidae	*Myliobatis tenuicaudatus* Hector, 1877	Eagle ray	EGR	Non-QMS	LC	NOT	DP,SO
Batoid	Mobulidae	*Manta birostris* (Walbaum, 1792)	Manta ray	RMB	Protected	VU	DD	TO
Batoid	Mobulidae	*Mobula mobular* (Bonnaterre 1788)	Spinetail devil ray	MJA	Protected	NT	DD	SO

Appendix 19.10.2: Indicative information on status of stocks for the eleven shark species subject to the QMS.

* denotes Highly Migratory Species, for which stock status cannot be determined for the portion of the stock found within New Zealand waters.

Species name	Plenary stock	Last assessment date	At or above target levels?	Below the soft limit?	Below the hard limit?	Overfishing?	Corrective management action
Blue shark*	BWS1	2014					-
Elephant fish	ELE2	-					-
Elephant fish	ELE3	2016	●	●●	●●●	■	-
Elephant fish	ELE5	2017	●	●●	●●●	■	-
Elephant fish	ELE7	2015		●●	●●●	●●	-
Ghost shark - dark	GSH1, GSH2, GSH7, GSH8	-					-
Ghost shark - dark	GSH3	-				●●	-
Ghost shark - dark	GSH4, GSH5, GSH6	-					-
Ghost shark - pale	GSP1, GSP5	2011		●●	●●●		-
Ghost shark - pale	GSP7	-					-
Hammerhead sharks (smooth)	HHS1	-					-
Mako shark*	MAK1	2014					TAC reduced in 2012
Porbeagle shark*	POS1	2014					TAC reduced in 2012
Rig	SPO1	2016					-
Rig	SPO2	2016	●	●●	●●●	●●	-
Rig	SPO3	2016	●●	●●●	●●●	■	-
Rig	SPO7, SPO8	2016	●●	●●●	●●●	●●●	-
School shark	SCH1, SCH2, SCH3, SCH4, SCH5, SCH7, SCH8	2014			●●		-
Skate - rough	RSK1, RSK3, RSK7, RSK8	2007					-
Skate - smooth	SSK1, SSK3, SSK7, SSK8	2007					-
Spiny dogfish	SPD1, SPD8	-					-
Spiny dogfish	SPD3, SDP7	2009			●●		-
Spiny dogfish	SPD4	2009			●●		-
Spiny dogfish	SPD5	-					-

NOTES

At or above target levels? The 'at or above target levels' indicator describes the present status of the stock relative to its target (usually B_{MSY}, the average biomass associated with a maximum sustainable yield (MSY) strategy, or F_{MSY}, the associated fishing mortality, or appropriate surrogates or proxies for these metrics, or alternative reference points that will result in higher average biomass – see Maximum Sustainable Yield Harvest Strategies for definitions and explanations of these terms).

Below the soft limit? Below the hard limit? Overfishing? In April 2009, the Ministry's Stock Assessment Methods Working Group adopted a probabilistic scale for categorising the 'at or above target levels', 'below the soft limit', 'below the hard limit' and 'overfishing' indicators (based on the scale developed by the Intergovernmental Panel on Climate Change (IPCC) in 2007). While these probability categories are best applied in situations where models give appropriate quantitative outputs, they can also be used subjectively, based on expert opinion, when such model outputs are not available, or are highly uncertain.

The stock status table uses the IPCC criteria, coded according to the following key:

At or above target levels?	Probability	Description	Below the soft limit? Below the hard limit? Overfishing?
●●●●	> 99 %	Virtually Certain	■■■■
●●●	> 90 %	Very Likely	■■■
●●	> 60 %	Likely	■■
●	40 - 60 %	About as Likely as Not	■
■■	< 40 %	Unlikely	●●
■■■	< 10 %	Very Unlikely	●●●
■■■■	< 1 %	Exceptionally Unlikely	●●●●

Note that green circles indicate a favourable status, while orange squares indicate an unfavourable status, with the number of circles or squares indicating the degree to which the status is favourable or unfavourable.

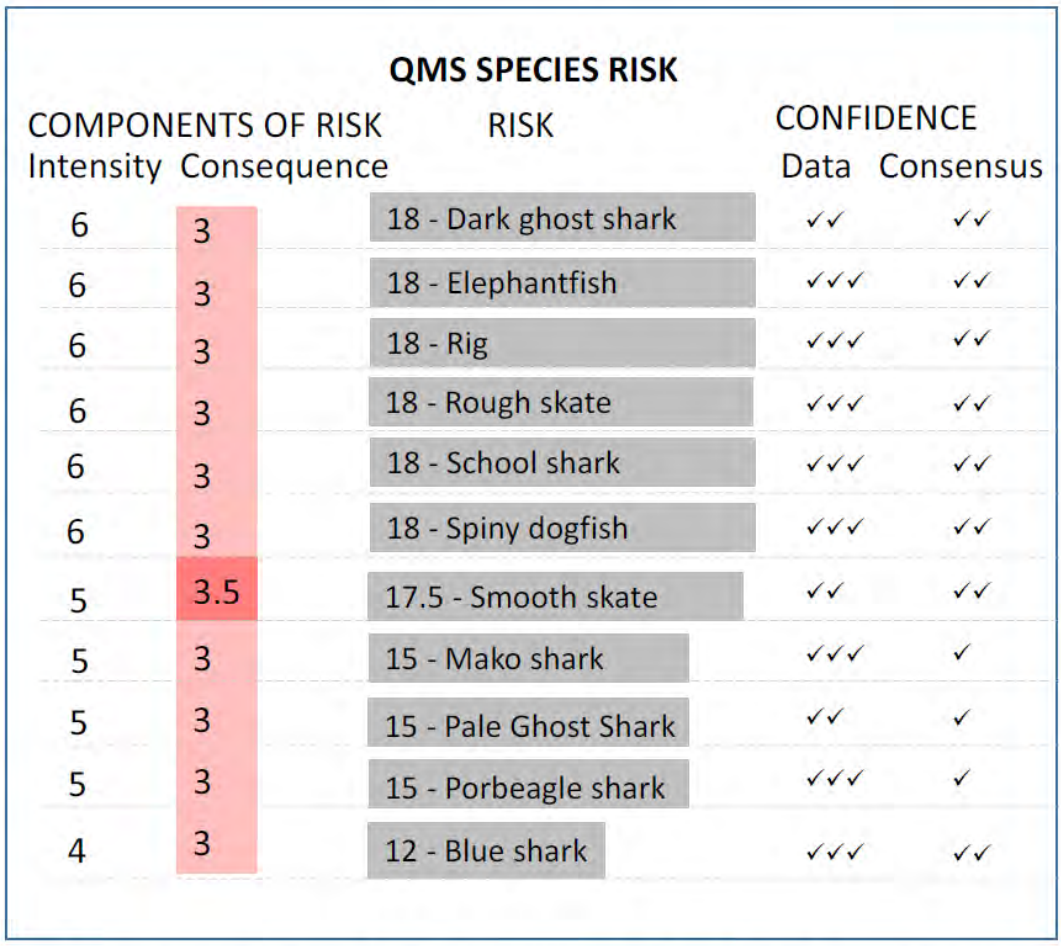
Whether or not a stock is likely to be at or above the target level, or to be below the soft or hard limits, or subject to overfishing, is based on the most recent stock assessment summarised in the Ministry's Fishery Assessment Plenary Reports. The current (2018) stock status may be better or worse than that indicated by the most recent stock assessment. Where several alternative assessment runs are reported (as is frequently the case), or if the assessment results are contentions, the result reported represents the best judgement on the part of the Chair of the appropriate Fisheries Assessment Working Group, and the Ministry's Principal Advisor Fisheries Science.

Corrective management action: This column describes corrective management action underway for those stocks believed to be below the target level, or the soft or hard limits, or subject to overfishing.

Grey shading indicates that stock status is unknown, because an appropriate quantitative analysis to ascertain stock status relative to a target or limit has not been undertaken, or because such an analysis was not definitive, generally because of insufficient or inadequate data.

Source: based on the Status of the Stocks 2017 data published by the Ministry for Primary Industries on its website (https://www.mpi.govt.nz/growing-and-harvesting/fisheries/fisheries-management/fish-stock-status/).

Appendix 19.10.3: QMS chondrichthyan risk assessment results from Ford et al. (2018). [Continued on next pages]



QMS SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
6	3	18 - Dark ghost shark	✓✓	✓✓
6	3	18 - Elephantfish	✓✓✓	✓✓
6	3	18 - Rig	✓✓✓	✓✓
6	3	18 - Rough skate	✓✓✓	✓✓
6	3	18 - School shark	✓✓✓	✓✓
6	3	18 - Spiny dogfish	✓✓✓	✓✓
5	3.5	17.5 - Smooth skate	✓✓	✓✓
5	3	15 - Mako shark	✓✓✓	✓
5	3	15 - Pale Ghost Shark	✓✓	✓
5	3	15 - Porbeagle shark	✓✓✓	✓
4	3	12 - Blue shark	✓✓✓	✓✓

Figure 5: QMS species risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 3 and Table 4). For RISK longer bars and larger numbers indicate higher risk, and for CONFIDENCE more ticks indicate higher confidence in the data, or greater consensus (Two ticks in the consensus column indicate full consensus). Where species scored identical risk scores they are presented in descending order of consequences and then alphabetically.

NON-QMS SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
5	4.5	22.5 – Plunket's shark	✓✓	✓
5	4	20 - Baxters dogfish	✓✓	✓✓
5	4	20 - Seal shark	✓✓	✓✓
5	4	20 - Shovelnose dogfish	✓✓	✓
5	4	20 - Thresher shark	✓✓	✓✓
4	4.5	18 - Leafscale gulper shark	✓✓	✓
4.5	4	18 - Longnose velvet dogfish	✓✓	✓✓
6	3	18 - Carpet Shark	✓✓	✓✓
5	3.5	17.5 - Longtail stingray	✓	✓✓
5	3.5	17.5 - Shorttail stingray	✓	✓✓
4	4	16 - Owston's dogfish	✓✓	✓
3.5	4.5	15.75 - Dawsons catshark	✓✓	✓
4.5	3.5	15.75 - Longnose spookfish	✓	✓
5	3	15 - Electric ray	✓✓	✓
3.5	4	14 - Bronze whaler	✓✓	✓
3.5	4	14 - Prickly dogfish	✓✓	✓
4	3.5	14 - Northern spiny dogfish	✓✓	✓
3.5	3.5	12.25 - Prickly deepsea skate	✓✓	✓
3.5	3.5	12.25 - Smooth deepsea skate	✓✓	✓
3	4	12 - *Brochiraja* complex	✓	✓
3	4	12 - Brown chimaera	✓	✓✓
3	4	12 - Catsharks	✓	✓
3	4	12 - Deepwater spiny skate	✓	✓
3	4	12 - Longnose deepsea skate	✓	✓
3	4	12 - Longtail skate	✓	✓
3	4	12 - Lucifer dogfish	✓✓	✓
3	4	12 - Pacific spookfish	✓	✓✓
3	4	12 - Pelagic stingray	✓	✓
3	4	12 - Portugese dogfish	✓✓	✓
3	4	12 - Slender smooth hound	✓	✓
4	3	12 - Hammerhead shark	✓✓	✓
4	3	12 - Blind electric ray	✓✓	✓
4	3	12 - Broadnose sevengill shark	✓✓	✓
4	2.5	10 – Eagle ray	✓✓	✓
3	3	9 – Sharpnose sevengill shark	✓✓	✓
3	2	6 – Sixgill shark	✓✓	✓

Figure 6: Non-QMS Species Risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 3 and Table 4). For RISK longer bars and larger numbers indicate higher risk, and for CONFIDENCE more ticks indicate higher confidence in the data, or greater consensus (Two ticks in the consensus column indicate full consensus). Where taxa risk scores were identical they are presented so that higher consequences are reported first and then in alphabetical order. Taxa that scored less than three for consequence were not scored further, see Section 2.3 for more details. See Ford et al. (2015) for available data on shark species not listed in the table above.

Appendix 19.10.3 [Continued]:

PROTECTED SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
3	4.5	13.5 – Basking shark	✓✓	✓
3	4.5	13.5 – Spinetail devil ray	✓	✓
3	4	12 – Great white shark	✓✓	✓

Figure 7: Protected Species Risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 3 and Table 4). For RISK longer bars and larger numbers indicate higher risk, and for CONFIDENCE more ticks indicate higher confidence in the data, or greater consensus and a cross indicates a lack of consensus (Two ticks in the consensus column indicate full consensus). Where species scored identical risk scores they are presented so that higher consequences are reported first and then taxa are in alphabetical order. Taxa that scored less than three for consequence were not scored further, see Section 2.3 for more details. See Ford et al. (2015) for available data on shark species not listed in the table above.

19.11 REFERENCES

Abraham, E (2008) Evaluating methods for estimating the incidental capture of New Zealand sea lions. *New Zealand Aquatic Environment and Biodiversity Report No. 15*. 25 p.

Abraham, E (2010) Warp strike in New Zealand trawl fisheries, 2004–05 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 60*. 29 p.

Abraham, E (2011) Probability of Mild Traumatic Brain Injury for sea lions interacting with SLEDs. Final Research Report for Ministry of Fisheries project SRP2011-03. 21 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Abraham, E; Kennedy, A (2008) Seabird warp strike in the southern squid trawl fishery, 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 16*. 39 p.

Abraham, E R; Berkenbusch, K (2017) Estimated captures of New Zealand fur seal, New Zealand sea lion, common dolphin, and turtles in New Zealand commercial fisheries, 1995–96 to 2014–15. *New Zealand Aquatic Environment and Biodiversity Report No. 188*. 66 p.

Abraham, E R; Neubauer, P; Berkenbusch, K; Richard, Y (2017) Assessment of the risk to New Zealand marine mammals from commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 189*. 123 p.

Abraham, E R; Richard, Y (2017) Summary of the capture of seabirds in New Zealand commercial fisheries, 2002–03 to 2013–14. *New Zealand Aquatic Environment and Biodiversity Report No. 184*. 88 p.

Abraham, E R; Richard, Y; Bell, E; Landers, T J (2015) Overlap of the distribution of black petrel (*Procellaria parkinsoni*) with New Zealand trawl and longline fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 161*. 30 p.

Abraham, E R; Richard, Y; Berkenbusch, K; Thompson, F (2016) Summary of the capture of seabirds, marine mammals, and turtles in New Zealand commercial fisheries, 2002–03 to 2012–13. *New Zealand Aquatic Environment and Biodiversity Report No. 169*. 205 p.

Abraham, E; Thompson, F (2009a) Capture of protected species in New Zealand trawl and longline fisheries, 1998–99 to 2006–07. *New Zealand Aquatic Environment and Biodiversity Report No. 32*. 197 p.

Abraham, E; Thompson, F (2009b) Warp strike in New Zealand trawl fisheries, 2004–05 to 2006–07. *New Zealand Aquatic Environment and Biodiversity Report No. 33*. 21 p.

Abraham, E; Thompson, F (2010) Summary of the capture of seabirds, marine mammals, and turtles in New Zealand commercial fisheries, 1998–99 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 80*. 172 p.

Abraham, E; Thompson, F (2011a) Estimated capture of seabirds in New Zealand trawl and longline fisheries, 2002–03 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 79*. 74 p.

Abraham, E; Thompson, F (2011b) Summary of the capture of seabirds, marine mammals, and turtles in New Zealand commercial fisheries, 1998–99 to 2008–09. Unpublished Final Research Report for the Ministry of Fisheries project PRO2007-01. 170 p.

Abraham, E, et al. (2010a) Summary of the capture of seabirds, marine mammals, and turtles in New Zealand commercial fisheries, 1998–99 to 2007–08. *New Zealand Aquatic Environment and Biodiversity Report No. 45*. 148 p.

Abraham, E, et al. (2010b) The capture of seabirds and marine mammals in New Zealand non-commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Review No. 64*. 52 p.

Allcock, A, et al. (2009) Cryptic speciation and the circumpolarity debate: a case study on endemic Southern Ocean octopuses using the COI barcode of life. CAML Symposium, Genoa.

Allcock, A, et al. (2010) Cryptic speciation and the circumpolarity debate: A case study on endemic Southern Ocean octopuses using the COI barcode of life. *Deep Sea Research 58 (1–2)*: 242–249. doi:10.1016/j.dsr2.2010.05.016.

Allcock, A, et al. (submitted) Bipolarity in marine invertebrates: myth or marvel. Submitted for consideration in the Census Nature cluster. 2009.

Alvaro, M, et al. (2011) Skin-digging tanaids: the unusal parasitic behaviour of Exspinia typica in Antarctic waters and worldwide deep basins. CCAMLR document WG-FSA-10/P3 23: 343–348.

Anderson, O (2004) Fish discards and non-target fish catch in the trawl fisheries for arrow squid, jack mackerel, and scampi in New Zealand waters. *New Zealand Fisheries Assessment Report 2004/10*. 61 p.

Anderson, O (2007) Fish discards and non-target fish catch in the New Zealand jack mackerel trawl fishery, 2001–02 to 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 8*. 36 p.

Anderson, O (2008) Fish and invertebrate bycatch and discards in ling longline fisheries, 1998–2006. *New Zealand Aquatic Environment and Biodiversity Report No. 23*. 43 p.

Anderson, O (2009a) Fish discards and non-target fish catch in the New Zealand orange roughy trawl fishery: 1999–2000 to 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 39*. 40 p.

Anderson, O (2009b) Fish and invertebrate bycatch and discards in southern blue whiting fisheries, 2002–07. *New Zealand Aquatic Environment and Biodiversity Report No. 43*. 42 p.

Anderson, O (2011) Fish and invertebrate bycatch and discards in orange roughy and oreo fisheries from 1990–91 until 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 67*. 60 p.

Anderson, O (2012) Fish and invertebrate bycatch and discards in New Zealand scampi fisheries from 1990–91 until 2009–10. *New Zealand Aquatic Environment and Biodiversity Report No. 100*. 65 p.

Anderson, O (2013a) Fish and invertebrate bycatch and discards in New Zealand arrow squid fisheries from 1990–91 until 2010–11. *New Zealand Aquatic Environment and Biodiversity Report No. 112*. 62 p.

Anderson, O (2013b) Fish and invertebrate bycatch in New Zealand deepwater fisheries from 1990–91 until 2010–11. *New Zealand Aquatic Environment and Biodiversity Report No. 113*. 57 p.

Anderson, O F (2014a) Fish and invertebrate bycatch and discards in New Zealand ling longline fisheries from 1992–93 until 2011–12. *New Zealand Aquatic Environment and Biodiversity Report No. 138*. 66 p.

Anderson, O F (2014b) Fish and invertebrate bycatch in New Zealand deepwater fisheries from 1990–91 until 2011–12. *New Zealand Aquatic Environment and Biodiversity Report No. 139*. 60 p.

Anderson, O; Smith, M (2005) Fish discards and non-target fish catch in the New Zealand hoki trawl fishery, 1999/2000 to 2002/03. *New Zealand Fisheries Assessment Report 2005/03*. 37 p.

Anderson, O, et al. (2001) Fish discards and non-target catch in the trawl fisheries for orange roughy and hoki in New Zealand waters for the fishing years 1990–91 to 1998–99. *New Zealand Fisheries Assessment Report 2001/16*. 57 p.

Andrew, N; Francis, M (Eds.) (2003) The Living Reef: the ecology of New Zealand's Living Reef. Craig Potton Publishing, Nelson.

Ayers, D, et al. (2004) Fish bycatch in New Zealand tuna longline fisheries, 2000–01 and 2001–02. *New Zealand Fisheries Assessment Report 2004/46*. 47 p.

Baird, S (1994) Nonfish Species and Fisheries Interactions Working Group Report. NZ Fisheries Assessment Working Group Report 94/1, Ministry of Agriculture and Fisheries, Wellington. 54 p.

Baird, S (1995) Nonfish Species and Fisheries Interactions Working Group Report. NZ Fisheries Assessment Working Group Report 95/1, Ministry of Agriculture and Fisheries, Wellington. 24 p.

Baird, S (1996) Nonfish Species and Fisheries Interactions Working Group Report. NZ Fisheries Assessment Working Group Report 96/1, Ministry of Agriculture and Fisheries, Wellington. 34 p.

Baird, S (1997) Report on the incidental capture of nonfish species during fishing operations in New Zealand waters. Unpublished Final Research Report for Ministry of Fisheries SANF01 contract. 15 p plus appendices on New Zealand fur seal-trawl fishery interaction (54 p) and seabird-tuna longline fishery interaction (34 p).

Baird, S (1999a) Determination of factors which affect nonfish bycatch in some New Zealand fisheries. Unpublished Final Research Report for Ministry of Fisheries project ENV9701, Objective 3. 83 p.

Baird, S (1999b) Estimation of nonfish bycatch in commercial fisheries in New Zealand waters, 1997–98. Unpublished Final Research Report for Ministry of Fisheries project ENV9801, Objective 1. 57 p.

Baird, S (2001) Estimation of the incidental capture of seabird and marine mammal species in commercial fisheries in New Zealand waters, 1998–99. *New Zealand Fisheries Assessment Report 2001/14*. 43 p.

Baird, S (2003) New Zealand breeding seabirds: human-induced mortality – a review. Unpublished Report prepared for Ministry of Fisheries project ENV2000/09. 74 p.

Baird, S (2004a) Estimation of the incidental capture of seabird and marine mammal species in commercial fisheries in New Zealand waters, 1999–2000. *New Zealand Fisheries Assessment Report 2004/41*. 56 p.

Baird, S (2004b) Incidental capture of seabird species in commercial fisheries in New Zealand waters, 2000–01. *New Zealand Fisheries Assessment Report 2004/58*. 63 p.

Baird, S (2004c) Incidental capture of seabird species in commercial fisheries in New Zealand waters, 2001–02. *New Zealand Fisheries Assessment Report 2004/60*. 51 p.

Baird, S (2005a) Incidental capture of *Phocarctos hookeri* (New Zealand sea lions) in New Zealand commercial fisheries, 2002–03. *New Zealand Fisheries Assessment Report 2005/8*. 17 p.

Baird, S (2005b) Incidental capture of *Phocarctos hookeri* (New Zealand sea lions) in New Zealand commercial fisheries, 2002–03. *New Zealand Fisheries Assessment Report 2005/9*. 13 p.

Baird, S (2005c) Incidental capture of New Zealand fur seals (*Arctocephalus forsteri*) in commercial fisheries in New Zealand waters, 2000–01. *New Zealand Fisheries Assessment Report 2005/11*. 34 p.

Baird, S (2005d) Incidental capture of New Zealand fur seals (*Arctocephalus forsteri*) in commercial fisheries in New Zealand waters, 2002–03. *New Zealand Fisheries Assessment Report 2005/12*. 35 p.

Baird, S (2005e) Incidental capture of New Zealand fur seals (*Arctocephalus forsteri*) in commercial fisheries in New Zealand waters, 2001–02. *New Zealand Fisheries Assessment Report 2005/12*. 33 p.

Baird, S (2005f) Incidental capture of seabird species in commercial fisheries in New Zealand waters, 2002/03. *New Zealand Fisheries Assessment Report 2005/02*. 50 p.

Baird, S (2007) Incidental capture of cetaceans in commercial fisheries in New Zealand waters 2003–04 and 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 13*. 27 p.

Baird, S (2008a) Incidental capture of cetaceans in commercial fisheries in New Zealand waters, 1994–95 to 2005–06. *New Zealand Aquatic Environment and Marine Biodiversity Report No. 21*. 29 p.

Baird, S (2008b) Incidental capture of New Zealand fur seals (*Arctocephalus jorsteri*) in longline fisheries in New Zealand waters, 1994–95 to 2005–06. *New Zealand Aquatic Environment and Biodiversity Report No. 20*. 21 p.

Baird, S (2011) New Zealand fur seals – summary of current knowledge. *New Zealand Aquatic Environment and Biodiversity Report No. 72*. 51 p.

Baird, S; Bradford, E (1999) Factors that may influence the bycatch of nonfish species in some New Zealand fisheries. Unpublished Final Research Report completed for Ministry of Fisheries project ENV9801, Objective 3. 106 p.

Baird, S; Bradford, E (2000a) Factors that may have influenced bycatch of New Zealand fur seals (*Arctocephalus forsteri*) in the west coast South Island hoki fishery. *NIWA Technical Report 92*. 35 p.

Baird, S; Bradford, E (2000b) Factors that may have influenced the capture of seabirds in New Zealand tuna longline fisheries. *NIWA Technical Report 93*. 61 p.

Baird, S; Doonan, L (2005) *Phocarctos hookeri* (New Zealand sea lions): incidental captures in New Zealand commercial fisheries during 2000–2001 and in-season estimates of captures during squid trawling in SQU 6T in 2002. *New Zealand Fisheries Assessment Report 2005/17*. 18 p.

Baird, S; Gilbert, D (2010) Initial assessment of risk posed by trawl and longline fisheries to selected seabird taxa breeding in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 50*. 98 p.

Baird, S; Griggs, L (2005) Estimation of within-season chartered southern bluefin tuna (*Thunnus maccoyii*) longline seabird incidental captures, 2003. *New Zealand Fisheries Assessment Report 2OO5/01*. 15 p.

Baird, S J; Hewitt, J E; Wood, B A (2015) Benthic habitat classes and trawl fishing disturbance in New Zealand waters shallower than 250 m. *New Zealand Aquatic Environment and Biodiversity Report No. 144.* 184 p.

Baird, S; Mormede, S (2014) Environmental preferences of seabirds and spatial distribution of seabirds and marine mammals in the Ross Sea area. *New Zealand Aquatic Environment and Biodiversity Report No. 121*. 43 p.

Baird, S; Smith, M (2007a) Incidental capture of New Zealand fur seals (*Arctocephalus forsteri*) in commercial fisheries in New Zealand waters, 2003–04 and 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 14*. 98 p.

Baird, S; Smith, M (2007b) Incidental capture of seabird species in commercial fisheries in New Zealand waters, 2003–04 and 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 9.* 108 p.

Baird, S; Smith, M (2008) Incidental capture of seabird species in commercial fisheries in New Zealand waters, 2005–06. *New Zealand Aquatic Environment and Biodiversity Report No. 18*. 124 p.

Baird, S; Wood, B (2010) Extent of coverage of 15 environmental classes within the New Zealand EEZ by commercial trawling with seafloor contact. *New Zealand Aquatic Environment and Biodiversity Report No. 89*. 43 p.

Baird, S, et al. (1998) Annual review of bycatch in southern bluefin and related tuna longline fisheries in the New Zealand 200 n. mile Exclusive Economic Zone. Report prepared for the Third Meeting of the Ecologically Related Species Working Group, Tokyo, 9–13 June 1998. CCSBT-ERS/9806/31.

Baird, S, et al. (1999) Estimation of nonfish bycatch in commercial fisheries in New Zealand waters, 1990–91 to 1993–94. Final Research Report for Ministry of Fisheries project ENV9701, Objective 1. 63 p.

Baird, S, et al. (2002) The spatial extent and nature of mobile bottom fishing methods within the New Zealand EEZ, 1989–90 to 1998–99. Unpublished Final Research Report for Ministry of Fisheries project ENV2000/05, Objective 1.

Baird, S, et al. (2006) Description of the spatial extent and nature of disturbances by bottom trawls in Chatham Rise and Southern Plateau fisheries. Unpublished Final Research Report for Ministry of Fisheries project ENV2003/03. 139 p.

Baird, S, et al. (2009) The extent of trawling on or near the seafloor in relation to benthic-optimised marine environment classes within the New Zealand EEZ. Unpublished Final Research Report for Ministry of Fisheries project BEN200601, Objective 5.

Baird, S, et al. (2010) To review the current threat status of selected associated or dependent species. Unpublished Final Research Report for the Ministry of Fisheries. 101 p.

Baird, S, et al. (2011) Nature and extent of commercial fishing effort on or near the seafloor within the New Zealand 200 n. Mile Exclusive Economic Zone, 1989–90 to 2004–05. *New Zealand Aquatic Environment and Biodiversity Report No. 73*. 143 p.

Baker, B, et al. (2008) Data collection of demographic, distributional and trophic information on the Westland petrel to allow estimation of effects of fishing on population viability. Unpublished Final Research Report to the Ministry of Fisheries. 47 p.

Baker, B, et al. (2009) Data collection of demographic, distributional and trophic information on the white-capped albatross to allow estimation of effects of fishing on population viability: 2009 Field Season. Unpublished Research Report for the Ministry of Fisheries. 15 p.

Baker, B, et al. (2010) Data collection of demographic, distributional, and trophic information on the flesh-footed shearwater to allow estimation of effects of fishing on population viability: 2009–10 Field Season. Unpublished Final Research Report for the Ministry of Fisheries. 62 p.

Ballara, S L (2015) Fish and invertebrate bycatch in New Zealand deepwater fisheries from 1990–91 until 2012–13. *New Zealand Aquatic Environment and Biodiversity Report No. 158*. 79 p.

Ballara, S; Anderson, O (2009) Fish discards and non-target fish catch in the trawl fisheries for arrow squid and scampi in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 38*. 102 p.

Ballara, S L; O'Driscoll, R L (2015) Fish and invertebrate bycatch and discards in New Zealand hoki, hake, and ling fisheries from 1990–91 until 2012–13. *New Zealand Aquatic Environment and Biodiversity Report No. 163.* 120 p.

Ballara, S, et al. (2010) Fish discards and non-target fish catch in the trawl fisheries for hoki, hake, and ling in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 48*. 100 P.

Batson, P (2003) Deep New Zealand: Blue Water, Black Abyss. Canterbury University Press, Christchurch.

Batson, P; Probert, P K (2000) Bryozoan thickets off Otago Peninsula. *New Zealand Fisheries Assessment Report 2000/46*. 31 p.

Battley, P, et al. (2005) Quantitative survey of the intertidal benthos of Farewell Spit, Golden Bay. *Marine Biodiversity Biosecurity Report No. 7*. 119 p.

Beaumont, J; MacDiarmid, A B; Morrison, M (2011) Rocky reef ecosystems – how do they function? Draft Final Research Report for Ministry of Fisheries project ZBD200509. 296 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Beentjes, M (2010) Toheroa survey of Oreti Beach, 2009, and review of historical surveys. *New Zealand Fisheries Assessment Report 2010/6*. 40 p.

Beentjes, M; Baird, S (2004) Review of dredge fishing technologies and practice for application in New Zealand. *New Zealand Fisheries Assessment Report 2004/37*. 40 p.

Beentjes, M, et al. (2005) Non-fishing mortality of freshwater eels (*Anguilla* spp.). *New Zealand Fisheries Assessment Report 2005/34*. 38 p.

Bentley, N, et al. (2004) A framework for evaluating spatial closures as a fisheries management tool. *New Zealand Fisheries Assessment Report 2004/25*. 25 p.

Berkenbusch, K, et al. (2013) New Zealand marine mammals and commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 119*. 104 p.

Black, J; Tilney, R (2015) Monitoring New Zealand's trawl footprint for deepwater fisheries: 1989–1990 to 2010–2011. *New Zealand Aquatic Environment and Biodiversity Report No. 142*. 56 p.

Black, J, et al. (2013) Monitoring New Zealand's trawl footprint for deepwater fisheries: 1989–1990 to 2009–2010. *New Zealand Aquatic Environment and Biodiversity Report No. 110*. 57 p.

Blackwell, R (2010) Distribution and abundance of deepwater sharks in New Zealand waters, 2000–01 to 2005–06. *New Zealand Aquatic Environment and Biodiversity Report No. 57*. 51 p.

Blackwell, R; Stevenson, M (2003) Review of the distribution and abundance of deepwater sharks in New Zealand waters. *New Zealand Fisheries Assessment Report 2003/40*. 48 p.

Booth, J, et al. (2002) Review of technologies and practices to reduce bottom trawl bycatch and seafloor disturbance in New Zealand. Unpublished Fisheries Research Assessment Report prepared for the Ministry of Fisheries as completion of ENV2000/06. 61 p.

Boren, L J; Clemens-Seely, K; Nicholas, G; Buckle, K; Lydon, G; Walker, N; Gooday, O (2016) Survey of New Zealand sea lion pups at Port Pegasus, Stewart Island. (Report held by Department of Conservation.)

Bowden, D (2011) Benthic invertebrate samples and data from the Ocean Survey 20/20 voyages to the Chatham Rise and Challenger Plateau, 2007. *New Zealand Aquatic Environment and Biodiversity Report No. 65*. 40 p.

Bowden D A; Clark, M R; Hewitt, J E; Rowden, A A; Leduc, D; Baird, S J (2015) Designing a programme to monitor trends in deep-water benthic communities. *New Zealand Aquatic Environment and Biodiversity Report No. 143*. 61 p.

Bowden, D; Hewitt, J (2012) Recommendations for surveys of marine benthic biodiversity: outcomes from the Chatham-Challenger Ocean Survey 20/20 Post-Voyage Analyses Project. *New Zealand Aquatic Environment and Biodiversity Report No. 91*. 34 p.

Bowden, D, et al. (2011a) A lost world? Archaic crinoid-dominated assemblages on an Antarctic seamount. *Deep Sea Research II 58*: 119–127.

Bowden, D, et al. (2011b) Evaluation of the New Zealand Marine Environment Classifications using Ocean Survey 20/20 data from Chatham Rise and Challenger Plateau. *New Zealand Aquatic Environment and Biodiversity Report No. 77*. 27 p.

Bowden, D, et al. (2014) Assessing the potential of multibeam echosounder data for predicting benthic invertebrate assemblages across Chatham Rise and Challenger Plateau. *New Zealand Aquatic Environment and Biodiversity Report No. 126*. 35 p.

Bowden, D, et al. (in prep) Population estimates of Ross Sea demersal fish: a comparison between video and trawl methods. (Submitted as short note to Fisheries Research).

Boyd, P; Law, C (2011) An Ocean Climate Change Atlas for New Zealand waters: A primer for a major new web-based tool to help predict how oceanic species will be affected by climate change. *NIWA Information Series No. 79*. 24 p.

Bradford, E (2002) Estimation of the variance of mean catch rates and total catches of non-target species in New Zealand fisheries. *New Zealand Fisheries Assessment Report 2002/54*. 60 p.

Bradford, E (2003) Factors that might influence the catch and discards of non-target fish species on tuna long-lines. *New Zealand Fisheries Assessment Report 2003/57*. 75 p.

Bradford-Grieve, J; Fenwick, G (2001a) A review of the current knowledge describing the biodiversity of the Balleny Islands. Unpublished Final Research Report to the Ministry of Fisheries.

Bradford-Grieve, J; Fenwick, G (2001b) A review of the current knowledge describing the biodiversity of the Ross Sea region. Unpublished Final Research Report to the Ministry of Fisheries.

Bradford-Grieve, J; Fenwick, G (2002) A review of the current knowledge describing the biodiversity of the Balleny Islands. Unpublished Final Research Report for Ministry of Fisheries project ZBD2000/01.

Bradford-Grieve, J; Livingston, M E (2011) Spawning fisheries and the productivity of the marine environment off the west coast of the South Island, New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 84*. 136 p.

Breen, P (2005) Managing the effects of fishing on the environment what does it mean for the rock lobster (*Jasus edwardsii*) fishery? *New Zealand Fisheries Assessment Report 2005/73*. 45 p.

Breen, P (2008) Sea lion data for use in the population model for Project IPA200609. Unpublished Final Research Report for Ministry of Fisheries. 18 p plus Confidential Appendix of 16 p.

Breen, P; Kim, S (2006) Exploring alternative management procedures for controlling bycatch of Hooker's sea lions in the SQU 6T squid fishery. Unpublished Final Research Report for Ministry of Fisheries project M0F2002/03L, Objective 3. Revision 5. 88 p.

Breen, P, et al. (2010) Sea lion population modelling and management procedure evaluations. Unpublished Report for Ministry of Fisheries project SAP2008/14, Objective 2: Presented to AEWG, 22 March 2010, Wellington, New Zealand.

Broom, J, et al. (2008) Utility of psbA and nSSU for phylogenetic reconstruction in the Corallinales based on New Zealand taxa. *Molecular Phylogenetics and Evolution 46*: 958–973.

Carroll, E, et al. (2015) Estimating nineteenth and twentieth century right whale catches and removals around east Australia and New Zealand. Unpublished Fisheries Research Report No. 4: 2 p.

Clark, M R; Anderson, O; Bowden, D; Chin, C; George, S; Glasgow, D; Guinotte, J; Hererra, S; Osterhage, D; Pallentin, A; Parker, S; Rowden, A; Rowley, S; Stewart, R; Tracey, D; Wood, S; Zeng, C (2015) Vulnerable Marine Ecosystems of the Louisville Seamount Chain: voyage report of a survey to evaluate the efficacy of preliminary habitat suitability models. *New Zealand Aquatic Environment and Biodiversity Report No. 149*. 86 p.

Clark, M; O'Driscoll, R (2003) Deepwater fisheries and aspects of their impact on seamount habitat in New Zealand. *Journal of Northwest Atlantic Fishery Science 31*: 441–458.

Clark, M; Roberts, C (2008) Fish and invertebrate biodiversity on the Norfolk Ridge and Lord Howe Rise, Tasman Sea (NORFANZ voyage, 2003). *New Zealand Aquatic Environment and Biodiversity Report No. 28*. 131 p.

Clark, M; Rowden, A (2009) Effect of deepwater trawling on the macro-invertebrate assemblages of seamounts on the Chatham Rise, New Zealand. *Deep Sea Research I 56*: 1540–1554.

Clark, M, et al. (submitted) Development of a methodology for ecological risk assessments for New Zealand deepwater fisheries. *New Zealand Aquatic Environment and Biodiversity Report*. 56 p.

Clark, M, et al. (2009) Voyage report of a survey of 'seamounts' on the northwest and southeast Chatham Rise (TAN0905). 49 p. (Unpublished report available from NIWA, Wellington.)

Clark, M, et al. (2010a) Effects of fishing on the benthic biodiversity of seamounts of the 'Graveyard' complex, northern Chatham Rise. *New Zealand Aquatic Environment and Biodiversity Report No. 46.* 40 p.

Clark, M, et al. (2010b) Latitudinal variation of demersal fish assemblages in the western Ross Sea. CCAMLR document WG-FSA-10/P3 22 (6): 782–792.

Clark, M, et al. (2010c) A global seamount classification to aid the scientific design of marine protected area networks. *Journal of Ocean and Coastal Management 54*: 19–36.

Clark, M, et al. (2011) Development of seamount risk assessment: application to the ERAEF approach to Chatham Rise seamount features. *New Zealand Aquatic Environment and Biodiversity Report No. 74*. 18 p.

Clarke, S, et al. (2013) Review of shark meat markets, discard mortality and pelagic shark data availability, and a proposal for a shark indicator analysis. *New Zealand Fisheries Assessment Report 2013/65*. 77 p.

Clement, D M; MacKenzie, D I (2016) Abundance and distribution of WCSI Hector's dolphin. *New Zealand Aquatic Environment and Biodiverstiy Report No. 168*. 67 p.

Clement, D; Mattlin, R (2010) Abundance, distribution and productivity of Hector's (and Māui's) dolphins. Unpublished Final Research Report for the Ministry of Fisheries.

Cole, R, et al. (2007) Information on benthic impacts in support of the Southern Blue Whiting Fishery Plan. Unpublished Final Research Report for Ministry of Fisheries project ZBD2005-16, Objectives 1–6.

Coleman, C; Lörz, A (2010) A new species of Camacho (Crustacea, Amphipoda, Aoridae) from the Chatham Rise, New Zealand. *Zeitschrift für Zoologische Systematik und Evolutionsforschung 86 (1)*: 33–40.

Compton, T, et al. (2012) Biophysical patterns in benthic assemblage composition across contrasting continental margins off New Zealand. *Journal of Biogeography 40 (1)*: 75–89. doi:10.1111/j.1365-2699.2012.02761.x.

Connell, A, et al. (2010) Diet and dietary variation of New Zealand hoki Macruronus novaezelandiae. *New Zealand Journal of Marine and Freshwater Research 44 (4)*: 289–308.

Constantine, R (2016) Humpback whale connectivity: Determining the migration path and Antarctic feeding grounds of New Zealand's humpback whales. Final Report to Ministry for Primary Industries project ZBD2013-08.

Cryer, M; Hartill, B (2002) Relative impacts on benthic invertebrate community structure of trawl fisheries for scampi and finfish in the Bay of Plenty. Unpublished Final Research Report for Ministry of Fisheries project ENV2000/05, Objective 2. 28 p.

Cryer, M, et al. (1999) Scampi stock assessment for 1998 and an analysis of the fish and invertebrate bycatch of scampi trawlers. *New Zealand Fisheries Assessment Research Document 99/4*. 74 p.

Cryer, M, et al. (2000) Distribution and structure of benthic invertebrate communities between North Cape and Cape Reinga Unpublished Final Research Report for Ministry of Fisheries project ENV1998-05, Objectives 1–4.

Cryer, M, et al. (2004) The effects of trawling and dredging on benthopelagic coupling processes in the New Zealand EEZ. Final Research Report. 66 p.

Cummings, V (2011) Ocean acidification: impacts on key NZ molluscs. Unpublished Progress Report for ZBD2009-13. 6 p.

Cummings, V; Lohrer, D (2011) Ross Sea ecosystem function: predicting consequences of shifts in food supply. Final Research Report for Ministry of Fisheries project ZBD2008-20. 53 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Cummings, V, et al. (2003) Ecology and biodiversity of coastal benthic communities in McMurdo Sound, Ross Sea: emerging results. Unpublished Final Research Report for the Ministry of Fisheries.

Cummings, V, et al. (2006a) Accounting for local scale variability in benthos: implications for future assessments of latitudinal trends in the coastal Ross Sea. CCAMLR document WG-FSA-10/P3 18 (4): 633–644.

Cummings, V, et al. (2006b) Ecology of coastal benthic communities of the northwestern Ross Sea. *New Zealand Aquatic Environment and Biodiversity Report No. 6*. 67 p.

Cummings, V, et al. (2008) Antarctic coastal marine ecosystems (ICECUBE) Unpublished Final Research Report for Ministry of Fisheries Research Project. 73 p.

Cummings, V, et al. (2010) Macrobenthic communities of the north western Ross Sea shelf: links to depth, sediment characteristics and latitude. CCAMLR document WG-FSA-10/P3 22 (6): 793–804.

Cummings, V, et al. (2011) Ocean acidification at high latitudes: potential effects on functioning of the Antarctic bivalve *Laternula elliptica*. *PLoS ONE 6 (1)*: e16069. doi:16010.11371/journal.pone.0016069.

Cummings, V, et al. (2013) Ocean acidification: impacts on key NZ molluscs, with a focus on flat oysters (*Ostrea chilensis*). Final Research Report for Ministry for Primary Industries project ZBD2009-13. 18 p.

Currey, R, et al. (2012) A risk assessment of threats to Māui's dolphins. Ministry for Primary Industries and Department of Conservation, Wellington. 51 p.

De Domenico, F, et al. (2006) Latitude versus local effects on echinoderm assemblages along the Victoria Land coast, Ross Sea, Antarctica. CCAMLR document WG-FSA-10/P3 18 (4): 655–662.

Dettai, A, et al. (2011) DNA barcoding and molecular systematics of the benthic and demersal organisms of the CEAMARC survey. *Polar Science 5 (2)*: 298–312.

Doonan, I (1995) Incidental catch of Hooker's sea lion in the southern trawl fishery for squid, summer 1994. *New Zealand Fisheries Assessment Research Document 95/22*. 13 p.

Doonan, I (1998) Estimation of sealion captures in southern fisheries in 1998. Final Research Report for Ministry of Fisheries project ENV1997/01, Objective 2. 7 p.

Doonan, I J (2000) Estimation of Hooker's sea lion, *Phocarctos hookeri*, captures in the southern squid trawl fisheries in 2000. *New Zealand Fisheries Assessment Report 2000/41*. 11 p.

Doonan, I (2001) Estimation of Hooker's sea lion, *Phocarctos hookeri*, captures in the southern squid trawl fisheries, 2001. *New Zealand Fisheries Assessment Report 2001/67*. 10 p.

Duckworth, K (1995) Analyses of factors which influence seabird bycatch in the Japanese southern bluefin tuna longline fishery in New Zealand waters, 1989–93. *New Zealand Fisheries Assessment Report 1995/26*. 59 p.

Dunn, M (2009) Feeding habits of the ommastrephid squid *Nototodarus sloanii* on the Chatham Rise, New Zealand. *Journal of Marine and Freshwater Research 43*: 1103–1113

Dunn, M, et al. (2009a) Fish abundance and climate trends in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 31*. 75 p.

Dunn, M, et al. (2009b) Ecosystem-scale trophic relationships: diet composition and guild structure of middle-depth fish on the Chatham Rise. New Zealand Final Research Report.

Dunn, M, et al. (2010a) Diet of Two Large Sympatric Teleosts, the Ling (*Genypterus blacodes*) and Hake (*Merluccius australis*). *PLoS ONE 5 (10)*: e13647. doi:13610.11371/journal.pone.0013647.

Dunn, M, et al. (2010b) Feeding habits and niche separation among the deep-sea chimaeroid fishes *Harriotta raleighana*, *Hydrolagus bemisi* and *Hydrolagus novaezealandiae*. *Marine Ecology Progress Series 407*: 209–225.

Dunn, M, et al. (2010c) The diet of deepwater sharks and the benefits of using DNA identification of prey. *Deep Sea Research I 57 (7)*: 923–930.

Eakin, R, et al. (2009) A new species and a molecular phylogenetic analysis of the Antarctic Fish Genus Pogonophryne (Notothenioidei: Artedidraconidae). *Copeia 4*: 705–713.

Edwards, C T T; Doonan, I; Anderson, O F (2015) Comparison of bycatch estimation for fish species using a ratio estimator and model-based method. *New Zealand Aquatic Environment and Biodiversity Report No. 154*. 30 p.

Edwards, C T T; Roberts, J O; Walker, K; Elliot, G (2017) Quantitative modelling of Antipodean wandering albatross. *New Zealand Aquatic Environment and Biodiversity Report No. 180*. 32 p.

Eléaume, M, et al. (2011) A large new species of the genus Ptilocrinus (Echinodermata, Crinoidea, Hyocrinidae) from Antarctic seamounts. *Polar Biology 34 (9)*: 1385–1397. doi:10.1007/s00300-011-0993-2.

Eléaume, M, et al. (in prep) Southern Ocean crinoid diversity patterns and processes. *Polar Biology*.

Farr, T, et al. (2009) Common coralline algae of northern New Zealand: an identification guide. *NIWA Information Series No. 70*. 120 p.

Fenaughty, J, et al. (2003) Diet of the Antarctic toothfish (*Dissostichus mawsoni*) from the Ross Sea, Antarctica (Subarea 88.1). CCAMLR Science 10: 113–123.

Fenwick, G; Bradford-Grieve, J (2002a) Human pressures on Ross Sea region marine communities: recommendations for future research. Unpublished Final Research Report for the Ministry of Fisheries.

Fenwick, G; Bradford-Grieve, J (2002b) Recommendations for future directed research to describe the biodiversity of the Ross Sea region. Unpublished Final Research Report for the Ministry of Fisheries.

Fletcher, D, et al. (2008) Modelling of impacts of fishing-related mortality on NZ seabird populations. Unpublished Final Research Report on ENV2004/05.

Floerl, O, et al. (2012) Chatham-Challenger Ocean Survey 20/20 Post Voyage analyses: Objective 9 – Patterns in Species Composition. *New Zealand Aquatic Environment and Biodiversity Report No. 97*. 40 p.

Ford, R B; Galland, A; Clark, M R; Crozier, P; Duffy, C A J; Dunn, M R; Francis, M P; Wells, R (2015) Qualitative (Level 1) Risk Assessment of the impact of commercial fishing on New Zealand Chondrichthyans. *New Zealand Aquatic Environment and Biodiversity Report No. 157*. 111 p.

Forman, J; Dunn, M (2010) The influence of ontogeny and environment on the diet of lookdown dory, *Cyttus traversi*. *New Zealand Journal of Marine and Freshwater Research 44 (4)*: 329–342.

Forrest, B, et al. (2007) Review of the Ecological Effects of Marine Finfish Aquaculture: Final Report. Unpublished Cawthron Report for the Ministry of Fisheries. 80 p.

Francis, M P (2015) Geographic distribution of commercial catches of cartilaginous fishes in New Zealand waters, 2008−13. *New Zealand Aquatic Environment and Biodiversity Report No. 156*. 15 p.

Francis, M; Lyon, W (2012) Review of research and monitoring studies on New Zealand sharks, skates, rays and chimaeras, 2008−2012. *New Zealand Aquatic Environment and Biodiversity Report No. 102*. 70 p.

Francis, M; Lyon, W (2013) Review of anthropogenic impacts other than fishing on cartilaginous fishes. *New Zealand Aquatic Environment and Biodiversity Report No. 107*. 17 p.

Francis, M; Smith, M (2010) Basking shark (*Cetorhinus maximus*) bycatch in New Zealand fisheries, 1994–95 to 2007–08. *New Zealand Aquatic Environment and Biodiversity Report No. 49*. 57 p.

Francis, M, et al. (1999) Fish bycatch in New Zealand tuna longline fisheries. *NIWA Technical Report 55*. 70 p.

Francis, M, et al. (2000) Fish bycatch in New Zealand tuna longline fisheries, 1988–89 to 1997–98. *NIWA Technical Report 76*. 79 p.

Francis, M, et al. (2001) Pelagic shark bycatch in the New Zealand tuna longline fishery. *Marine and Freshwater Research 52*: 165–178.

Francis, M, et al. (2004) Fish bycatch in New Zealand tuna longline fisheries, 1998–99 to 1999–2000. *New Zealand Fisheries Assessment Report 2004/22*. 62 p.

Francis, M, et al. (2012) Rig nursery grounds in New Zealand: a review and survey. *New Zealand Aquatic Environment and Biodiversity Report No. 95*. 50 p.

Francis, R (2012) Fisheries Risks to the Population Viability of White-capped Albatross *Thalassarche steadi*. *New Zealand Aquatic Environment and Biodiversity Report No. 104*. 24 p.

Francis, R; Bell, E (2010) Fisheries risks to the population viability of black petrel (*Procellaria parkinsoni*). *New Zealand Aquatic Environment and Biodiversity Report No. 51*. 57 p.

Francis, R I C C; Elliott, G; Walker, K (2015) Fisheries risks to the population viability of Gibson's wandering albatross *Diomedea gibsoni*. *New Zealand Aquatic Environment and Biodiversity Report No. 152*. 48 p.

Francis, R, et al. (2005) Environmental predictors of hoki year-class strengths: an update. *New Zealand Fisheries Assessment Report 2005/58*. 22 p.

Fu, D; Sagar, P (2016) The 2014 Quantitative modelling of southern Buller's albatross on the Snare Island. *New Zealand Aquatic Environment and Biodiversity Report No. 165.* 40 p.

Garcia, A (2010) Comparative study of the morphology and anatomy of octopuses of the family Octopodidae. Auckland University of Technology. 247 p.

Gardner, J, et al. (2010) Multi-species coastal marine connectivity: a literature review with recommendations for further research. *New Zealand Aquatic Environment and Biodiversity Report No. 58*: 47 p.

Ghiglione, J-F, et al. (2012) Pole to pole biogeography of surface and deep marine bacterial communities. *Proceedings of the National Academy of Sciences of the United States of America 109 (43)*: 17633–17638. doi:10.1073/pnas.1208160109.

Gordon, D (2000) The Pacific Ocean and global OBIS: a New Zealand perspective. *Oceanography 13*: 41–47.

Grange, K, et al. (2003) The spatial extent and nature of the bryozoan communities at Separation Point, Tasman Bay. *Marine Biodiversity Biosecurity Report No. 4*. 22 p.

Grayling, S (2004) A review of scientific studies conducted on the Macquarie Ridge. Unpublished Final Research Report for Ministry of Fisheries project ZBD2003-09. 33 p.

Griggs, L, et al. (2007) Fish bycatch in the New Zealand tuna longline fisheries 2002–03 to 2004–05. *New Zealand Fisheries Assessment Report 2007/18*. 58 p.

Griggs, L, et al. (2008) Fish bycatch in New Zealand tuna longline fisheries in 2005–06. *New Zealand Fisheries Assessment Report 2008/27*. 47 p.

Grotti, M, et al. (2008) Natural variability and distribution of trace elements in marine organisms from Antarctic coastal environments. CCAMLR document WG-FSA-10/P3 20 (1): 39–51.

Guidetti, M, et al. (2006) Exchange between populations of Adamussium colbecki (Mollusca: Bivalvia) in the Ross Sea. CCAMLR document WG-FSA-10/P3 18 (4): 645–653.

Hanchet, S (2000) Future research requirements for the Ross Sea Antarctic toothfish (*Dissostichus mawsoni*) fishery. Unpublished Final Research Report for MFish Research Project MOF2000/02A, Objective 8. 10 p.

Hanchet, S C (2009) New Zealand IPY-CAML Progress Report. 76 p.

Hanchet, S C (2010) New Zealand IPY-CAML Progress Report. 76 p.

Hanchet, S, et al. (2008a) Indicative estimates of biomass and yield of Whitson's grenadier (*M. whitsoni*) on the continental slope of the Ross Sea in Subareas 88.1 and 88.2. WG-FSA-08/32.

Hanchet, S, et al. (2008b) Ocean survey 20/20: New Zealand IPY-CAML Final Voyage Report. *NIWA Client Report WLG2008-74*. 193 p.

Hanchet, S, et al. (2008c) Preliminary report of the New Zealand RV Tangaroa IPY-CAML survey of the Ross Sea region, Antarctica in February–March 2008. Unpublished NIWA report WG-EMM-08/18 to CCAMLR working group on ecosystem monitoring and management. 15 p.

Hanchet, S, et al. (2008d) Biomass estimates and size distributions of demersal finfish on the Ross Sea shelf and slope from the New Zealand IPY-CAML survey, February–March 2008. WG-FSA-08/31.

Hanchet, S, et al. (2013) Diversity, relative abundance, new locality records, and updated fish fauna of the Ross Sea, Antarctica. CCAMLR document WG-FSA-10/P3 25: 619–636.

Hartill, B, et al. (2006) Reducing bycatch in New Zealand's scampi trawl fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 4*. 53 p.

Hartstein, N D; Oldman, J W (2015) Nitrogen Levels and Adverse Marine Ecological Effects from Aquaculture. *New Zealand Aquatic Environment and Biodiversity Report No. 159*. 77 p.

Harvey, A, et al. (2005) Coralline algae of central New Zealand: an identification guide to common 'crustose' species. *NIWA Information Series No. 57*.

Heimeier, D, et al. (2010) Using DNA barcoding and phylogenetics to identify Antarctic invertebrate larvae: Lessons from a large scale study. *Marine Genomics 3*: 165–177.

Hemery, L, et al. (in prep) Circumpolar distribution of a complex of cryptic species.

Hewitt, J E; Bell, R; Costello, M; Cummings, V; Currie, K; Ellis, J; Francis, M; Froude, V; Gorman, R; Hall, J; Inglis, G; MacDiarmid, A; Mills,

G; Pinkerton, M; Schiel, D; Swales, A; Law, C; McBride, G; Nodder, S; Rowden, A; Smith, M; Thompson, D; Torres, L; Tuck, I; Wing, S (2014) Development of a National Marine Environment Monitoring Programme (MEMP) for New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 141*. 126 p.

Hewitt, J, et al. (2011a) Chatham-Challenger Ocean Survey 20/20 Post-Voyage Analyses: Objective 10 – Biotic habitats and their sensitivity to physical disturbance. *New Zealand Aquatic Environment and Biodiversity Report No. 81*. 36 p.

Hewitt, J, et al. (2011b) Chatham-Challenger Ocean Survey 20/20 Post Voyage Analyses: Diversity Metrics. *New Zealand Aquatic Environment and Biodiversity Report No. 83*. 64 p.

Horn, P; Dunn, M (2010) Inter-annual variability in the diets of hoki, hake, and ling on the Chatham Rise from 1990 to 2009. *New Zealand Aquatic Environment and Biodiversity Report No. 54*. 57 p.

Horn, P, et al. (2010) Feeding habits of alfonsino Beryx splendens. *Journal of Fish Biology 76*: 2382–2400.

Horn, P, et al. (2013) Evaluation of the diets of highly migratory species in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 116*. 141 p.

Howard, S (2015) Mitigation options for shark bycatch in longline fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 148*. 47 p.

Hurst, R, et al. (2000) Areas of importance for spawning, pupping or egg-laying and juveniles of New Zealand coastal fish. Unpublished Final Research Report for Ministry of Fisheries project ENV1999/03, Objective 1. 56 p.

Hurst, R, et al. (2012) Climate and ocean trends of potential relevance to fisheries in the New Zealand Region, 2010. *New Zealand Aquatic Environment and Biodiversity Report No. 90*. 202 p.

Jellyman, D (1994) The fishery for freshwater eels (*Anguilla* spp.) in New Zealand. *New Zealand Fisheries Assessment Research Document 94/14*. 25 p.

Jones, E G, et al. (2016) Biogenic habitats on New Zealand's continental shelf. Part 1: Local Ecological Knowledge. *New Zealand Aquatic Environment and Biodiversity Report No. 174*. 113 p.

Jones, E G, et al. (submitted) Research on biogenic habitat-forming biota and their functional role maintaining biodiversity in the Inshore Region (~5–~150 m depths). Draft New Zealand Aquatic Environment and Biodiversity Report.

Keeley, N, et al. (2009) Sustainable aquaculture in New Zealand: Review of the Ecological Effects of farming shellfish and other Non-finfish species. *Cawthron Report No. 1476*, prepared for the Ministry of Fisheries. 150 p. plus appendices.

Key, J (2002) A review of current knowledge describing New Zealand's deepwater benthic biodiversity. *Marine Biodiversity Biosecurity Report No. 1*. 25 p.

Koubbi, P, et al. (2011) Ecological importance of micronektonic fish for the ecoregionalisation of the Indo-Pacific sector of the Southern Ocean: role of myctophids. *Deep Sea Research II 58*: 170–180.

Lalas, C; MacDiarmid, A (2014) Rapid recolonisation of south-eastern New Zealand by New Zealand fur seals *Arctocephalus forsteri*. *New Zealand Fisheries Research Report 2014/13*.

Lalas, C, et al. (2014) Estimates for annual consumption rates by New Zealand sea lions. *New Zealand Fisheries Research Report. 2014/11*.

Lavery, S, et al. (2006) Molecular identification of cryptogenic/invasive gobies in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 5*. 48 p.

Law, C, et al. (2012) No stimulation of nitrogen fixation by non-filamentous diazotrophs under elevated CO2 in the South Pacific. *Global Change Biology 18*: 3004–3014. doi:3010.1111/j.1365-2486.2012.02777.x.

Leathwick, J, et al. (2006a) Development of a marine environmental classification optimised for demersal fish. *NIWA Client Report HAM2006-063*.

Leathwick, J, et al. (2006b) Variation in demersal fish species richness in the oceans surrounding New Zealand: an analysis using boosted regression trees. *Marine Ecology Progress Series 321*: 267–281.

Leathwick, J, et al. (2006c) Development of a demersal fish community map for New Zealand's Exclusive Economic Zone. *NIWA Client Report HAM2006-062*, prepared for Department of Conservation, Hamilton, New Zealand.

Leathwick, J, et al. (2010) Development of a benthic-optimised marine environment classification for waters within the New Zealand EEZ Unpublished Final Research Report for Ministry of Fisheries project BEN200601, Objective 5.

Leathwick, J, et al. (2012) A Benthic-optimised Marine Environment Classification (BOMEC) for New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 88*. 54 p.

Leduc, D, et al. (2012a) Direct evidence for a unimodal relationship between productivity and deep-sea benthic diversity. *Marine Ecology Progress Series 454*: 53–64.

Leduc, D, et al. (2012b) A unimodal relationship between biomass and species richness of deep-sea nematodes: implications for the link between productivity and diversity. *Marine Ecology Progress Series 454*: 53–64.

Leduc, D, et al. (2012c) Nematode beta diversity on the continental slope of New Zealand: spatial patterns and environmental drivers. *Marine Ecology Progress Series 545*: 37–52.

Leduc, D, et al. (2013) Is there a link between deep-sea biodiversity and ecosystem function? *Marine Ecology 34*: 334–344.

Leduc, D, et al. (2014) Unusually high food availability in Kaikoura Canyon linked to distinct deep-sea nematode community. *Deep Sea Research II 104*: 310–318.

Linse, K, et al. (2007) Is the Scotia Sea a centre of Antarctic marine diversification? Some evidence of cryptic speciation in the circum-Antarctic bivalve *Lissarca notorcadensis* (Arcoidea: Philobryidae). *Polar Biology 30*: 1059–1068.

Livingston, M (2004) A sampling programme to construct and quantify food-webs in two key areas supporting important fish and invertebrate species in New Zealand. Unpublished Final Research Report for Ministry of Fisheries project ENV2002-07, Objective 1.

Livingston, M (2009) Towards a National Marine Environment Monitoring Programme in New Zealand: A discussion paper submitted to the Biodiversity Research Advisory Group Workshop on Marine Environmental Monitoring. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Livingston, M, et al. (2003) Trends in incidental catch of major fisheries on the Chatham Rise for fishing years 1989–90 to 1998–99. *New Zealand Fisheries Assessment Report 2003/52*. 74 p.

Lohrer, A, et al. (2010) Isolated and interactive effects of two key species on ecosystem function and trophic linkages in New Zealand soft-sediment habitats. *New Zealand Aquatic Environment and Biodiversity Report No. 44*. 69 p.

Lohrer, A, et al. (2013) Altered sea ice thickness and permanence affects benthic ecosystem functioning in coastal Antarctica. *Ecosystems 16 (2)*: 224–236.

Lorrey, A, et al. (2013) A synthesis of climatic and geophysical driver activity in New Zealand and environmental changes during the last 1000 years. Unpublished Final Research Report for Ministry for Primary Industries project ZBD2005-05. Wellington, New Zealand. 23 p.

Lörz, A (2009) Synopsis of Amphipoda from two recent Ross Sea voyages with description of a new species of Epimeria (Epimeriidae, Amphipoda, Crustacea). *Zootaxa 2167*: 59–68.

Lörz, A (2010a) Deep-sea Rhachotropis (Crustacea: Amphipoda: Eusiridae) from New Zealand and the Ross Sea with key to the Pacific, Indian Ocean and Antarctic species. *Zootaxa 2482*: 22–48.

Lörz, A (2010b) New Zealand Amphipoda, Conference presentation. XIVth International Colloquium on Amphipoda, Seville, Spain.

Lörz, A (2011a) Biodiversity of unknown New Zealand habitat: bathyal invertebrate assemblages in the benthic boundary layer. *Marine Biodiversity 41 (2)*: 287–297.

Lörz, A (2011b) Pacific Epimeriidae (Amphipoda, Crustacea): Epimeria. *Journal of the Marine Biological Association of the United Kingdom 91 (2)*: 471–477.

Lörz, A N; Coleman, O (2009) Living jems: jewel-like creatures from the deep. *Water and Atmosphere 17 (1)*: 16–17.

Lörz, A, et al. (2007) *Epimeria schiaparelli* sp. nov, an amphipod crustacean (family Epimeriidae) from the Ross Sea, Antarctica, with molecular characterisation of the species complex. *Zootaxa 1402*: 23–37.

Lörz, A, et al. (2009) Do circum-Antarctic species exist in peracarid Amphipoda? A case study in the genus Epimeria Costa, 1851 (Crustacea, Peracarida, Epimeriidae). *In*: Bruce, N (Ed.), Advances in the taxonomy and biogeography of Crustacea in the Southern Hemisphere. *Zookeys 18*: 91–128.

Lörz, A, et al. (2012a) First Molecular Evidence for Underestimated Biodiversity of Rhachotropis (Crustacea, Amphipoda), with Description of a New Species. *PLoS ONE 7 (3)*: e32365. doi:32310.31371/journal.pone.0032365.

Lörz, A-N, et al. (2012b) A review of deep-sea biodiversity associated with trench, canyon and abyssal habitats deeper than 1500 m in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 92*. 133 p.

Lörz, A, et al. (2012c) High genetic diversity within Epimeria georgiana (Amphipoda) from the southern Scotia Arc. *Marine Biodiversity 42 (2)*: 137–159.

Lörz, A, et al. (in prep) Molecular insights to deep-sea Rhachotropis from New Zealand and the Ross Sea with description of a new species.

Lundquist, C J; Julian, K; Costello, M; Gordon, D; Mackay, K; Mills, S; Neill, K; Nelson, W; Thompson, D (2015) Development of a Tier 1 National Reporting Statistic for New Zealand's Marine Biodiversity. *New Zealand Aquatic Environment and Biodiversity Report No. 147*. 61 p.

Lundquist, C, et al. (2010) Interactions between distributions and dispersal decrease persistence thresholds of a marine benthic community. *Marine Ecology Progress Series 413*: 217–228.

Lundquist, C, et al. (2013) Bottom disturbance and seafloor community dynamics: Development of a model of disturbance and recovery dynamics for marine benthic ecosystems. *New Zealand Aquatic Environment and Biodiversity Report No. 118*. 59 p.

Lyle, J (2011) SRP2010-03: Fur seal interactions with SED excluder device. Unpublished Final Research Report for Ministry of Fisheries. 20 p.

Maas, E, et al. (2010a) Bacterial and Archaeal diversity and exo-enzyme activity in Ross Sea, Antarctica sediments. Poster presented at SAME 11 – Symposium on Aquatic Microbial Ecology, Piran, Slovenia, 30 August–4 September 2010.

Maas, E, et al. (2010b) Impacts of ocean acidification on planktonic ecosystems of the southern ocean. Final Research Report for Ministry of Fisheries project ZBD2008-22. 44 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

McCowan, T (2013) Genetic approaches to management of New Zealand's blackfoot pāua, *Haliotis iris*. Unpublished Final Research Report for Project ENH2007-01, Ministry for Primary Industries, Wellington, New Zealand. 271 p.

MacDiarmid, A; Stewart, R (2015) Ross Sea and Balleny Islands biodiversity: routine observations and opportunistic sampling of biota made during a geophysical survey to the Ross Sea in 2006. *New Zealand Aquatic Environment and Biodiversity Report No. 153*. 97 p.

MacDiarmid, A, et al. (2012) Assessment of anthropogenic threats to New Zealand marine habitats. *New Zealand Aquatic Environment and Biodiversity Report No. 93*. 255 p.

MacDiarmid, A, et al. (2016a) Taking Stock – the impacts of humans on New Zealand marine ecosystems since first settlement: synthesis of major findings, and policy and management implications. *New Zealand Aquatic Environment and Biodiversity Report No. 170*. 48 p.

MacDiarmid, A, et al. (2016b) Top-down effects on rocky reef ecosystems in north-eastern New Zealand: a historic and qualitative modelling approach. *New Zealand Aquatic Environment and Biodiversity Report No. 171*. 20 p.

MacDiarmid, A, et al. (2018) Historical evidence for the state and exploitation of marine fish and invertebrate resources in the Hauraki Gulf and along the Otago-Catlins shelf 1769–1950. *New Zealand Aquatic Environment and Biodiversity Report No. 194*. 69 p + 520 p (Supplement).

MacDiarmid, A, et al. (submitted) A complete history of the exploitation of an ecologically important inshore finfish species in the Hauraki Gulf, New Zealand: a synthesis of archaeological, historical and fisheries data. Draft New Zealand Aquatic Environment and Biodiversity Report.

McElderry, H, et al. (2007) Electronic Monitoring to Assess Protected Species Interactions in New Zealand Longline Fisheries: A Pilot Study. Unpublished report prepared for the New Zealand Ministry of Fisheries, Wellington, by Archipelago Marine Research Ltd, Victoria, British Columbia, Canada. 45 p.

McElderry, H, et al. (2008) Electronic monitoring to assess protected species interactions in New Zealand longline fisheries: a pilot study. *New Zealand Aquatic Environment and Biodiversity Report No. 24*. 39 p.

McGregor, V; Horn, P L (2015) Factors affecting the distribution of highly migratory species in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No 146*. 119 p.

MacKenzie, D; Clement, D (2014) Abundance and distribution of ECSI Hector's dolphin. *New Zealand Aquatic Environment and Biodiversity Report No. 123*. 79 p.

MacKenzie, D; Fletcher, D (2006) Characterisation of seabird captures in NZ fisheries. Unpublished Final Research Report prepared for the Ministry of Fisheries, Proteus Wildlife Research Consultants. 99 p.

MacKenzie, D; Fletcher, D (2010) Designing a programme of indicators of population performance with respect to captures of seabirds in New Zealand fisheries. Final Research Report for Ministry of Fisheries project ENV2005-08. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

MacKenzie, D, et al. (2013) Abundance, distribution, and productivity of Hector's (and Māui's) dolphins. Unpublished Final Research Report for Ministry for Primary Industries project PRO2009-01B. 121 p.

McKenzie, J, et al. (2013) Can juvenile yellowbelly and sand flounder abundance indices and environmental variables predict adult abundance in the Manukau and Mahurangi Harbours? *New Zealand Fisheries Assessment Report 2013/10*. 31 p.

McMillan, P, et al. (2007) Field identification guide to the main fishes caught in the Ross Sea long-line fishery. Unpublished Final Research Report for Ministry of Fisheries project ANT2005/02, Objective 7. 14 p.

McMillan, P, et al. (2011a) New Zealand fishes. 1: A guide to common species caught by trawling. *New Zealand Aquatic Environment and Biodiversity Report No. 68*. 331 p.

McMillan, P, et al. (2011b) New Zealand fishes. 2: A guide to less common species caught by trawling. *New Zealand Aquatic Environment and Biodiversity Report No. 78.* 184 p.

McMillan, P, et al. (2011c) New Zealand fishes. 3: A guide to common species caught by surface fishing methods. *New Zealand Aquatic Environment and Biodiversity Report No. 69*. 147 p.

McMillan, P, et al. (2012) A new species of grenadier, genus Macrourus (Teleostei, Gadiformes, Macrouridae) from the Antarctic and a revision of the genus. *Zootaxa 3165*: 1–24.

Marriot, P, et al. (2003) Species identification and age estimation for the ridge-scaled Macrourid (*Macrourus whitsoni*) the Ross Sea. *CCAMLR Science 10*: 37–51.

Mattlin, R H (1993) Incidental catch of fur seals in the west coast South Island hoki trawl fishery, 1989–92. *New Zealand Fisheries Assessment Research Document 93/19*. 17 p.

Mattlin, R (2004) QMA SQU6T New Zealand sea lion incidental catch and necropsy data for the fishing years 2000–01, 2001–02 and 2002–03. Unpublished Report prepared for the Ministry of Fisheries, Wellington. 21 p.

Maunder, M (2007) Assessment of risk of yellow-eyed penguins *Megadyptes antipodes* from fisheries incidental mortality in New Zealand fisheries and definition of information requirements for managing fisheries related risk. Unpublished Final Research Report for the Ministry of Fisheries. 29 p.

Maxwell, K; MacDiarmid, A B (2016) Taking Stock: Oral histories about changes in marine fauna observed locally in the Hauraki Gulf and along the Otago-Catlins coast, New Zealand, 1926–2008. *New Zealand Aquatic Environment and Biodiversity Report No. 173*. 45 p.

Meynier, L (2010) New Zealand sea lion bio-energetic modelling. Unpublished Final Report for project IPA2009-09, presented to AEWG, 17 May 2011, Wellington, New Zealand. 34 p.

Michael, K, et al. (2006) Summary of information in support of the Foveaux Strait Oyster Fishery Plan: the Foveaux Strait ecosystem and effects of oyster dredging. Unpublished Final Research Report for Ministry of Fisheries project ZBD2005-04.

Middleton, D; Abraham, E (2007) The efficacy of warp strike mitigation devices: trials in the 2006 squid fishery. Unpublished Final Research Report for Ministry of Fisheries project IPA2006-02.

Mills, S, et al. (2013) Identification of benthic invertebrate samples from research trawls and observer trips 2012–2013. Unpublished Final Research Report for the Ministry of Fisheries. 62 p.

Ministry of Fisheries (2008) Bottom Fishery Impact Assessment: Bottom Fishing Activities by New Zealand Vessels Fishing in the High Seas in the SPRFMO Area during 2008 and 2009. 102 p. Retrieved from http://www.southpacificrfmo.org/benthic-impact-assessments.

Mitchell, J (2008) Initial environmental evaluation (IEE) NZ IPY-CAML voyage 2008 report. Unpublished Report prepared for Antarctic Policy Unit, Ministry of Foreign Affairs and Trade. 35 p.

Mitchell, J; Clark, M (2004) Voyage report TAN04-02. Western Ross Sea voyage 2004. Hydrographic and biodiversity survey RV Tangaroa 27 January to 13 March 2004. Cape Adare, Cape Hallett, Possession Islands and Balleny Islands, Antarctica. *NIWA Voyage Report TAN04-02*. 102 p. (Unpublished report held in NIWA library, Wellington.)

Mitchell, J; MacDiarmid, A (2006) Voyage Report TAN06-02 Eastern Ross Sea Voyage. Unpublished Report for Ministry of Fisheries project ZBD2005-03.

Mormede, S; Dunn, A (2013) An initial development of spatially explicit population models of benthic impacts to inform Ecological Risk Assessments in New Zealand deepwater fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 106*. 16 p.

Mormede, S, et al. (2008) Factors that may influence the probability of fur seal capture in selected New Zealand fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 19*. 42 p.

Morrison, M; Parkinson, D (2008) Distribution and abundance of toheroa (*Paphies ventricosa*) on Ninety Mile Beach, March 2006. *New Zealand Fisheries Assessment Report 2008/26*. 27 p.

Morrison, M, et al. (2009) A review of land-based effects on coastal fisheries and supporting biodiversity in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 37*. 100 p.

Morrison, M, et al. (2014a) Linking marine fisheries species to biogenic habitats in New Zealand: a review and synthesis of knowledge. *New Zealand Aquatic Environment and Biodiversity Report No. 130*. 156 p.

Morrison, M, et al. (2014b) Habitats and areas of particular significance for coastal finfish fisheries management in New Zealand: A review of concepts and life history knowledge, and suggestions for future research. *New Zealand Aquatic Environment and Biodiversity Report No. 125*. 202 p.

Morrison, M, et al. (2014c) Seagrass meadows as biodiversity and productivity hotspots. *New Zealand Aquatic Environment and Biodiversity Report No. 137*. 147 p.

Morrison, M, et al. (2014d) Habitats of particular significance for fisheries management: the Kaipara Harbour. *New Zealand Aquatic Environment and Biodiversity Report No. 129*. 169 p.

Morrison, M, et al. (2016) Can otolith chemistry predict the natal origin of grey mullet (*Mugil cephalus*)? *New Zealand Fisheries Assessment Report 2016/15*. 68 p.

Murray, T, et al. (1992) Seabird bycatch by Southern Fishery longline vessels in New Zealand waters. *New Zealand Fisheries Assessment Research Document 92/22*. 21 p.

Naylor, J, et al. (2005) A guide to common offshore crabs in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 2*. 47 p.

Neil, H, et al. (2012) Insights into historical New Zealand marine shelf productivity using ancient fish otoliths. Unpublished Final Research Report for Ministry of Fisheries project ZBD200505 MS6 - Part B.

Neill, K, et al. (2012) Macroalgal diversity associated with soft sediment habitats in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 87*. 130 p.

Nelson, W, et al. (2002) Marine Macroalgae of Fiordland, New Zealand. *Tuhinga 13*: 117–152.

Nelson, W, et al. (2010) Macroalgae and benthic biodiversity of the Balleny Islands, Southern Ocean. *New Zealand Aquatic Environment and Biodiversity Report No. 55*. 99 p.

Nelson, W, et al. (2012) Rhodolith Beds in Northern New Zealand: Characterisation of Associated Biodiversity and Vulnerability to Environmental Stressors. *New Zealand Aquatic Environment and Biodiversity Report No. 99*. 102 p.

Nielsen, J, et al. (2009) Synopsis of a new collection of sea spiders (Arthropoda: Pycnogonida) from the Ross Sea, Antarctica. *Polar Biology 32*: 1147–1155.

Nodder, S (2008) OS 20/20 Chatham Rise & Challenger Plateau Hydrographic, Biodiversity & Seabed Habitats. *NIWA Client Report WLG2008-27*. NIWA, Wellington, New Zealand.

Nodder, S, et al. (2011) Physical, biogeochemical, and microbial characteristics of sediment samples from the Chatham Rise

and Challenger Plateau. *New Zealand Aquatic Environment and Biodiversity Report No. 70*. 39 p.

Nodder, S, et al. (2012) Seafloor habitats and benthos of a continental ridge: Chatham Rise, New Zealand. *In*: Harris, P; Baker, E (Eds.), Seafloor geomorphology as benthic habitat: GEOHAB atlas of geomorphic features and benthic habitats. pp 763–776. Elsevier, London.

Norkko, A, et al. (2002) Ecology and biodiversity of coastal benthic communities in McMurdo Sound, Ross Sea: development of sampling protocols and initial results. Unpublished Final Research Report for Ministry of Fisheries Research Project ZBD2001/02, Objectives 1, 2 & 3. 119 p.

Norkko, A, et al. (2004) Ecological role of *Phyllophora antarctica* drift accumulations in coastal soft-sediment communities of McMurdo Sound, Antarctica. *Polar Biology 27*: 482–494.

Norkko, J, et al. (2005) Growth under environmental extremes: spatial and temporal patterns in nucleic acid ratios in two Antarctic bivalves. *Journal of Experimental Marine Biology and Ecology 326*: 114–156.

Norkko, A, et al. (2007) Trophic structure of coastal Antarctic food webs associated with changes in sea ice and food supply. *Ecology 88*: 2810–2820.

O'Driscoll, R (2009) Preliminary acoustic results from the New Zealand IPY-CAML survey of the Ross Sea region in February–March 2008. Unpublished Final Research Report for Ministry of Fisheries project IPY200701, Objective 8. 14 p.

O'Driscoll, R, et al. (2003) Areas of importance for spawning, pupping or egg-laying, and juveniles of New Zealand deepwater fish, pelagic fish, and invertebrates. *NIWA Technical Report 119*. 377 p.

O'Driscoll, R, et al. (2009) Preliminary acoustic results from the New Zealand IPY-CAML survey of the Ross Sea region in February–March 2008. CCAMLR SG-ASAM-09-05. 37 p.

O'Driscoll, R, et al. (2010) Distribution, abundance and acoustic properties of Antarctic silverfish (*Pleuragramma antarcticum*) in the Ross Sea. *Deep-Sea Research II 58 (1–2)*: 181–195.

O'Driscoll, R, et al. (2011) Trends in relative mesopelagic biomass using time series of acoustic backscatter data from trawl surveys. *New Zealand Aquatic Environment and Biodiversity Report No. 96*. 99 p.

O'Driscoll, R, et al. (2012) Can acoustic methods be used to monitor grenadier (Macrouridae) abundance in the Ross Sea region. *Journal of Ichthyology 53*: 700–708.

O'Loughlin, M, et al. (2011) The Antarctic region as a marine biodiversity hotspot for echinoderms: Diversity and diversification of sea cucumbers. *Deep Sea Research II 58*: 264–275.

Opresko, D, et al. (2014) ANTIPATHARIA (BLACK CORALS) FOR THE NEW ZEALAND REGION. A field guide of commonly sampled New Zealand black corals including illustrations highlighting technical terms and black coral morphology. *New Zealand Aquatic Enviornment and Biodiversity Report No. 136*. 20 p.

Page, M, et al. (2001) Preliminary assessment of the biodiversity of benthic macrofauna of the western Ross Sea, Antarctica. Unpublished Final Research Report to the Ministry of Fisheries. 29 p.

Pakhomov, E, et al. (2011) Biology of *Salpa thompsoni* in waters adjacent to the Ross Sea, Southern Ocean, during austral summer 2008. *Polar Biology 34*: 257–271.

Parker, S (2008) Development of a New Zealand High Seas Bottom Trawling Benthic Assessment Standard for Evaluation of Fishing Impacts to Vulnerable Marine Ecosystems in the South Pacific Ocean. Unpublished Final Research Report for Ministry of Fisheries project IFA2007-02, Objectives 3 and 4.

Parker, S; Bowden, D (2010) Identifying taxonomic groups vulnerable to bottom longline fishing gear in the Ross Sea region. *CCAMLR Science 17*: 105–127.

Parker, S; Francis, M (2012) Productivity of two species of deepwater sharks, *Deania calcea* and *Centrophorus squamosus* in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 103*. 48 p.

Parker, S, et al. (2008) Classification guide for potentially vulnerable invertebrate taxa in the Ross Sea long-line fishery. Unpublished Final Research Report for Ministry of Fisheries project ANT2007-01, Objective 3. 7 p.

Parker, S, et al. (2009a) Detection criteria for measuring trawl impacts on vulnerable marine ecosystems in high sea fisheries of the South Pacific Ocean. *Marine Ecology Progress Series 397*: 309–317.

Parker, S, et al. (2009b) Evaluation of VME taxa monitoring by scientific observers from five vessels in the Ross Sea Antarctic toothfish longline fishery during the 2008–09 season. Unpublished Final Research Report for the Ministry of Fisheries.

Parker, S, et al. (2009c) CCAMLR VME taxa Identification guide. Version 2009. Hobart, Tasmania, Australia. 4 p. Retrieved from http://www.ccamlr.org/pu/e/sc/obs/vme-guide.pdf.

Parker, S, et al. (2010) A bathymetric data framework for fisheries management in the Ross Sea region. Unpublished Final Research Report for the Ministry of Fisheries. 14 p.

Parsons, D, et al. (2011) Long-term effects of climate variation and human impacts on the structure and functioning of New Zealand's Shelf Ecosystems. Unpublished Final Research Report for the Ministry for Primary Industries.

Paul, L (2012) A history of the Firth of Thames dredge fishery for mussels: use and abuse of a coastal resource. *New Zealand Aquatic Environment and Biodiversity Report No. 94*. 27 p.

Paul, L (2014) History of and trends in the commercial landings of finfish from the Hauraki Gulf, 1850–2006. *New Zealand Aquatic Environment and Biodiversity Report No. 124*. 178 p.

Penney, A (2014) Review of the biodiversity component of the New Zealand Vulnerable Marine Ecosystem Evidence Process. *New Zealand Aquatic Environment and Biodiversity Report No. 135*. 40 p.

Penney, A, et al. (2009) Protection measures implemented by New Zealand for vulnerable marine ecosystems in the South Pacific Ocean. *Marine Ecology Progress Series 397*: 341–354.

Pierre, J P; Richard, Y; Abraham, E R (2015) Assessment of cryptic seabird mortality due to trawl warps and longlines. Report prepared by Dragonfly Data Science for the New Zealand Department of Conservation, Wellington. 51 p.

Pinkerton, M H; Bell, R; Chiswell, S M; Currie, K; Mullan, A B; Rickard, G; Stevens, C; Sutton, P (2015a) Reporting on the state of the New Zealand marine environment: recommendations for ocean indicators as part of the Atmospheric and Ocean Climate Change Tier 1 Statistic. *New Zealand Aquatic Environment and Biodiversity Report No. 151*. 85 p.

Pinkerton, M H; MacDiarmid, A; Beaumont, J; Bradford-Grieve, J; Francis, M P; Jones, E; Lalas, C; Lundquist, C J; McKenzie, A; Nodder, S D; Paul, L; Stenton-Dozey, J; Thompson, D; Zeldis, J (2015b) Changes to the food-web of the Hauraki Gulf during the period of human occupation: a mass-balance model approach. *New Zealand Aquatic Environment and Biodiversity Report No. 160*. 346 p.

Pinkerton, M, et al. (2005) Modelling the effects of fishing in the Ross Sea. Unpublished Final Research Report for Ministry of Fisheries project ANT2004/05. 130 p.

Pinkerton, M, et al. (2006) Modelling the effects of fishing in the Ross Sea. Unpublished Final Research Report for Ministry of Fisheries project ANT2004–05. 169 p.

Pinkerton, M, et al. (2007a) Stable isotope analysis of Southern Ocean fish tissue samples: preliminary results. Unpublished Final Research Report for Ministry of Fisheries project ANT2005/04, Objective 1. 32 p.

Pinkerton, M, et al. (2007b) Ecological risk management and the fishery for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea, Antarctica. CCAMLR Document EMM-07-24, Hobart, Australia. 22 p.

Pinkerton, M, et al. (2009a) Inputs to the Ross Sea Bioregionalisation/Spatial Management planning workshop. Unpublished Final Research Report to the Ministry of Fisheries. 40 p.

Pinkerton, M, et al. (2009b) Trophic connections in the Ross Sea: information from stomach contents analysis and stable isotopes of carbon and nitrogen. Unpublished Report for Ministry of Fisheries project IPY200701, Objective 10. 18 p.

Pinkerton, M, et al. (2010) A balanced model of the food web of the Ross Sea, Antarctica. *CCAMLR Science 17*: 1–31.

Pinkerton, M, et al. (2013) Diet and trophic niche of Antarctic silverfish (Pleuragramma antarcticum) in the Ross Sea, Antarctica. *Journal of Fish Biology 82 (1)*: 141–164.

Ponte, G, et al. (2010) Impact characteristics of the New Zealand Fisheries sea lion exclusion device stainless steel grid. Report prepared for Ministry of Fisheries project IPA2009-16. 24 p.

Ponte, G, et al. (2011) Further analysis of the impact characteristics of the New Zealand Fisheries sea lion exclusion device stainless steel grid. Unpublished Report prepared for Ministry of Fisheries project SRP2010-05. 24 p.

Richard, Y; Abraham, E (2013a) Application of Potential Biological Removal methods to seabird populations. *New Zealand Aquatic Environment and Biodiversity Report No. 108*. 30 p.

Richard, Y; Abraham, E (2013b) Risk of commercial fisheries to New Zealand seabird populations. *New Zealand Aquatic Environment and Biodiversity Report No. 109*. 58 p.

Richard, Y; Abraham, E (2013c) Risk of commercial fisheries to New Zealand seabird populations (supplementary material). *New Zealand Aquatic Environment and Biodiversity Report No. 117*. 78 p.

Richard, Y; Abraham, E (2015) Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006–07 to 2012–13. *New Zealand Aquatic Environment and Biodiversity Report No. 162*. 85 p.

Richard, Y; Abraham, E (2017) Sensitivity of the Seabird Risk Assessment to three years without captures. *New Zealand Aquatic Environment and Biodiversity Report No. 182*. 8 p.

Richard, Y; Abraham, E R; Berkenbusch, K (2017) Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006–07 to 2014–15. *New Zealand Aquatic Environment and Biodiversity Report No. 191*. 112 p. plus Supplement (148 p).

Richard, Y, et al. (2011) Assessment of the risk to seabird populations from New Zealand commercial fisheries. (Unpublished Final Research Report held by the Ministry for Primary Industries, Wellington.)

Roberts, C; Stewart, A (2001) Ross Sea fishes: a collection-based biodiversity research programme. *Seafood New Zealand 9 (11)*: 79–84.

Robinson, V, et al. (2014) Continuous Plankton Recorder Time Series. *New Zealand Aquatic Environment and Biodiversity Report No. 128*. 76 p.

Roe, W (2007) Necropsy of marine mammals captured in New Zealand fisheries in the 2005–06 fishing year. *New Zealand Aquatic Environment and Biodiversity Report No. 11*. 24 p.

Roe, W (2009a) Necropsy of marine mammals captured in New Zealand fisheries in the 2007–08 fishing year. *New Zealand Aquatic Environment and Biodiversity Report No. 34*. 35 p.

Roe, W (2009b) Necropsy of marine mammals captured in New Zealand fisheries in the 2006–07 fishing year. *New Zealand Aquatic Environment and Biodiversity Report No. 29*. 32 p.

Roe, W (2010a) External review of NZ sea lion bycatch necropsy data and methods. Unpublished Report prepared for the Ministry of Fisheries, Wellington. 8 p.

Roe, W (2010b) Necropsy of marine mammals captured in New Zealand fisheries in the 2008–09 fishing year. *New Zealand Aquatic Environment and Biodiversity Report No. 47*. 22 p.

Roe, W; Meynier, L (2012) Review of necropsy records for bycaught NZ sea lions (Phocarctos hookeri), 2000–2008. *New Zealand Aquatic Environment and Biodiversity Report No. 98*. 46 p.

Rowden, A; Clark, M (2010) Benthic biodiversity of seven seamounts on the southern end of the Kermadec volcanic arc, northeast New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 62*. 31 p.

Rowden A A; Clark M R; Lundquist C J; Guinotte J M; Anderson O F; Julian K A; Mackay K A; Tracey D M; Gerring P K (2015) Developing spatial management options for the protection of vulnerable marine ecosystems in the South Pacific Ocean region. *New Zealand Aquatic Environment and Biodiversity Report No. 155*. 76 p.

Rowden, A, et al. (2002) Benthic biodiversity of seamounts on the northwest Chatham Rise. *Marine Biodiversity and Biosecurity Report No. 2*: 1–21.

Rowden, A, et al. (2003) Benthic biodiversity of seamounts on the southern Kermadec volcanic arc. *Marine Biodiversity Biosecurity Report No. 3*. 23 p.

Rowden, A, et al. (2004) Benthic biodiversity of seamounts on the Northland Plateau. *Marine Biodiversity Biosecurity Report No. 5*. 21 p.

Rowden, A, et al. (2007) Benthic macrofauna bycatch of oyster dredging in Foveaux Strait. *New Zealand Fisheries Assessment Report 2007/10*. 27 p.

Rowden, A, et al. (2008) New Zealand's 'SEAMOUNT' database: recent updates and its potential use for ecological risk assessment. *New Zealand Aquatic Environment and Biodiversity Report No. 27*. 49 p.

Rowden, A, et al. (2012a) Biodiversity of macroinvertabrates and fish assemblages of the northwestern Ross Sea shelf. *New Zealand Aquatic Environment and Biodiversity Report No. 101*. 111 p.

Rowden, A, et al. (2012b) A review of the marine soft-sediment assemblages of New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 92*. 165 p.

Rowden, A, et al. (2013a) Biodiversity of macroinvertebrate and fish assemblages of the Balleny Islands seamounts. *New Zealand Aquatic Environment and Biodiversity Report No. 115*. 76 p.

Rowden, A, et al. (2013b) Predictive modelling of the distribution of vulnerable marine ecosystems in the South Pacific Ocean region. *New Zealand Aquatic Environment and Biodiversity Report No. 120*. 74 p.

Sagar, P; Thompson, D (2008) Data collection of demographic, distributional, and trophic information on selected petrels. *New Zealand Aquatic Environment and Biodiversity Report No. 17*. 12 p.

Sagar, P, et al. (2009a) Population assessment of Salvin's albatrosses at the Snares Western Chain, 29 September–17 October 2008. Unpublished Final Research Report for the Ministry of Fisheries. 4 p.

Sagar, P, et al. (2009b) Population dynamics of the Chatham Albatross at The Pyramid, 9 November–7 December 2008. Unpublished Final Research Report for the Ministry of Fisheries. 5 p.

Sagar, P, et al. (2010a) Population assessment of Northern Buller's Albatross and Northern Giant Petrels at the Forty-Fours, Chatham Islands, 1–8 December 2009. Unpublished Final Research Report for Ministry of Fisheries project PRO2006-05. 11 p.

Sagar, P, et al. (2010b) Population assessment of Salvin's albatrosses at the Snares Western Chain, 29 September–14 October 2009. Unpublished Final Research Report for the Ministry of Fisheries. 4 p.

Sagar, P, et al. (2010c) Population dynamics of the Chatham Albatross at The Pyramid, 20 November–14 December 2009. Unpublished Final Research Report for the Ministry of Fisheries. 6 p.

Savage, C (2009) Development of bioindicators for the assimilation of terrestrial nutrient inputs in coastal ecosystems as a tool for watershed management. *New Zealand Aquatic Environment and Biodiversity Report No. 30*. 35 p.

Savage, C, et al. (2012) Ecosystem Services Transcend Boundaries: Estuaries Provide Resource Subsidies and Influence Functional Diversity in Coastal Benthic Communities. *PLoS ONE 7 (8)*: e42708. doi:42710.41371/journal.pone.0042708.

Schiaparelli, S, et al. (2006) Diversity and distribution of mollusc assemblages on the Victoria Land coast and the Balleny Islands, Ross Sea, Antarctica. CCAMLR document WG-FSA-10/P3 18 (4): 615–631.

Schiaparelli, S, et al. (2008) Circumpolar distribution of the pycnogonid-ectoparasitic gastropod *Dickdellia labioflecta* (Dell, 1990) (Mollusca: Zerotulidae). CCAMLR document WG-FSA-10/P3 20: 497–498.

Schiaparelli, S, et al. (2010) 'Hitchhiker' polynoid polychaetes in cold deep waters and their potential influence on benthic soft bottom food webs. CCAMLR document WG-FSA-10/P3 22: 399–407.

Schwarz, A-M, et al. (2003) Macroalgal photosynthesis near the southern global limit for growth; Cape Evans, Ross Sea, Antarctica. *Polar Biology 26*: 789–799.

Schwarz, A-M, et al. (2005) Primary production potential of non-geniculate coralline algae at Cape Evans, Ross Sea, Antarctica. *Marine Ecology Progress Series 294*: 131–140.

Schwarz, A, et al. (2006) Impacts of terrestrial runoff on the biodiversity of rocky reefs. *New Zealand Aquatic Environment and Biodiversity Report No. 7*. 109 p.

Sewell, M (2005) Examination of the meroplankton community in the southwestern Ross Sea, Antarctica, using a collapsible plankton net. *Polar Biology 28*: 119–131.

Sewell, M (2006) The meroplankton community of the northern Ross Sea: a preliminary comparison with the McMurdo Sound region. CCAMLR document WG-FSA-10/P3 18: 595–602.

Sewell, M, et al. (2006) Whose larva is that? Molecular identification of larvae of the Ross Sea. *New Zealand Aquatic Environment and Biodiversity Report No. 3*. 57 p.

Sharp, B, et al. (2009) An Impact Assessment Framework for Bottom Fishing Methods in the CCAMLR Convention Area. *CCAMLR Science 16*: 195–210.

Sharp, B, et al. (2010) Bioregionalisation and Spatial Ecosystem Processes in the Ross Sea Region. CCAMLR document WG-EMM-10/30, Hobart, Australia.

Sharp, B, et al. (2011) A risk assessment framework for incidental seabird mortality associated with New Zealand fishing in the New Zealand EEZ. 39 p. (Unpublished report held by the Ministry for Primary Industries, Wellington.)

Slooten, E, et al. (2005) Distribution of Māui's dolphin, *Cephalorynchus hectori Māui*. *New Zealand Fisheries Assessment Report 2005/28*. 21 p.

Smith, A; Gordon, D (2011) Bryozoans of southern New Zealand: a field identification guide. *New Zealand Aquatic Environment and Biodiversity Report No. 75*. 65 p.

Smith, F (2006) Balleny Islands Ecology Research Voyage Report, R.V. Tiama. Unpublished Report for Ministry of Fisheries project ZBD2005/01.

Smith, I (2011) Estimating the magnitude of pre-European Maori marine harvest in two New Zealand study areas. *New Zealand Aquatic Environment and Biodiversity Report No. 82*. 72 p.

Smith, M; Baird, S (2005a) Factors that may influence the level of incidental mortality of New Zealand sea lions (*Phocarctos hookeri*) in the squid (*Nototodarus* spp.) trawl fishery in SQU 6T. *New Zealand Fisheries Assessment Report 2005/20*. 35 p.

Smith, M; Baird, S (2005b) Representativeness of past observer coverage, and future coverage required for estimation of New Zealand sea lion (*Phocarctos hookeri*) captures in the SQU 6T fishery. *New Zealand Fisheries Assessment Report 2005/05*. 39 p.

Smith, M; Baird, S (2007a) Estimation of incidental captures of New Zealand sea lions (*Phocarctos hookeri*) in New Zealand fisheries in 2003–04, with particular reference to the SQU 6T squid trawl fishery. *New Zealand Fisheries Assessment Report 2007/7*. 32 p.

Smith, M; Baird, S (2007b) Estimation of incidental captures of New Zealand sea lions (*Phocarctos hookeri*) in New Zealand fisheries in 2004–05, with particular reference to the SQU 6T squid trawl fishery. *New Zealand Aquatic Environment and Biodiversity Report No. 12*. 31 p.

Smith, M; Baird, S (2008a) Observer coverage required for the prediction of incidental capture of New Zealand fur seals in New Zealand commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 26*. 76 p.

Smith, M; Baird, S (2008b) Observer coverage required for the prediction of incidental capture of seabirds in New Zealand commercial fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 25*. 107 p.

Smith, M; Baird, S (2009) Model-based estimation of New Zealand fur seal (*Arctocephalus forsteri*) incidental captures and strike rates for trawl fishing in New Zealand waters for the years 1994–95 to 2005–06. *New Zealand Aquatic Environment and Biodiversity Report No. 40*. 90 p.

Smith, M; Baird, S (2011) Predicted incidental captures of New Zealand sea lions (*Phocarctos hookeri*) in the Auckland Islands SQU 6T squid trawl fishery for 1995 to 2006. *New Zealand Aquatic Environment and Biodiversity Report No. 71*. 40 p.

Smith, P; Notman, P (2005) Inventory of biological samples collected from Antarctic toothfish *Dissostichus mawsoni* over summer 2005. Unpublished Final Research Report for Ministry of Fisheries project ANT200401, Objective 5.

Smith, P, et al. (2007) Identification and speciation of Antarctic skates. Unpublished Final Research Report for Ministry of Fisheries project ANT2005/02, Objective 5. 27 p.

Smith, P, et al. (2008) DNA database for commercial marine fish. *New Zealand Aquatic Environment and Biodiversity Report No. 22*. 62 p.

Smith, P, et al. (2011a) DNA barcoding highlights a cryptic species of grenadier (genus Macrourus) in the Southern Ocean. *Journal of Fish Biology 78 (1)*: 355–365.

Smith, P, et al. (2011b) DNA barcoding of morids (Actinopterygii, Moridae) reveals deep divergence in the anti tropical *Halargyreus johnsoni* but little distinction between *Antimora rostrata* and *A. microlepis*. *Mitochondrial DNA 22*: 21–26. doi:10.3109/19401736.2010.532329.

Snelder, T H, et al. (2005) The New Zealand Marine Environmental Classification. Unpublished Report for the Ministry for the Environment. 80 p.

Snelder, T, et al. (2006) Development of an ecological marine classification in the New Zealand region. *Environmental Management 39*: 12–29.

Speed, S (2005) Review of the effects of aquaculture and enhancement stock sources on wild fisheries resources and the marine environment – report on Objective 1 & 2. Unpublished Final Research Report for the Ministry of Fisheries. 128 p.

Speed, S, et al. (2006) The implications of marine reserves for fisheries resources and management in the New Zealand context. Unpublished Final Research Report for the Ministry of Fisheries. 72 p.

Stein, D (2012) Snailfishes (Family Liparidae) of the Ross Sea, Antarctica, and closely adjacent waters. *Zootaxa 3285*: 120.

Stevens, D (2006) Stomach contents of sub-adult Antarctic toothfish (*Dissostichus mawsoni*) from the western Ross Sea, Antarctica. Unpublished Final Research Report for Ministry of Fisheries project ANT2004-01, Objective 4.

Stevens, D; Dunn, M (2010) Different food preferences in four sympatric deep-sea Macrourid fishes. *Marine Biology 158 (1)*: 59–72. doi:10.1007/s00227-010-1542-1.

Stevens, D, et al. (2010) Age, growth, and maturity of four New Zealand rattail species. *New Zealand Aquatic Environment and Biodiversity Report No. 59*. 39 p.

Stevens, D, et al. (2011) Feeding habits of New Zealand fishes: a literature review and summary of research trawl database records 1960 to 2000. *New Zealand Aquatic Environment and Biodiversity Report No. 85.* 218 p.

Strugnell, J, et al. (2012) Persistent genetic signatures of historic climatic events in an Antarctic octopus. *Molecular Ecology 21 (11)*: 2775–2787.

Sutherland, D (2008) Surface-associated diatoms from marine habitats at Cape Evans, Antarctica, including the first record of living *Eunotogramma marginopunctatum*. *Polar Biology 31 (7)*: 879–888. doi:10.1007/s00300-008-0426-z.

Sutton, C, et al. (2006) Biological parameters for icefish (*Chionobathyscus dewitti*) in the Ross Sea, Antarctica. Unpublished Final Research Report for Ministry of Fisheries project ANT2005/02, Objective 5. 27 p.

Taylor, P (1992) Incidental catch of non-fish species by setnets in New Zealand waters. *New Zealand Fisheries Assessment Research Document 92/21*. 23 p.

Thatje, S; Lörz, A (2005) First record of lithodid crabs from Antarctic waters off the Balleny Islands. *Polar Biology 28 (4)*: 334–337.

Thompson, F; Abraham, E (2009a) Dolphin bycatch in New Zealand trawl fisheries, 1995-96 to 2006–07. *New Zealand Aquatic Environment and Biodiversity Report No. 36*. 24 p.

Thompson, F; Abraham, E (2009b) Estimation of the capture of New Zealand sea lions (*Phocarctos hookeri*) in trawl fisheries from 1995–96 to 2006–07. *New Zealand Aquatic Environment and Biodiversity Report No. 41*. 31 p.

Thompson, F; Abraham, E (2009c) Six monthly summary of the capture of protected species in New Zealand commercial fisheries, summer 2007–08. *New Zealand Aquatic Environment and Biodiversity Report No. 35*. 22 p.

Thompson, F; Abraham, E (2010) Estimation of fur seal (*Arctocephalus forsteri*) bycatch in New Zealand trawl fisheries, 2002–03 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 61*. 37 p.

Thompson, F; Abraham, E (2011) Estimation of the capture of New Zealand sea lions (*Phocarctos hookeri*) in trawl fisheries, from 1995–96 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 66*. 25 p.

Thompson, F N; Berkenbusch, K; Abraham, E R (2016) Incidental capture of marine mammals in New Zealand trawl fisheries, 1995–96 to 2011–12. *New Zealand Aquatic Environment and Biodiversity Report No. 167*. 78 p.

Thompson, F N; Berkenbusch, K; Beritzhoff-Law, M (2015) Reported New Zealand sea lion (*Phocarctos hookeri*) captures in commercial trawl fisheries, 1991–92 to 2012–13. *New Zealand Aquatic Environment and Biodiversity Report 145*. 43 p.

Thompson, F, et al. (2010a) Common dolphin (*Delphinus delphis*) bycatch in New Zealand mackerel trawl fisheries, 1995–96 to 2008–09. *New Zealand Aquatic Environment and Biodiversity Report No. 63*. 20 p.

Thompson, F, et al. (2010b) Estimation of fur seal bycatch in New Zealand trawl fisheries, 2002–03 to 2007–08. *New Zealand Aquatic Environment and Biodiversity Report No. 56*. 29 p.

Thompson, F, et al. (2010c) Estimation of the capture of New Zealand sea lions (*Phocarctos hookeri*) in trawl fisheries, from 1995–96 to 2007–08. *New Zealand Aquatic Environment and Biodiversity Report No. 52*. 25 p.

Thompson, F, et al. (2012) Marine mammal bycatch in New Zealand trawl fisheries, 1995–96 to 2009–10. Unpublished Final Research Report for Ministry for Primary Industries project PRO2010-01, Wellington, New Zealand. 68 p.

Thompson, F, et al. (2013) Marine mammal bycatch in New Zealand trawl fisheries, 1995–96 to 2010–11. *New Zealand Aquatic Environment and Biodiversity Report No. 105*. 73 p.

Thrush, S; Cummings, V (2011) Massive icebergs, alteration in primary food resources and change in benthic communities at Cape Evans, Antarctica. *Marine Ecology 32 (3)*: 289–299. doi:10.1111/j.1439-0485.2011.00462.x.

Thrush, S, et al. (2006) Broad-scale factors influencing the biodiversity of coastal benthic communities of the Ross Sea. *Deep Sea Research II 53*: 959–971.

Thrush, S, et al. (2010) β-diversity and species accumulation in Antarctic coastal benthic communities: the role of habitat, distance and productivity on ecological connectivity. *PLoS ONE 5 (7)*: e11899. doi:10.1371/journal.pone.0011899.

Thrush, S, et al. (2011) Carbonate sediments: the positive and negative effects of land-coast interactions of functional diversity. Draft New Zealand Aquatic Environment and Biodiversity Report.

Tracey, D; Mills, S. (in prep) Identification of benthic invertebrate samples from research trawls and observer trips 2015–16. Unpublished Fisheries Report.

Tracey, D, et al. (2005) A guide to common deepsea invertebrates in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 1*. 160 p.

Tracey, D, et al. (2007) A guide to common deepsea invertebrates in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 10*. 282 p.

Tracey, D, et al. (2008) Classification guide for potentially vulnerable invertebrate taxa in the SPRFMO Area. Idenitifcation guide for Ministry of Fisheries. 1 p.

Tracey, D, et al. (2010) Evaluation of VME taxa monitoring by scientific observers. Unpublished Final Research Report for Ministry of Fisheries project ANT2009/01, Objective 8. 17 p.

Tracey, D, et al. (2011a) A guide to common deepsea invertebrates in New Zealand waters. *New Zealand Aquatic Environment and Biodiversity Report No. 86*. 317 p.

Tracey, D, et al. (2011b) Habitat-forming cold-water corals show affinity for seamounts in the New Zealand region. *Marine Ecology Progress Series 430*: 1–22.

Tracey, D, et al. (2014) Live Coral Experiment. Unpublished Final Research Report for Ministry of Fisheries project SEA2013-01. 9 p.

Tuck, I; Hewitt, J (2013) Monitoring change in benthic communities in Spirits bay. *New Zealand Aquatic Environment and Biodiversity Report No. 111*. 50 p.

Tuck, I D; Hewitt, J E; Handley, S J; Lundquist, C J (2017) Assessing the effects of fishing on soft sediment habitat, fauna and process. *New Zealand Aquatic Environment and Biodiversity Report No. 178*. 143 p.

Tuck, I, et al. (2006a) Information on benthic impacts in support of the coromandel Scallops fishery plan. Unpublished Ministry of Fisheries Final Research Report. 64 p.

Tuck, I, et al. (2006b) Information on benthic impacts in support of the Coromandel scallop fisheries plan. Info for SCA-CS fish plan. Unpublished Final Research Report for Ministry of Fisheries project ZBD2005-15.

Tuck, I, et al. (2009) Ecosystem indicators for New Zealand fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 42*. 188 p.

Tuck, I, et al. (2010) Designing a programme to monitor the recovery of the benthic community between North Cape and Cape Reinga. *New Zealand Aquatic Environment and Biodiversity Report No. 53*. 78 p.

Tuck, I, et al. (2014) Ecosystem and environmental indicators for deepwater fisheries. *New Zealand Aquatic Environment and Biodiversity Report No. 127*. 143 p.

Varian, S (2005) A summary of the values of the Balleny Islands, Antarctica. *Marine Biodiversity Biosecurity Report No. 6*. 13 p.

Waugh, S; Filippi, D P; Sharp, B R; Weimerskirch, H (2012) Ecological Risk Assessment for seabird interactions in surface longline fisheries managed under the Convention for the Conservation of Southern Bluefin Tuna. CCSBT-ERS/1203/09 (Rev.1).

Waugh, S, et al. (2008) Updated preliminary results of an ecological risk assessment for seabirds and marine mammals with risk of fisheries interactions Western and Central Pacific Fisheries Commission (WCPFC) Working Group – Fisheries Stock Assessment 08/51.

Waugh, S, et al. (2009) Ecological Risk Assessment for Seabirds in New Zealand fisheries. Unpublished Final Research Report for the Ministry of Fisheries. 58 p.

Williams, A, et al. (2010) Seamount megabenthic assemblages fail to recover from trawling impacts. *Marine Ecology 31 (supp1)*: 183–199.

Williams, G, et al. (2014) PENNATULACEA (SEA PENS) DESCRIPTIONS FOR THE NEW ZEALAND REGION. A field guide of commonly sampled New Zealand sea pens including illustrations highlighting technical terms and sea pen morphology. *New Zealand Aquatic Environment and Biodiversity Report No. 135*. 22 p.

Williams, J, et al. (2013) Review of factors affecting the abundance of toheroa (*Paphies ventricosa*). *New Zealand Aquatic Environment and Biodiversity Report No. 114*. 76 p.

Wing, S (2005) Fiordland Biodiversity Research Cruise. Unpublished Final Research Report for ZBD2003–04.

Zemke-White, W L, et al. (2005) Beach-cast seaweed: a review. *New Zealand Fisheries Assessment Report 2005/44*. 47 p.