Fisheries New Zealand

Tini a Tangaroa

Hauraki Gulf Marine Park habitat restoration potential

New Zealand Aquatic Environment and Biodiversity Report No. 265

M.A. Morrison

ISSN 1179-6480 (online)
ISBN 978-1-99-100963-0 (online)

August 2021

New Zealand Government

Requests for further copies should be directed to:

Publications Logistics Officer
Ministry for Primary Industries
PO Box 2526
WELLINGTON 6140

Email: brand@mpi.govt.nz
Telephone: 0800 00 83 33
Facsimile: 04-894 0300

This publication is also available on the Ministry for Primary Industries websites at:
http://www.mpi.govt.nz/news-and-resources/publications
http://fs.fish.govt.nz go to Document library/Research reports

TABLE OF CONTENTS

EXECUTIVE SUMMARY

Morrison, M.A. (2021). Hauraki Gulf Marine Park habitat restoration potential.

New Zealand Aquatic Environment and Biodiversity Report No. 265. **132 p.**

The Hauraki Gulf / Tīkapa Moana / Te Moananui-ā-Toi is one of New Zealand's most valued and intensively used coastal spaces. Declining health indicators of the Hauraki Gulf point to a range of issues including poor water quality due to land-originated sediments, the degradation and changes of habitats and ecosystems, the declines in ecologically important marine species such as mussels and their associated water-filtering function, and the reduction in some other kaimoana. The Sea Change process was initiated in 2013 to develop a marine spatial plan that responds to these negative impacts and seeks to restore the health of the Hauraki Gulf. The Sea Change – Tai Timu Tai Pari Hauraki Gulf Marine Spatial Plan was released in 2016. While the plan was being evaluated by Fisheries New Zealand (previously Ministry for Primary Industries, MPI), the Department of Conservation, and regional councils for real world implementation, MPI commissioned a small knowledge review to help inform possible restoration efforts for the Hauraki Gulf.

This review briefly covers some of the terminology and definitions used in the marine restoration literature, as well as related themes of humans being considered as part of the coastal landscape, positive and negative feedback mechanisms, scale issues, and landscape effects. Brief links to recommended restoration practices are given. For this report 'Recovery' is defined as "*The shift of an ecosystem back to a previous historical natural state, including its animal populations*". Restoration is defined as '*The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. A fully restored ecosystem is inferred to be self-sustaining and resilient*'. True full Recovery to a state analogous to before the arrival of humans is not possible with the continuation of human use of the gulf, and restoration goals need to be compatible with the human dominance of the coastal ecosystem.

The past environmental history of the Hauraki Gulf is described to give context to the fundamental changes that have occurred both on the land (through extensive land clearances, kauri and kahikatea forest logging, the clearance of wetlands, gold mining, and the urbanisation of Auckland) and the sea (through extensive fishing). Historical past records, observations, and personal accounts are used to provide an idea of what the Hauraki Gulf used to be like, and what has been lost or changed.

Present day marine habitat distributions and extents are then presented, with a focus on biogenic (living) habitats, including salt marsh, mangroves, seagrass, green-lipped and horse mussels, scallops, infaunal bivalves, sponges, mound-building tubeworms, macroalgae, and other species groups. Most show evidence of abundance declines and spatial distribution losses, except for mangroves. Knowledge gaps are highlighted, including a lack of knowledge of both past and present biogenic habitat distributions for the Hauraki Gulf, ecological connectivity around the idea of source and sink dynamics, and actual on-the-ground experimental restoration trials. Key points by species are:

- Salt marsh extents have greatly declined (more than 90%) due to land-use practices, and reclamation. Successful replanting of key species has been done in experimental trials. The use of salt marsh by whitebait in the Gulf would help valuing of these habitats.
- Mangrove forests have substantially increased in the Hauraki Gulf and their spatial expansions have been well described since the 1940s. They are not a highly valuable habitat for fish but are well used by a range of bird species. Restoration is not needed.
- Seagrass meadows have greatly declined from the past, but detailed historical maps are lacking. Subtidal seagrass is largely functionally extinct from the gulf. Intertidal seagrass has recently re-established in many areas (for unknown reasons), and small-scale transplant trials elsewhere in New Zealand have been successful. Restoration of subtidal seagrass meadows would provide substantial ecological functions around juvenile fish nurseries.

- Green-lipped mussel beds once covered around 500 km^2 of the inner Hauraki Gulf and Firth of Thames but were fished to total loss from the 1920s to the 1960s. Re-establishment trials using adult mussels from aquaculture operations have shown that beds can be formed on the seafloor for a while, but suffer high mortality rates over time, and do not attract settling mussel spat. How to turn these beds into permanent self-sustaining habitats remains to be solved.
- Scallop populations have declined from the past, both in spatial distributions and densities. Many beds still exist, including larger areas that have been commercially fished till recently. Restoration may be possible using broad stock reserves, as well as larger scale enhancement of areas of sandy seafloors where they are not often found, if increasing fisheries yields is a goal.
- Horse mussel beds are thought to be much less abundant than in the past, due to fishing gear impacts and land-based sedimentation, with losses continuing in the present day for unknown reasons. Restoration of these habitats would improve ecological functions, such as providing attachments for diverse epifaunal assemblages, and as fish nursery habitats.
- Infaunal bivalves are known to have experienced large declines, with cockles and pipi being the most obvious (and hence documented) coastal fringe harvested species. Although declines in fished beds are often well recorded (and sometimes recovery through beach closures), loss of these species more widely from the system is not. Restoration of these species has largely been passive and has focused on rebuilding greatly reduced fished beds using beach closures.
- Sponge gardens are composed of many species, which vary with depth and setting. Declines are thought to have occurred but are essentially undocumented. They are likely to have contributed to the ‘live ground’ described by retired fishers. Restoration is not logical until better knowledge of what may have been lost is available.
- *Galeolaria hystrix* (tubeworm) biogenic reef fields were only discovered in the gulf in 2019, and more knowledge about their past and present distributions is needed before the relevance of restoration can be assessed.
- Kelp forests and macroalgal meadows occur across much of the gulf in the present day, but historical distributions remain unknown. Restoration at present is focused on how to address sea urchin barrens, which are a trophic cascade effect from overfishing of predatory fish and crayfish.
- Locations that might be targeted for restoration are also discussed, where fishing impacts are known to have fundamentally changed the species assemblages and putatively high biodiversity values. These include North-West Reef and the southern side of the Colville Passage. Tamaki Strait is also included, as a system that may have been profoundly different in the past.

There is no doubt that the Hauraki Gulf is fundamentally different in the present day than it was historically. Large declines in the numbers and sizes of many species have occurred, which are best captured for fisheries species where empirical data have been collected (e.g., snapper, scallops, and lobsters). Habitats, especially biogenic habitats, are much less well documented, and there are no historical maps of most habitats (aside from some information on mangrove forests and green-lipped mussels). That information gap continues today and makes it problematic to set restoration targets and objectives, when it is not known exactly what used to be there. Green-lipped mussel and, to a lesser extent, scallop beds are the only real exceptions, based on knowledge from 50+ years of intensive fishing of these beds as a resource (mussels 1910s–1960s, scallops 1970s–present). Restoration work on those species is logical and justifiable, assuming that it is undertaken in areas where beds are well documented to have occurred historically. Information on other species is less certain. Information including present day seafloor habitat mapping, reconstructions from in-situ death assemblages, species distribution extraction from historical photography, and further detailed interviewing of relevant retired fishers (before their knowledge is lost forever) would be a significant step towards trying to reduce this uncertainty.

1. INTRODUCTION

The Hauraki Gulf / Tīkapa Moana / Te Moananui-ā-Toi is one of New Zealand's most valued and intensively used coastal spaces. In recognition of the gulf's long-standing cultural value as a taonga of national significance, the Hauraki Gulf Marine Park was established through the Hauraki Gulf Marine Park Act 2000 (HGMPA). The overall purpose of the HGMPA is to improve the environmental management of the gulf through the integration of the efforts of the various statutory authorities whilst recognising the historic, traditional, cultural, and spiritual relationship of tangata whenua with the Hauraki Gulf and its islands.

Declining health indicators of the Hauraki Gulf point to a range of issues including poor water quality due to land-originated sediments, the degradation and changes of habitats and ecosystems, the declines in ecologically important marine species such as mussels and their associated water-filtering function, and the reduction in some other kaimoana.

The Sea Change process was initiated in 2013 to develop a marine spatial plan that responds to these negative impacts and seeks to restore the health of the Hauraki Gulf. The Sea Change – Tai Timu Tai Pari Hauraki Gulf Marine Spatial Plan (the Plan), developed by an independent 14-member Stakeholder Working Group was released in 2016. It outlines a range of proposals to address the environmental decline while providing for social, cultural, and economic wellbeing.

There are four sections in the Plan: Kaitiakitanga, Mahinga Kai, Kai Uta Ki Tai, and Kotahitanga. Within the Fish Stocks chapter of Mahinga Kai, a theme relating to restoring marine habitats is described. One objective of this theme is to initiate a programme of action to achieve long-term habitat restoration including developing and testing innovative ways to restore degraded habitats.

Fisheries New Zealand (previously the Ministry for Primary Industries, MPI) and the Department of Conservation (DOC) have worked to develop a Government Response Strategy to the Sea Change Plan with input from the Ministry for the Environment. This involves testing and reviewing the proposals put forward in the Plan and determining how the Government should progress action to support the outcomes identified in the Plan. This response strategy was released while this current report was in final editorial, so is not discussed further here.

Assessment of the Plan's proposals for habitat restoration requires a common understanding of the concept of habitat restoration, an understanding of the different estuarine and marine habitats present in the Hauraki Gulf (past and present), the values and services these habitats provide, and the ecological requirements conducive to their successful establishment and ongoing health. An understanding of what restoration methods may be possible for these habitats and in which areas is also essential.

To facilitate this, MPI commissioned a review detailing this information along with other factors, to provide a guidance document to support active restoration projects in the Hauraki Gulf.

Objectives

Overall Objective:
To inform agencies' approach to estuarine and marine habitat restoration work in the Hauraki Gulf Marine Park.

Specific Objectives

1. Review estuarine and marine habitat restoration concepts and practice, both active and passive.
2. Describe as far as possible the estuarine and marine habitats and shellfish resources present within the Hauraki Gulf Marine Park during the early 20th century prior to the development of modern commercial fisheries and extensive urbanisation around Auckland.
3. Evaluate the services/values of these habitats provided.

4. Describe the ecological requirements of these habitat/species.
5. Identify which have been lost or substantially reduced in area, and any that may have increased in extent since 1950.
6. Identify the suite of habitats, including sites that would need to be restored within the Hauraki Gulf Marine Park to meet the outcomes of the Plan.
7. Prioritise possible active restoration efforts and describe the most practicable restoration approaches.

1.1 Scope and limitations of review

This small review is written to provide an overview of what the Hauraki Gulf once looked like ecologically, the human-induced changes that have occurred, what exists today, and what restoration might be best advanced. This review was commissioned by MPI to help inform the implementation of the Tai Timu Tai Pari Hauraki Gulf Marine Spatial Plan. Written material, from both the primary and grey literature, was sourced using a combination of key word web search engines and professional contacts in both science and regulatory agencies. The quality and type of material varied widely across different sources. Grey literature is included because this holds important information in the Hauraki Gulf context that does not exist in other forms. The international literature has been used to provide a short overview of restoration theory, as well as provide relevant research and recommendations from overseas. Because the topic is very large, even for any one species group (e.g., bivalves) or systems (e.g., rocky reefs, estuaries), the use of material has necessarily been selective and does not include all of the ecological and bio-physical work that has been undertaken in the Hauraki Gulf (and/or that might be applied from other regions). Its scope also does not include any new data analysis nor GIS plotting, with figures, maps, and other material taken from existing written sources. Biogenic (living) habitats are a focus, as an area which is tractable to address with restoration in more open systems. Some previous New Zealand focused reviews provide additional complementary material to that covered here for some species groups and issues, for example: seagrass physiology and ecology (Turner & Schwarz 2006), mangrove physiology and ecology (Morrisey et al. 2007, 2010), land-based effects on coastal fisheries and associated biodiversity (Morrison et al. 2009), biogenic habitats supporting New Zealand marine fisheries (Morrison et al. 2014a), coastal finfish species life-histories (Morrison et al. 2014b), and a stocktake of data on New Zealand's biogenic habitats (Anderson et al. 2019).

It was not possible to include in this review the (potentially very) large storehouse of knowledge held by the many hapū and iwi whose rohe falls within the Hauraki Gulf. This includes present day knowledge held by people alive today, oral knowledge passed down generations, and even the names of many places; all integrated into mātauranga Māori. It is acknowledged that such knowledge remains largely untapped as a framework to help guide restoration priorities; as well as the aspirations and priorities of iwi and hapū themselves around what restoration of the gulf should 'look like'.

2. RESTORATION CONCEPTS AND PRACTICE

Marine restoration science and its practice has been a rapidly growing area over the last several decades, as society and governments have come to appreciate the true degradation and losses of marine habitats and ecosystems.

2.1 Definitions

Elliot et al. (2007) argued that differing terminologies, and differing definitions, were causing confusion and slowing down the progress of marine restoration science and practitioner efforts. These authors reviewed the literature and recommended clearer terminology and definitions. Much of the following section is directly derived from Elliott et al. (2007), and the reader is referred to that paper for more detailed interpretation and discussion. Authors since that review have also further clarified definitions and terminology use (e.g., Geist & Hawkins 2016, Palmer et al. 2016).

Recovery

Definition: The shift of an ecosystem back to a previous historical natural state, including its animal populations.

Recovery is regarded as a return to an original state (Hawkins et al. 1999, Elliott et al. 2007). This differs from restoration, where the rehabilitated state is not expected to match the original state or be as healthy—but needs to be an improvement on the degraded state (Bradshaw 2002).

The terms recovery/recover and restoration/restore are often used interchangeably in the literature, including in the definitions of other terms, which creates confusion. For this report, a capitalised 'Recovery' is used to denote a true full return to a previous historical natural state; and a lower-case 'recovery' to denote where the term is used more often and broadly in the literature.

Restoration

Definition: The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. A fully restored ecosystem is inferred to be self-sustaining and resilient (Society for Ecological Restoration 2004).

This implies a shift of the system 'back' towards a previous natural condition, after being in poor ecological health due to degradation/disruptions, usually through human activities (direct and/or indirect). A full Recovery (as defined above in the Recovery section) would result in species compositions, population densities, and size/biomass structures matching the previous site/system natural conditions. As discussed later, restoration success is often suggested to be measured against current day, comparable 'unimpacted reference sites', in lieu of the historically lost site/system. This includes the recovery of the ecosystem goods and services, and carrying capacity, provided by the original natural system. However, as several authors including Simenstad et al. (2006) discuss, it is questionable whether a return to the original state can ever be fully achieved, even if its characteristics are fully known.

Note: Ecosystem goods and services are the economic benefits (goods and services) arising from the ecological functions of ecosystems. Such benefits accrue to all living organisms, including animals and plants, rather than to humans alone. An example of goods include fish taken by coastal fisheries; and for services, the water filtration roles provided by shellfish beds.

It is useful to keep in mind that in the restoration literature, spatial scale has no common 'measurement units' when authors report on their work. Most studies are conducted at small spatial (1s–10s–100s metre distances) and temporal (months to several years) scales, have a limited number of sites, and focus on one habitat type or species only. Work is usually focused on singular patches with no connectivity, rather than inter-connected patch mosaics. In discussing such work, the word 'system' is often used (e.g., for seagrass, oyster, and mangrove habitat studies). But habitats occur in ecosystems as many 'patches' at different spatial scales, connected to each other in their dynamics. When 'restoring a system' is referred to, this usually implies the full ecosystem (with its range of many different habitats), which can occur at the 10s–100s–1000s km^2 spatial scale, and 10s of decades temporal scale. The reader should keep these scale mismatches and ambiguities between empirical field studies, and their interpretation in terms of system restoration, clearly in mind.

Palmer et al. (2016) provide a useful table of some of the essential attributes of intact ecosystems that ecological projects aim to recover, and which ecological theories relate to a given attribute (Table 1).

Table 1: Three essential attributes of intact ecosystems that ecological restoration projects aim to recover. (Source: table 1.1. of Palmer et al. 2006). From Foundations of Restoration Ecology edited by Donald A. Falk, Margaret A. Palmer, Joy B. Zedler. Copyright © 2006 Island Press. Reproduced by permission of Island Press, Washington, D.C.

Attributes of restored ecosystems and ecological theory	Explanation	Relevant ecological concepts and theory
Biological assemblages are characteristic of a reference system of similar type.	Assemblage refers to the identity, relative abundance, and functional attributes of co-occurring taxa. References are relatively undisturbed systems where assemblages are within the historical range of variability	• Trait-based theories • Functional ecology • Diversity indices • Species range limits and controls on abundance • Community assembly and succession • Cross-scale spatial heterogeneity
Features and processes are needed to sustain the characteristic biological assemblages and support ecological functions are present.	Features include local habitat and system-level structure and spatial pattern in the watershed or landscape that are within the range of variability of reference sites. Processes include dynamic functions characteristic of the system that are necessary to the maintenance of the assemblages and features.	• Relationship of biodiversity to ecosystem functions (BEF) • Biogeochemical cycles • Hydrologic dynamics • Ecosystem engineers • Disturbance regimes • Dispersal, migration theory
Restored system has the potential to be self-sustaining	Self-sustaining systems require little or no human intervention or maintenance over the long term, in part due to suitable landscape and environmental contexts and exchanges of organisms, matter and energy.	• Range of variability • Metapopulation dynamics • Ecological resilience • Stability theory (alternative states, tipping points) • Nonlinear dynamics and feedbacks • Landscape and spatial ecology

Passive restoration

Definition: Passive restoration is the natural recovery of a system back towards its natural state, once human-driven stressor/s are removed/reduced, without further human intervention.

It is assumed that in ecosystems where the stressors have been removed, the system will naturally move back towards its original state. However, the level of recovery possible will depend on the properties of the system, which control its ability to absorb change, and to move back towards its natural state as stressors are removed. These properties include *recoverability*, *resilience*, and *adaptation*, as well as *carrying capacity*.

Restoration does not necessarily mean a return to the original, pristine state; as opposed to the term recovery, which is regarded as a return to an original state (Elliott et al. 2007).

Recoverability

Definition: The ability of a habitat, community or individual (or individual colony) of species to redress damage sustained because of an external factor (MarLIN Glossary 2005).

As an inherent property of ecosystems (and habitats), recoverability can vary widely for different ecosystem components. For instance, biogenic (living) habitats are regarded as much more vulnerable to fishing impacts (even from just one pass of a bulk harvesting method such as trawling or dredging) than 'simpler' seafloor habitat types such as sand-flats, with their recoverability being low and likely to occur over much longer time scales.

Ecosystem resilience

Definition: The inherent properties of the ecosystem which indicate its ability to absorb change, above and beyond an ecosystem's natural innate variability (Elliot et al. 2007).

Ecosystem (or ecological) resilience (or robustness) are terms that have been used in conflicting ways (Elliott et al. 2007). MarLIN Glossary (2005) defines resilience as simply "*the ability of an ecosystem to return to its original state after being disturbed*"; and Peterson (2000) defined ecological resilience to be "*the amount of change or disruption that will cause an ecosystem to switch from being maintained by one set of mutually reinforcing processes and structures to an alternative set of processes and structures*". Combining these and other definitions, Elliot et al. (2007) concluded that "*resistance and resilience, by their various definitions, are inherent properties of the ecosystem which indicates its ability to absorb change against a background of the complexity and/or variability of the ecosystem*".

This includes within system redundancy, e.g., more complex systems (species diverse) that are generally considered to be less suspectable to change with the loss of a few species. However, where keystone or habitat foundation species are impacted, a strong cascade effect can occur (Kaiser et al. 2005). An example of these for northern New Zealand is the trophic cascade that can occur on shallow rocky reefs. With overfishing of larger snapper (tamure, *Chrysophrys auratus*), and especially rock lobsters (koura, *Jasus edwardsii*), sea urchins (kina, *Evechinus chloroticus*) are freed from strong predation pressures that normally keep their population numbers in check, allowing them to systematically graze down kelp forests to bare rock (known as urchin/kina barrens), and then maintain that less productive alternative habitat state by preventing kelp spores/recruits from re-establishing (Shears & Babcock 2002).

Highly (naturally) variable ecosystems such as estuaries are more likely to be able to resist and/or absorb anthropogenic stresses than less variable ones (Elliott & Quintino 2007). Despite this, as resilience reduces with increasing numbers of smaller disturbances, a system's vulnerability may cumulatively increase until, eventually, even small disturbances that an undisturbed natural system could absorb can no longer be withstood, and the system collapses (the colloquial 'death by a thousand cuts'). Even in the absence of (direct) disturbance, slowly changing conditions (e.g., nutrient loading, climate change, habitat change, and invasive species) may ultimately exceed a system's threshold level, resulting in an abrupt system response (Kaiser et al. 2005). Due to this, Elliott et al. (2007) suggest that the terms resilience and recoverability are synonymous, and that only the former needs be used.

The paths of decline and recovery of systems are regarded as trajectories or performance curves (Simenstad et al. 2006, Elliott et al. 2007). Restoration effects (passive and active) effectively aim to broadly track back along these paths, towards the system's original natural state (Figure 1). Based on the Tett et al. (2007) conceptual model, Elliot et al. (2007) proposed a revised model to indicate that systems do not necessarily recover their former state, and that their ability to recover is termed resilience. Included in this are the concepts of Type I and II hysteresis (defined as the dependence of the state of a system on the history of its state), further explained in the Figure 1 caption.

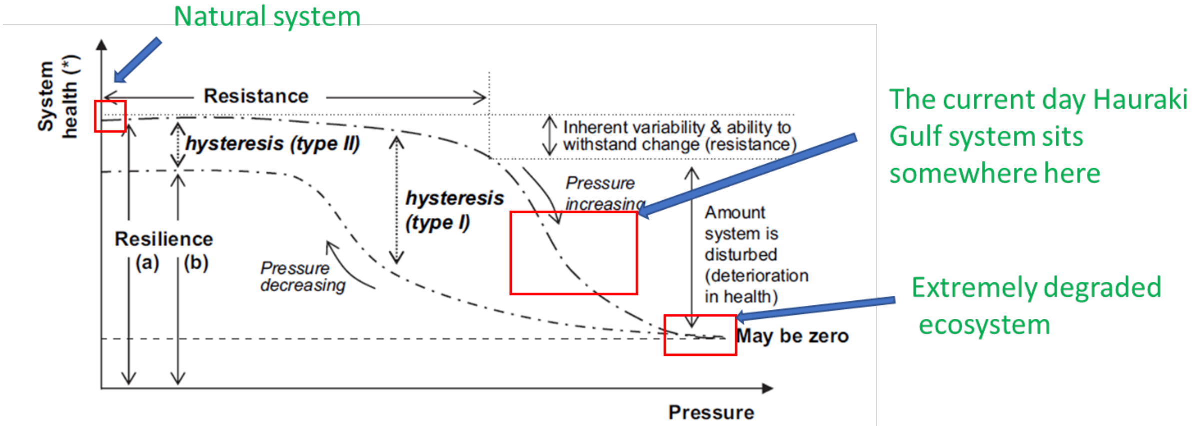


Figure 1: A conceptual model of changes to the state of a system with increasing pressure (modified from Elliott et al. 2007, as revised from Tett et al. 2007). As pressure on the system increases (from left to right), the system's health (*, as defined by a defined metric of structure or function) resists change, until the pressure exceeds the natural resistance of the system, and loss of system health occurs (top curve, left to right), often rapidly (i.e., non-linear decline). As the system is restored (pressure decreases, from right to left), systems that are fully resilient (a), show a Type 1 hysteresis recovery (the time lag in recovery); where system status may not improve for some time after the pressure is removed, but eventually restore to the original level. For partially resilient systems (b), full system recovery can never be achieved, with the system returning to a lower level that that of its original state. This Type II hysteresis is the difference from the Type I hysteresis. The current day Hauraki Gulf ecosystem probably sits to the right in this conceptual model (as indicated); and while only an objective opinion, is highly likely to be a Type II hysteresis system.

Adaption

Definition: The processes or coping strategies used by communities to increase their resilience (or decrease their vulnerability) to ecosystem changes (Elliott et al. 2007).

Species mixes may change during adaption, with some species replacing others but with the system still maintaining its functionality. Natural system changes due to climate variation fall within this term, where temperture regime shifts result in warmer water species migrating into some areas, whereas colder water species migrate out of the same areas. Similarly, some invasive species may functionally replace native species, although their effect is usually neutral at best, whereas a subset of these species 'damage' ecosytem functions (e.g., Mediterranean tubeworm *Sabella spallanzanii* in the Hauraki Gulf, Tait et al. 2020).

Carrying capacity

Definition:

1. Ecological. The maximal population (and/or community) that can be supported by the area's resources, principally space, food, and reproductive partners.
2. Human use. The maximum number of users (population and community) that can be supported by the ecological or economic goods and services provided by an area.

The most fundamental question for any restoration effort is '*what are we restoring to*?' (Simenstad et al. 2006). Usually the generic aim is to reinstate the loss of ecosystem goods and services that have been caused by environmental damage, which equates to a loss of carrying capacity (Elliott et al. 2007). Cohen (1997) defined the term 'carrying capacity' ecologically as "*the number of individuals in a population that the resource of a habitat can support*", "*the point at which the recruitment equals mortality*", "*the average size of a population that is neither increasing nor decreasing*", or, as related to limiting conditions, "*under steady state conditions, the population … is constrained by whatever resource is in the shortest supply*" (as quoted in Elliott et al. 2007).

For fisheries species, this term is less relevant for fished populations held at lower abundances than would occur in unfished systems. Stocks are usually fished down from their original biomass, known as B_0, which is acknowledged to not be a fixed number but rather an average around which a population fluctuates over time with environmental variation. In the 'fishing down' phase of new fisheries, usually the larger, older fish are preferentially removed, eventually resulting in an ongoing fishery (and supporting population) composed of younger, faster growing fish (assuming a fishery is sustainably managed). If fishing was the only stressor on the system and removing fish did not create any wider negative tropic cascades into the system, then the carrying capacity of the system would remain unchanged, and fish stocks should recover when fishing pressure (removals) are reduced (assuming that the fished population has not been reduced to such a low level that it can no longer replenish itself). This is the general theme under which fisheries management operates (managing the level of fish extractions). However, the issue of carrying capacity arises when other stressors/pressures act to reduce the system's carrying capacity from that which existed prior to fishing (time of B_0), such as increasing land-based sedimentation degrading the nursery, feeding, and spawning areas of fish (e.g., Morrison et al. 2009, 2014a). Fishing itself can also both directly (e.g., fishing gear impacts on seafloor habitats) and indirectly (e.g., trophic cascades) reduce the carrying capacity of the environment, both for the fished species and other organisms (Tuck et al. 2017).

Moving beyond just commercially extracted species, Elliott et al. (2007) defined carrying capacity as "*the maximal population (and/or community) that can be supported by the area's resources, principally space, food and reproductive partners*." As a Hauraki Gulf example of high carrying capacity habitats, the intertidal mudflats of the western Firth of Thames support seasonally high numbers of wading birds (Veitch & Habraken 1999); and subtidal seagrass meadows (now functionally extinct) historically probably supported high juvenile fish abundances (especially of 0+ snapper) (Morrison et al. 2014c).

Elliott et al. (2007) also recommended that (human) societal aspects should be included in defining carrying capacity, in terms of the availability of an area to support a given human activity. One example given was that a well-mixed, high-energy area might have a high carrying capacity to absorb organic wastes without adverse effects being detected; also described as the system's assimilative capacity, a term often used to indicate the ability of an area to accommodate (as in disperse, degrade, and assimilate) polluting discharges without damage (McLusky & Elliott 2004). MacLeod & Cooper (2005) also suggested such things as physical carrying capacity, i.e., the number of activities an area can withstand before there is some change to quality, such as the number of berths in a marina. Social carrying capacity refers to the human population densities an area can sustain before numbers start to decline due to negative feedback (e.g., actual/perceptions of amenity decline for coastal tourism). Economic carrying capacity refers to the extent to which an area can become changed before the economic goods and services are adversely affected; for example, excessive coastal development for tourism can ultimately damage those qualities that attracted tourism to begin with. Ongoing research and policy work on aquaculture operations addresses similar issues; e.g., how many aquaculture operations can be undertaken in an area before adverse effects occur on the farmed species themselves (e.g., depletion of pelagic food supplies for mussel and oyster farming, and/or the surrounding area) or the loss of aesthetic values becomes significant.

To account for this plurality, Elliott et al. (2007) proposed "*a composite definition that carrying capacity is the maximum number of users (population and community) that can be supported by the ecological or economic goods and services provided by an area. The aim of successful restoration therefore is to regain, maximise or enhance the carrying capacity*". This fit wells with current efforts to expand and add value to New Zealand's 'blue economy', included in the Hauraki Gulf Marine Spatial Plan.

Active restoration

Definition: Active restoration is the human-facilitated recovery of a system back towards its natural state, once human-driven stressor/s are removed/reduced (i.e. direct further human interventions continues).

Direct human interventions to actively help habitats and systems recover include activities such as transplant of seagrass sprigs to re-establish seagrass meadows (e.g., Matheson & Wadhwa 2012, Matheson et al. 2017), and the reseeding of bivalve beds using shellfish spat or adult transplants (e.g., Hewitt & Cummings 2013).

Other definitions workers have suggested include '*the process of re-establishing, following degradation by human activities, a sustainable habitat or ecosystem with a natural (healthy) structure and functioning*' (Elliott et al. 2007, based on Bradshaw 2002 and Livingston 2006), and, '*the process of assisting the recovery and management of ecological integrity. Ecological integrity includes a critical range of variability in biodiversity, ecological processes and structures, regional and historical context, and sustainable cultural practices*' (from Society for Ecological Restoration 1996, cited by Bradshaw 2002).

Further terms fall under the umbrella of active restoration, as below.

Rehabilitation

Definition: The act of partially or, more rarely, fully replacing structural or functional characteristics of an ecosystem that have been reduced or lost (Elliott et al. 2007).

This may not include the exact species mix that provided the original important structural or functional roles. Some species may be substituted by others. It is the structure/function that is the focus, rather than actual species identities *per se*. An example of this is the restoration of some juvenile fish nursery habitats, where structural complexity may be a key factor, regardless of what the identity of that structure is.

Habitat enhancement

Definition: A management approach which directly or indirectly increases the ecological value and goods and services of the habitat.

This is viewed from a qualitative, human-based perception, because the enhancement is improving a habitat that may already have good ecological functioning. A single species example is scallop enhancement, where human interventions in an area aim to greatly increase scallop abundance and biomass, to allow for greater fishery harvests. This usually includes removing natural spat-fall limitations that act as bottlenecks to juvenile recruitment to the seafloor, through deploying artificial spat-catching materials.

Habitat creation

Definition: An anthropogenic intervention which produces a habitat not previously there (Elliott et al. 2007).

This includes structures such as artificial reefs and deliberately sunken ships (as well as wrecks). This inevitably causes the loss of another habitat, and whether is it is an enhancement of the overall system is a value judgement. For instance, Erftemeijer & Lewis (1999) argued that the planting of mangroves in intertidal habitats where they did not historically occur, as a way to replace mangrove areal loss to degradation and shrimp farming, was habitat conversion (or rather habitat creation, as defined above), rather than habitat restoration. The loss of intertidal flats, as rich and productive systems, even if mangrove plantings were successful, would really be changing one valuable system into another. The potential gains were considered less than if the replanting efforts were to focus on the original degraded mangrove areas and shrimp farming areas.

To clearly distinguish between habitat creation and habitat restoration, Elliott et al. (2007) recommended that "*producing new habitat can be termed creation or enhancement whereas re-creating habitat that was present within historical records, no matter how old, should be termed restoration.*"

2.2 Concepts

Humans as part of the coastal landscape

Key message: Restoration goals need to be compatible with the human dominance of many coastal ecosystems.

Weinstein (2008) argued that any meaningful efforts to reverse (estuarine) degradation had to "*recognise that humans are an integral part of the landscape, particularly in urban estuarine settings, and that natural resource baselines have permanently shifted*". He stated that new approaches to coastal governance and management required both new transdisciplinary sciences, and the acknowledgement that "*sacrifices and compromises…will be necessary to achieve a balance between human uses of estuarine resources and biotic integrity*".

The concept that restoration practices are partly made up of cultural fabric' was noted, including the 'two-culture' conflict, being "*distinguishing between the need for scientific rigour and the broader outcomes of restoration based on societal perceptions and desires*" (see associated reference list given by Weinstein 2008).

The "*paradox of the dual mandate*" was also raised as needing to be resolved, "*where high levels of complexity, uncertainty and dynamicism, traits that prevent competitive dominance, sustain the biotic integrity of natural ecosystems; human-dominated ecosystems must be predictable and stable in order to buffer core technologies and human safety against the vicissitudes of nature"* (Roe & van Eeten 2001, 2002; Weinstein et al. 2007). The challenge is to "*reconcile society's desire to conserve, preserve, restore, and rehabilitate ecosystems and their concomitant natural variability, complexity, resilience and biodiversity – while ensuring the reliable and predictable provision of goods and services from the very same systems*" (Weinstein 2008).

Weinstein (2008) suggested that in trying to restore/rehabilitate coastal ecosystems at the landscape scale, this paradox had to be addressed, with the goals set being compatible with human dominance of many systems and the needs of other biota (besides humans). One potential solution was the spatial zoning (partitioning) of systems into different categories, some weighted towards human use (the anthropocentric component) and others towards the conservation/preservation of lesser disturbed systems (the ecocentric component) (Weinstein & Reed 2005, Weinstein 2005). This matches well with many of the tenets of Ecosystem Based Management, where zoning is a key tool, including aspects of the Hauraki Gulf Marine Spatial Plan. With respect to estuaries, Weinstein & Reed (2005) suggested that these be managed along a continuum based on human population density, anthropogenic disturbance levels (human footprint), the reliability of the goods and services provided, and/or by the degree of self-organisation and dynamic natures. Estuaries were categorised into three distinct types (Table 2, Figure 2).

Table 2: Summary of three different estuary categories, from Weinstein (2008).

URBAN INDUSTRIAL ESTUARIES

- Physiography and geological settings support intense human uses and populations, principally for living space, navigation, marine transportation, and commercial activity related to port commerce, energy production, other water intensive uses.
- As ecological baselines have shifted dramatically in urban industrial estuaries, concomitant losses in habitat/biodiversity are probably irreversible. Consequently, management priorities in urban industrial systems focus on reliability criteria, around predictable navigation depths, stable shorelines/berthing areas, cost-effective dredged material management methods, transportation infrastructure, storage facilities, and management of species adapted to human colonisation.
- Ecosystem health is not necessarily sacrificed; rather, contaminant source control, suitable sediment and water quality, and human endeavours to address them are equally important to commercial activity viability as to extant biota wellbeing.
- Concomitant restoration/biotic integrity goals in developing policy for management of urban industrial estuaries include conservation and preservation of remnant critical habitat (proximate biodiversity reservoirs) and rehabilitation of other habitats supporting species co-adapted to human's presence.

PRODUCTION ESTUARIES

- Have dominant eco-societal service vested in sustainable harvest or culture of estuarine-dependent species. Support, or recently supported, extensive commercial and recreational fisheries and/or aquaculture. Human population density moderate.
- Priority given to restoration/rehabilitation and fishery optimisation approaches focusing on habitat or other factors directly affecting economically/culturally important fish and shellfish production. Ecosystem management incorporates artificial reefs use, bycatch reduction methods, crop rotation techniques to allow bottom habitats/organisms to recover from fishing gear impacts. Other important societal uses; recreation, education, ecotourism, landscape aesthetics.

CONSERVATION ESTUARIES

- Largely undisturbed, self-sustaining systems with minimal human population density; virtually no human extractive uses. Nature and culture largely separated; natural processes and high levels of uncertainty dominate.
- Support non-consumptive uses on moderate scale, e.g., research, education, and ecotourism and/or provide aesthetic and cultural benefits to both local communities/wider society.
- Priority is conservation and restoration efforts focused on habitats that support finfish and shellfish and wildlife, including endangered/threatened species, migratory birds, resident species of the estuary.

Although focused on estuaries specifically, and with a focus on North American examples, this categorisation provides a useful lens through which to assess where restoration efforts are best focused. Weinstein (2008) was careful to emphasize that such a scheme was not advocating a total sacrifice of natural systems in the more urban-industrial category, but rather that "*restoration and/or rehabilitation practices should not be targeted to the historical condition (NRC 1992; Hildebrand, Watts & Randle 2005) but towards a baseline that is at once practical and realistic given the density of human presence and needs*". This is analogous to the concept of habitat 'triage', for essential coastal fisheries habitats, where management efforts are best focused on key habitats still remaining in the present (analogous to 'production estuaries'), and then restoration focused on the 'highest return' habitats that have been lost (Palmer et al. 2006, Morrison et al. 2014a, Geist & Hawkins 2016).

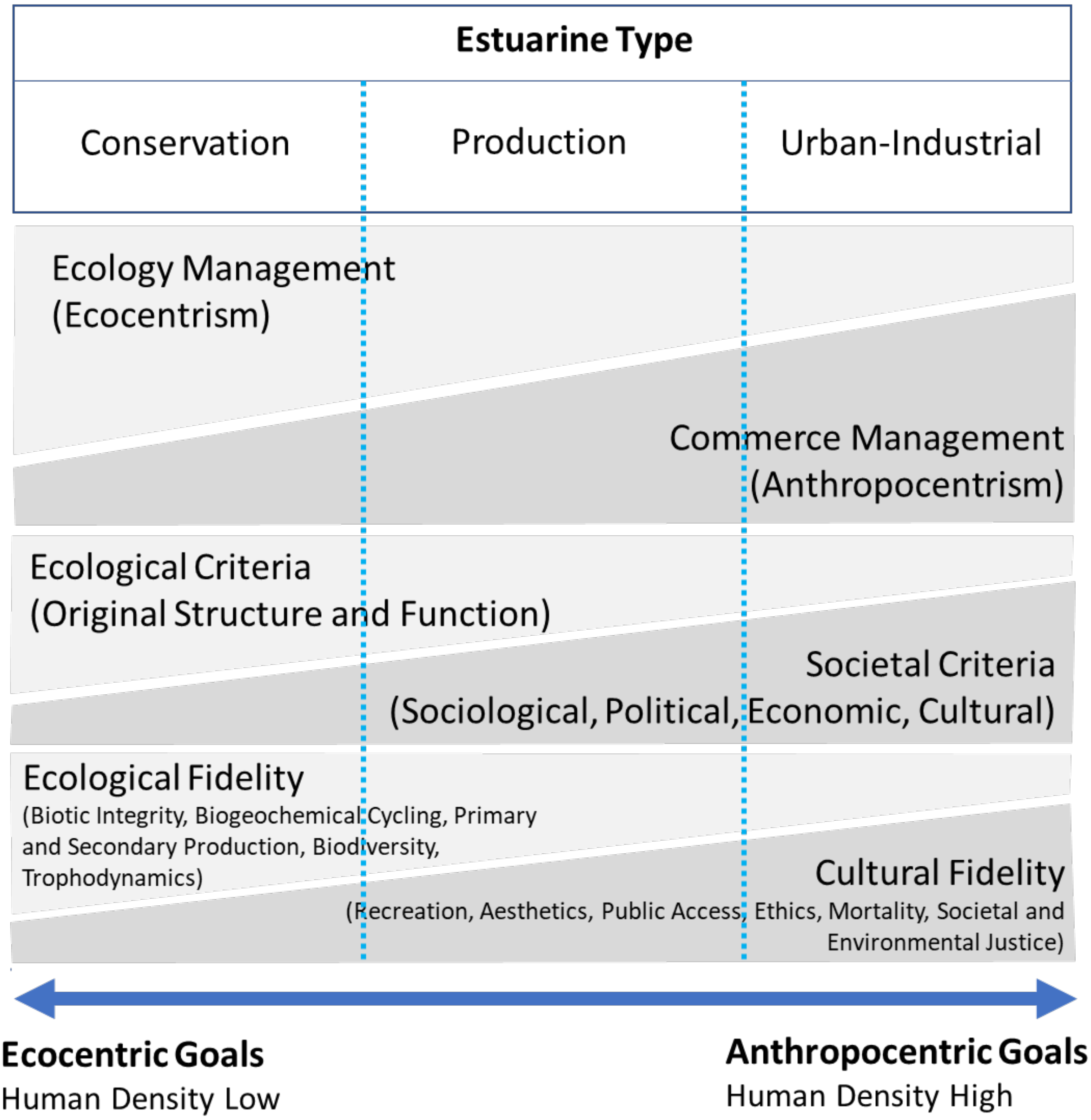


Figure 2: Conceptual management scheme for restoration/rehabilitation of degraded estuarine ecosystems based upon human landscape domination. Defining characteristics of the three estuarine types are given in Table 2. The proportion of each management criterion applied depends on the type of estuary in question and a series of drivers defined by ecological and/or societal criteria and the level of ecological and/or cultural fidelity desired (from figure 3 of Weinstein 2008).

Positive and negative feedbacks

Key message: Restoration needs to allow for ecosystem feedbacks that can help or hinder change, including time lags – for example, positive feedbacks such as species facilitation can be used to reduce time lags.

Marine ecosystems that have become degraded may be slow to revert to their original states (or not do so at all), because of the introduction of new ecological feedbacks that work to reinforce the continuation of the degraded state, despite the removal of the original stressors causing the degradation (Nyström et al. 2012). The new system state which may result from strong degradation, which can be called a 'regime shift/phase shift', can be so resistant to any change back towards its original state, that the system exists indefinitely as an alternative stable state (Petraitis et al. 2009). Even where the system can move back towards its original state, the time to recovery lag (known as hysteresis – see Figure 1) can be very slow, and completely different in its time trajectory from that of the degradation time-period. This locking of ecosystems in a degraded state "*is the bane of managers and policy makers*" (Nyström et al. 2012). Such dynamics exist across many coastal systems, including seagrass beds (van der Heide et al. 2007), kelp forests (Steneck et al. 2002), oyster reefs and mussel beds (Schulte et al. 2009, Petraitis et al. 2009), and coastal pelagic ecosystems (Beaugrand 2004, Mollmann et al. 2009). A species-specific example is the failed attempt to restore western North Atlantic cod dominance (and associated highly valuable fisheries), despite the introduction of large no-take regions (Hutchings & Reynolds 2004). Such issues can also result from 'legacy effect's'. For example, the inner Firth of Thames is now a very fine sediment (mud) dominated system with poor water clarity, whereas historical evidence points to a much clearer water system, with coarser sediments, dominated by bivalve beds (Greenway 1969, Reid 1969). This sub-system of the Hauraki Gulf is unlikely to recover, as millions of tonnes of fine sediment probably now overlay the past coarser sediment seafloor (though some better-quality biodiversity hotspots still appear to persist).

Nyström et al. (2012) split feedbacks into negative ('dampening' or 'stabilizing') and positive ('amplifying' or 'destabilizing') feedbacks. They noted that negative feedbacks tended to reduce fluctuations, create attractors, and stabilise key drivers (keep the system in a particular state), whereas positive feedbacks amplified changes in processes which destabilise the system. Pulse (sharp and short) perturbations (e.g., storms, hurricanes, outbreaks) or press (prolonged) perturbations (e.g., loss of habitat, fishing pressure, eutrophication, increased temperatures) can overwhelm negative feedbacks, allowing positive feedbacks to be dominant. Given strong enough positive feedbacks, a system can cross a threshold (sometimes referred to as a tipping point) and move to an alternative state, characterised by a new set of negative feedbacks, which might convey stability to the new degraded state (Scheffer et al. 2001). As with the restoration literature (as discussed by Elliott et al. 2007), the terminology used by workers to describe feedback systems is diverse and sometimes contradictory, with Nyström et al. (2012) suggesting that the most straightforward terminology was to call feedbacks that push marine ecosystems away from their original state as destabilisers, and those that reinforce contrasting states as stabilisers.

Nyström et al. (2012) highlighted that although feedbacks are now recognised as an important part of the dynamics and behaviour of marine ecosystems, they are poorly understood and, even less so, are ways to successfully break them. Those driven primarily by biotic interactions usually require approaches that control undesirable species and/or restoring processes that promote desirable species, with restoring trophic structures being key. A clear example of such a 'trophic cascade' in the Hauraki Gulf, and more broadly the northeastern New Zealand coast, is shallow rocky reefs and the dynamics of associated large brown kelp (mainly *Ecklonia radiata*), kina (sea urchins), snapper, and rock lobster populations (Shears & Babcock 2002). In brief, the fishing down of high density populations of large snapper and rock lobster on shallow rocky reefs freed sea urchins from the predation pressure that kept their numbers down, allowing them to reach densities (including forming urchin feeding fronts) and sizes that allowed them to graze out the large brown kelp forests previously dominant on the reefs. Kelp recruitment was then repressed by the ongoing grazing of new spores/sporelings by the urchins, maintaining the new 'alternative state' of 'urchin barrens', which were both ecologically (lower productivity, highly reduced species diversity) and anthropologically (low densities of fishable species, kina in poor condition for human harvest/consumption due to their high densities having a limited food supply) less valuable. Recovery of these systems was only apparent in marine reserves where large predatory snapper and rock lobster densities were able to recover (fishing as a stressor was (largely) removed), and, even then, there was a long time lag (hysteresis) before recovery. This was attributed to the large old urchins now present in these reef systems (not favoured by the predators) dying out naturally, while the predators suppressed new small urchin recruitment before the kelp forest could recruit back in and re-establish itself.

Breaking biotic-abiotic (physical) feedbacks was considered more complicated by Nyström et al. (2012), because such abiotic changes can 'profoundly' change site characteristics (e.g., eutrophication and sedimentation). Resetting a driving process/processes was suggested as being enough to break stabiliser feedbacks that kept systems in a degraded alternative stable state. An example given was that of the Ringkøbing Fjord (confusingly, a coastal lagoon rather than a fjord) in Denmark, which is about 30 km long and 2–3 m deep, covering around 300 km^2, and which opens to the North Sea via a canal with a floodgate. Historically stressed through high nutrient loading and associated oxygen depletion, its salinity was slightly changed through altered sluice management, which allowed for new recruitment of the suspension feeding bivalve *Mya arenaria*. The feeding by this species cleared the water column of pelagic algae, causing a rapid shift from a bottom-up controlled turbid state to a top-down controlled clear water state (Petersen et al. 2008) and a return of abundant fish and plant life. That example involved a native species that evolved as part of the local ecosystem. One of the key issues the Firth of Thames in the Hauraki Gulf faces is poor water quality caused by large human-driven increases in land-derived sediment and nutrient inputs. The large-scale restoration of green-lipped mussel beds has been suggested as a potential means to help move the area back towards its historical state (Hauraki Gulf Marine Spatial Plan).

Not all bivalve species have such positive outcomes. In the Great Lakes system of North America, zebra, and quagga mussels (native to Europe's Caspian Sea) first appeared in the 1980s, and since then have boomed in abundance, causing major damage to the ecosystems of the Great Lakes (Alexander 2019). Now widespread in all the lakes apart from Lake Superior (protected from mussel invasion because calcium levels are too low to support shell formation), the mussels have drastically increased the water clarity of the lakes through their water filtering, allowing sunlight (i.e., the photic zone) to penetrate into deeper water. Although this has been a huge boon in increasing the rate of discovery of the numerous historical shipwrecks lying on the lake bottoms, it has also supported fundamentally higher growth of benthic *Cladophora* algae, further bolstered by fertilisation from mussel waste. Around 950 trillion quagga mussels (270 000 tons) exist in Lake Michigan, and over 1531 km^2 of *Cladophora* bottom cover was discovered in 2012. Just offshore from scenic beaches in this lake, mats of decaying algae more than half a metre thick have also been seen. These algal drifts, when washed up on beaches, are a human health hazard because they harbour faecal bacteria and are also unsightly. *Cladophora* has also been linked to Type E botulism outbreaks, which killed more than 70 000 Great Lakes water birds from 2002 to 2012. The mussels themselves have damaged the $US7 billion Great Lakes freshwater fisheries through food chain disruptions and have disrupted power stations and water treatment facilities through heavy fouling. Other issues include the displacement (local extinction) of native freshwater mussels, smothering of rocky reefs used by fish for spawning resulting in reduced fish spawning success, and reduction of recreational and aesthetic human values because of dead shells littering beaches and the sharp shells of zebra mussels. No viable solutions have been suggested to reverse these large environmental degradation issues (Alexander 2019).

Scale issues and landscape effects

Key message: Where you choose to attempt restoration in an ecosystem really matters.

Landscape ecology is a relatively new field of science, with the primary focus to understand how an ecosystem's different components are organised spatially, and the implications of this structure on ecological processes, across different spatio-temporal scales (Turner 1989). At its simplest, 'place' (position in the seascape) matters (Metzger & Brancalion 2016) (seascape being the arrangement of habitats as a landscape)

Landscape setting and scale are key elements in achieving restoration objectives. The same restoration efforts, but performed in different locations of the landscape, can result in distinctly different ecological benefits and outcomes (Villard & Metzger 2014). Landscapes have two components: composition and configuration. Composition is how much of each type of a given habitat (called a landscape unit) is present, e.g., the overall spatial extent of biogenic habitats such as seagrass or horse mussel beds. Configuration is how those habitats (landscape units) are spatially distributed; e.g., how many patches of a given habitat exist, what spatial forms they take (e.g., patch shapes and sizes), and how they are spatially arranged relative to each other (e.g., as clusters or as solitary patches far distant from each other). An example for the Hauraki Gulf would be horse mussel beds ('habitat patches'); these occur around the gulf in a range of bed sizes and a variety of shapes and are more common in some gulf regions than others.

Landscape composition is a non-spatial attribute, whereas configuration is explicitly spatial, where the exact location of each habitat patch (landscape unit) needs to be known (i.e., mapped). Landscape configuration is inherently spatial, with landscape units (habitat patches) ranging from relatively small and compact forms (sometimes arranged in clusters to form larger-scale mosaics) to long, greatly elongated forms (often called habitat corridors, implying that organisms preferentially move along them). The main landscape unit (e.g., the one that is dominant in area or function) is called the ‘landscape matrix’ (Metzger & Brancalion 2016). An example of these would be a mosaic of seagrass patches (landscape units) growing on a general background of a sandflat (the landscape matrix).

Sometimes this main landscape matrix is defined as all habitats that are less critical from a functional viewpoint to a focal species reproductive fitness, or which provide some level of barrier to movement of a species. The identity of the focal species defines the spatial and temporal scales to be used, e.g., the large difference in spatial habitat use between a small, site-attached fish/bird species versus a larger, wide-ranging fish/bird species.

This is the simplest patch-corridor-matrix framework concept; ideally this should be replaced by a more continuous definition of habitat quality/suitability because each species responds to different environmental factors in a continuous way (McGarigal et al. 2009, Lausch et al. 2015). However, such true gradient approaches require a lot more data and a strong understanding of species life-history, neither of which are available for virtually all restoration projects (Metzger & Brancalion 2016). The use of such concepts allows researchers and resource managers to better understand how different species use habitat landscapes to maintain their populations, and how the ecosystem integrates through their dynamics.

Where landscape setting is ignored in restoration planning, the overall cumulative performance and outcome of restoration can be greatly hindered.

As noted by Simenstead et al. (2006), “*The cumulative response to restoration that is based on opportunistic, ad hoc selection of restoration sites and designs is likely to be additive at best; only strategic, spatially explicit restoration planning incorporating landscape scale processes is likely to create a cumulative response that is synergistic and complementary*”… “*If it is scientifically prudent to incorporate landscape context in designing and implementing restoration projects, it is imperative also to plan strategically and deploy restoration at the landscape scale.*

The demand for instant gratification often results in a “gardening” approach to restoration that circumvents life-history, natural variability and meso- or long-term cycles, disturbance, succession and other long-term facilitating processes that dynamically shape landscapes”.

In other words, where you choose to attempt restoration in an ecosystem really matters.

Despite this, although landscape ecology principles are often suggested to be important in restoration plans, at the global scale, only 12% of restoration sites have been placed strategically within landscapes to enhance likely outcomes for animals (note: this figure includes freshwater, terrestrial, and marine restoration) (Gilby et al. 2018).

2.3 Restoration practice

As a short generic introduction, the Society of Ecological Restoration (2004) listed nine ecosystem attributes to be achieved under successful restoration. Elliot et. al (2007) expanded and modified these to provide twelve objectives for marine restoration, which can be measured and relate to structural and functional attributes (Table 3). The authors note that many of these are site- and case-specific, that it is unlikely that the resources will be available to measure them all at a site, and that some will be “*nebulous* *to assess*” (e.g., maintenance of ecosystem functioning). Nevertheless, they are considered to match well with a range of management initiatives (a range of European examples are given by Elliot et al. 2007).

Table 3: Examples of estuarine, coastal, and marine indicators of successful restoration. S, structural; F, functional, and potential Hauraki Gulf examples (modified from table 5 of Elliot et al. 2007). The potential Hauraki Gulf examples in the right-hand column are given to provide some local context. The importance of scale is a key caveat here. The Hauraki Gulf will never return to a pre-human state (full Recovery); rather, the overall restoration objective is to move it partially back some way towards that state as practical, so that its ecosystem goods and services dynamics are as healthy as they can, while also allowing for human use.

Objectives	S/F	Potential Hauraki Gulf examples
(1) Spatial extent of biotopes, habitats, and/or ecosystems are as expected and self-sustaining under the natural physicochemical conditions and in comparison, to reference conditions.	S, F	Restoration of biogenic habitats such as green-lipped and horse mussel beds, seagrass meadows.
(2) Community diversity and structure, population abundance and reproduction, and species distribution are as expected under natural prevailing conditions and resilient to natural disturbances.	S	Fishing and land-based sedimentation impacts reduced.
(3) Threatened and/or protected species are in good status.	S, F	Cetaceans and sea bird populations (e.g., black petrels).
(4) Genetic diversity of farmed and wild-caught fish and shellfish species is not compromised, i.e., socio-economic importance.	S	Wild stock meta-population structure maintained (e.g., scallops).
(5) Extent of protected areas is maintained.	S	Current Marine Protected Areas (MPAs).
(6) The area under sustainable management for aquaculture or wild fisheries is as large as possible given societal demands and ecosystem constraints.	S	Firth of Thames shellfish/finfish farming zones.
(7) Nutrient dynamics are as expected under the prevailing hydrographic conditions (e.g., residence time) and not greatly modified by anthropogenic activities.	S, F	Improved nutrient status of the Firth of Thames.
(8) Invasive, alien, or introduced species are absent or in low numbers and have not affected the integrity of other species, the habitat, or ecosystem.	S, F	Invasive species abundance reduced (e.g., Mediterranean fan worm).
(9) Potential threats internal and external to the system have been eliminated, minimised, mitigated, or compensated and there is no detectable change in biodiversity through climate change (exogenic unmanaged pressures).	S	Better land catchment management, greater resilience of system to climate change through adaptive management frameworks.
(10) Community functioning and functional groups, e.g., as shown by marine trophic index and structure, are as expected and sustainable/stable in the long-term.	S, F	Kina/urchin barrens replaced by kelp forests for appropriate rocky reef systems.
(11) There is no physical or chemical disruption in connectivity of migration routes (i.e., no fragmentation) within and between ecosystems, e.g., water quality is protected so there is no permanent or temporary habitat loss.	S, F	Migratory diadromous fish species sea-freshwater pathways maintained and improved.
(12) There is the potential for the sustainable exploitation of species and or materials (e.g., marine bio-pharma-chemicals).	S, F	New blue economy industries developed. Could include exploitation of invasive/NIS species (e.g., greasy green-back prawns).

Many agencies and authors have produced ‘end-to-end how-to’ generic and/or habitat focused guides of how to undertake a successful restoration programme, especially for coastal wetlands and shellfish reefs. It is not proposed to reproduce those exhaustive documents here in detail. The reader is referred to the following documents that are listed in the References section:

International

- Brumbaugh & Hancock (2019). Shellfish reef restoration: An introduction. In: Restoration guidelines for shellfish Reefs
- Fitzsimons et al. (2020). Restoring shellfish reefs: Global guidelines for practitioners and scientists
- Layton et al. (2020). Kelp forest restoration in Australia
- Tan et al. (2020). Seagrass restoration is possible: insights and lessons from Australia and New Zealand

New Zealand

- Turner et al. (1998). Guidelines for community shellfish restoration and enhancement projects
- The National Institute of Water and Atmospheric Sciences (NIWA) (2010). Restoring shellfish beds to harbours and estuaries A guide for community groups

All of these guides consistently and fundamentally emphasise the importance of good project planning and management (Table 4) and having clear objectives from the start (Thom 2000). Table 4 (from Fitzsimons et al. 2020), although discussing shellfish reef restoration projects, is generically applicable to the restoration of any species group.

Table 4: Key guidelines for practitioners when undertaking shellfish reef restoration projects (from Fitzsimons et al. 2020). (Continued on next page)

Know the system you are working in
Become familiar with the ecosystem in its local setting (e.g., consider its historical distribution), causes for decline, current threats (including diseases), bivalve lifecycle and reproduction methods, and associated community assemblages. Gather evidence of recruitment strength and timing from previous research, observation, aquaculture operators, and settlement plates.

Develop a restoration concept and socialize with potential project stakeholders and supporters
Consider developing a short document that outlines project aspirations and potential approaches. Use this to receive feedback and support for establishing a more detailed feasibility plan and funding proposals. Include regulators in the outreach.

Establish a feasibility plan
Consider including the following in a feasibility plan:

- Identification of reference ecosystems or reference models and derived targets
- Clearly defined S.M.A.R.T. (Specific, Measurable, Attainable, Relevant, and Time-bound) objectives
- Identification of project stakeholders and supporters
- Likely funding streams
- Different restoration approaches
- Availability and disease tolerance of brood stock and source of seed (if larvae limited)

Identify funding sources and secure funding
Consider linking ecosystem service outcomes to beneficiaries and targeting funding opportunities linked to ecosystem service outcomes. Explore opportunities to leverage and match initial support.

Establish project management systems
Establish detailed project and implementation plans, communication plans, volunteer management, legal framework and contracts, detailed risk assessments, site management plans, tenders and quotes, and so on.

Know biosecurity risks and permitting requirements
Identify biosecurity and disease risks to wild populations and to aquaculture and fishing industries. Understand requirements and development times to secure permits. Understand/address the potential threat of the harvest of shellfish from the restored reef.

Undertake habitat suitability assessments and pilot studies
Identify optimal places for restoration with the system using suitability assessments, history of the most recent shellfish reefs, and pilot studies.

Confirm technical approach(es) required to support recovery including reef designs
Does the ecosystem require reconstruction (e.g., addition of substrate and shellfish), assisted regeneration (e.g., addition of substrate or shellfish), or management to limit threats (e.g., sediment, disease, or predation)? What reef designs will be used to support these technical approaches?

Undertake restoration
Work with community volunteers, contractors, and third parties to mobilize and deploy substrate, shellfish, and reduce/remove threats.

Undertake monitoring, evaluation, and reporting
Measure progress against predefined restoration targets and reference ecosystems and models. Measure universal indicators.

Effectively communicate outcomes of your project to stakeholders, practitioners, and the research community
Plan for communication, do the basics, and target visual media and social media.

3. HISTORICAL ECOLOGY OF THE GREATER HAURAKI GULF

Following is a short overview of major human activities and histories on the land and sea, after the arrival of humans to New Zealand/Aotearoa. It is not an exhaustive review of all activities and events, but rather provides some overview of the fundamental changes that humans have brought about in the gulf. Figure 3 shows the location of places mentioned in the text.

Figure 3: Map of places mentioned in this report.

3.1 Hauraki Gulf land catchments

Land clearances by humans and increased sedimentation

The Hauraki Gulf has a long history of human habitation and marine resource and exploitation (MacDiarmid et al. 2016). Within the first 200 years of Polynesian settlement around 1250 (± 30 years) (Wilmhurst et al. 2010), around 40–50% of New Zealand's native forest cover was removed by burning, especially along the east coasts of the North and South islands (Poole & Adams 1984, McGlone 2009). Most Maori land use was based around coastal plains or near lakes and rivers (Glade 2003), and hilly regions were only marginally influenced. With the arrival of European settlers, a further 35% of the original forest cover was removed over a 100-year period, starting around 1840 (Campbell 1946, Gomez et al. 2007, Morrison et al. 2009, Marden 2011).

Associated with these large-scale forest clearances were large increases in the rate of sediment run-off into adjacent estuary and coastal environments. Around the city of Auckland, marine sediment coring work in the Tamaki Estuary has found early to late Holocene (the last 10 000 years) Sediment Accumulation Rate (SAR) to be about 0.11–1.6 mm yr^{-1}, when the surrounding catchments were vegetated in podocarp hardwood forests. Following Māori settlement and associated forest clearance, SAR increased to 2.4 mm yr^{-1}, and, following European land clearances from about 1840 onwards, SAR further increased to 6.25 mm yr^{-1}, with significant increases of heavy metals (Cd, Cu, Pb, and Zn) in the most recent layers (Abrahim 2005). In the Papukura Estuary, pre-human SAR ranged from 0.2 to 0.5 mm yr^{-1}; these rates increased three-fold to 0.8–1.6 mm yr^{-1} following European forest clearance and subsequent agriculture in the mid-1800s, and at the top of the estuary rates have averaged 32.6 mm yr^{-1} since 1960 (Swales et al. 2002).

In the Mahurangi Harbour, following catchment deforestation (1850–1900), 3 m of sediment has accumulated at the head of the harbour, 70% of this since 1900 (Swales et al. 1997). Infrequent floods were found to drive much of the erosion, with one-third of the total catchment erosion generated from nine floods from 1953 to 1995. In Lucas Creek, in the upper Waitematā Harbour, rates increased from less than 1.5 mm yr^{-1} before human arrival to 2.5 mm yr^{-1} during Polynesian forest clearance (700–110 BP), and then to 3 mm yr^{-1} after Europeans arrived, with associated logging, gum digging, and land clearance (AD 1841 to the present (Hume & McGlone 1986)).

1800–1900s kauri logging

The kauri (*Agathis australis*) forests of Auckland and Coromandel (and more widely across northern New Zealand) supported a major industry in the early days of European colonisation (Figure 4).

Prior to European arrival, around 1.6 million hectares of kauri forest occurred in New Zealand; today around 7000 hectares remain, largely in Waipoua Forest, Northland (home of Tane Mahuta), and in state forest on the Coromandel Peninsula. Most kauri timber was extracted in the 1800s, being milled at an average rate of 236 000 cubic metres a year; dropping to an average of 3000 cubic metres in the late 1800s and early 1900s. Logs were extensively cut from kauri forests around the sheltered harbours of Northland, Auckland, and the Coromandel Peninsula. Digging for kauri gum was an industry for a while, and though more common in Northland, a gum field was worked in the Takapuna area.

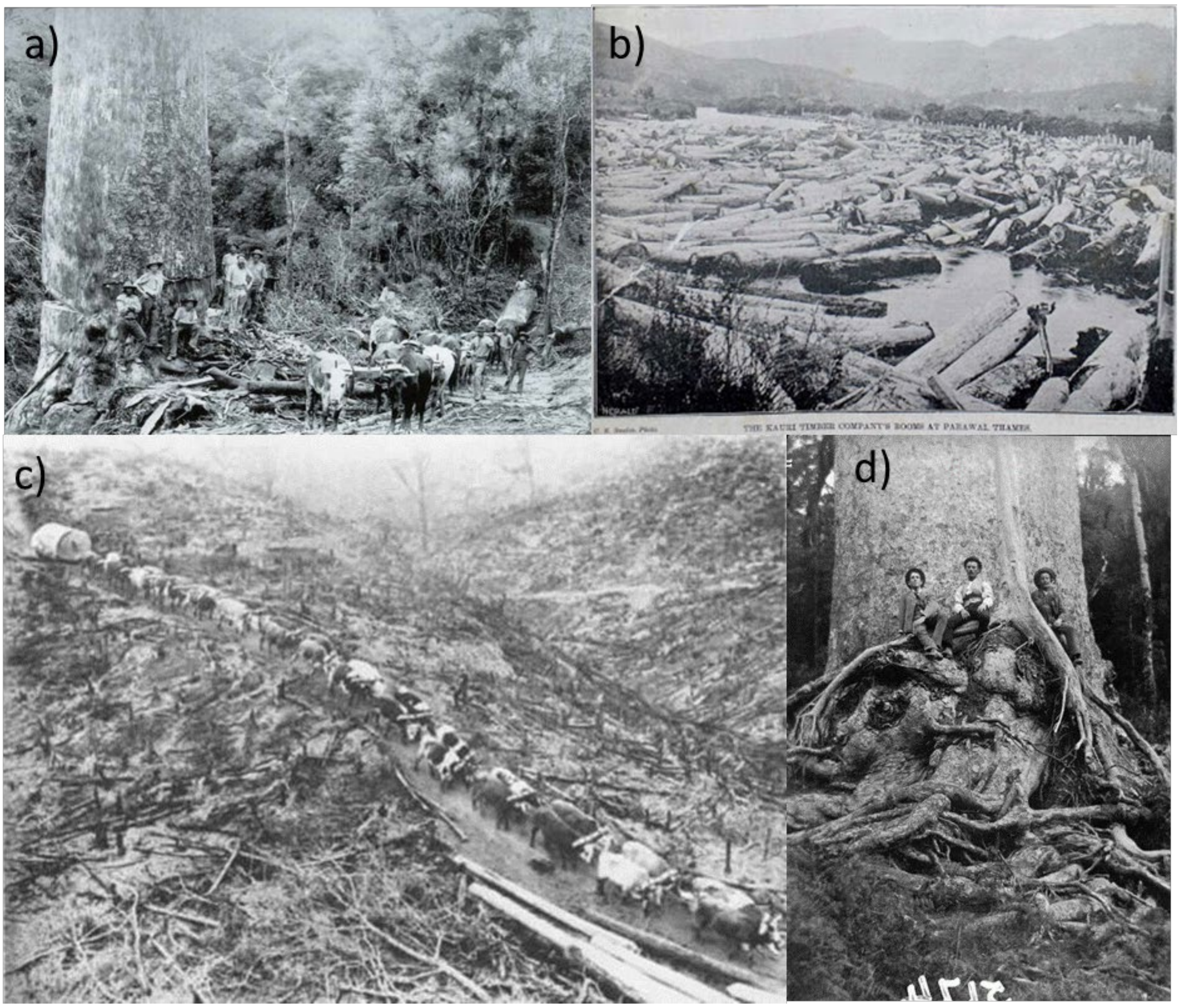


Figure 4: a) kauri logging operations in 1897 from an old postcard; b) kauri logs in booms across the Kauaeranga River, Parawai, Thames, 1890s; c) using oxen teams for log transport; d) three man sitting on top of root system of large mature kauri tree in the Nihotupu bush of the Waitakere Ranges. The tree's circumference is just under 16 metres. 1890s. Sources a) from an old postcard, *Agathis australis* (kauri) description (conifers.org); b) Auckland Libraries Heritage Collections AWNS-18980723-2-1; c) kauri log hauling – Coromandel – Thames – Kauaeranga Valley kauri logging history (wordpress.com); d) Auckland Libraries Heritage Collections, JTD-08F-05174.

1875 to 1920 clearing of the extensive kahikatea wetlands of the Hauraki Plains

The floodplains of the Hauraki Plains were once the largest wetland complex in New Zealand (Hatvany 2008). It was largely ignored by European settlers until the mid-1860s, due to the control asserted by independent Māori tribes and its flood-prone condition. With an almost flat elevation between the Matamata plains and the edge of the Firth of Thames, and surrounded by the Coromandel, Kaimai, Hunua, and Hapuakoke ranges, water flow to the sea was slow through the thick peat and sediment lowlands (Hatvany 2008). Two tidally influenced rivers, the Waihou and the Piako, meander through the plains. Prior to European settlement, these were surrounded by extensive wetlands of sedges, rushes, flax, and mānuka, with the two rivers full of sunken logs and frequently overflowing their banks. On drier ground, there were large dense stands of the celery pine kahikatea ('white pine', *Dacrycarpus dacrydioides*), with canopies reaching 55 m in height and, prior to logging, individual trees up to 80 m and more than 500 years in age existed in New Zealand.

Hatvany (2008) gives a detailed account of four generations of a European saw-milling family (the Bagnalls) on the Hauraki Plains. Arriving in New Zealand in 1863 from Canada, they moved from

Auckland to Turua in 1870 on a contract to build a wharf there, when the land was in the first stages of Pākehā settlement. Turua is located on the western bank of the Waihou River (Figure 4), five kilometres upriver from today's Kopu Bridge, and translates as 'twice seen', "*as one could simultaneously see the forest in the land and reflected in the river*" (Hatvany 2008). In 1875 the Bagnalls leased the economically struggling Hauraki Sawmill Company Ltd, which included a large area of forested wetlands acquired from the Māori owners, two large wharves, and miles of serviceable tramways radiating out into nearly 360 km^2 of partially forested wetlands (Figure 5). Intensive logging commenced over the next 30 years, with the wood being used mostly to make butter boxes, rabbit crates, and packing cases for the trade between New Zealand, Australia, and Britain (Elderton 1974). By 1890, the second Bagnall generation (brothers) turned to draining and selling the cleared wetlands for dairying. In 1908, the government passed the 1908 Hauraki Plains Act, to further foster drainage and settlement. By 1910, the Bagnalls were milling logs only intermittently, and the work of 'cleaning up' the swamp by transforming the remains into farmland through stumping, burning, and scattering grass seed was also reaching its end. In 1917 the last stands of kahikatea were exhausted and the Turua mill was closed.

Today, only the 7000-hectare Kopouatai peat dome remains (New Zealand's largest freshwater wetland), saved from total loss by being too low-lying to be drained Figure 5). Some small kahikatea stands remain in the Waikato lowlands, as relatively young forest fragments, 80–100 years old. These have grown around the few large old trees spared from logging, with some of these 'seed' trees still standing. Nationally, only 2% of the original kahikatea forest remains. The adjacent Firth of Thames now has large sedimentation and nutrient run-off issues, as well as a large expanding mangrove forest, and the large-scale water run-off filtration services that the Hauraki Plains once provided are gone.

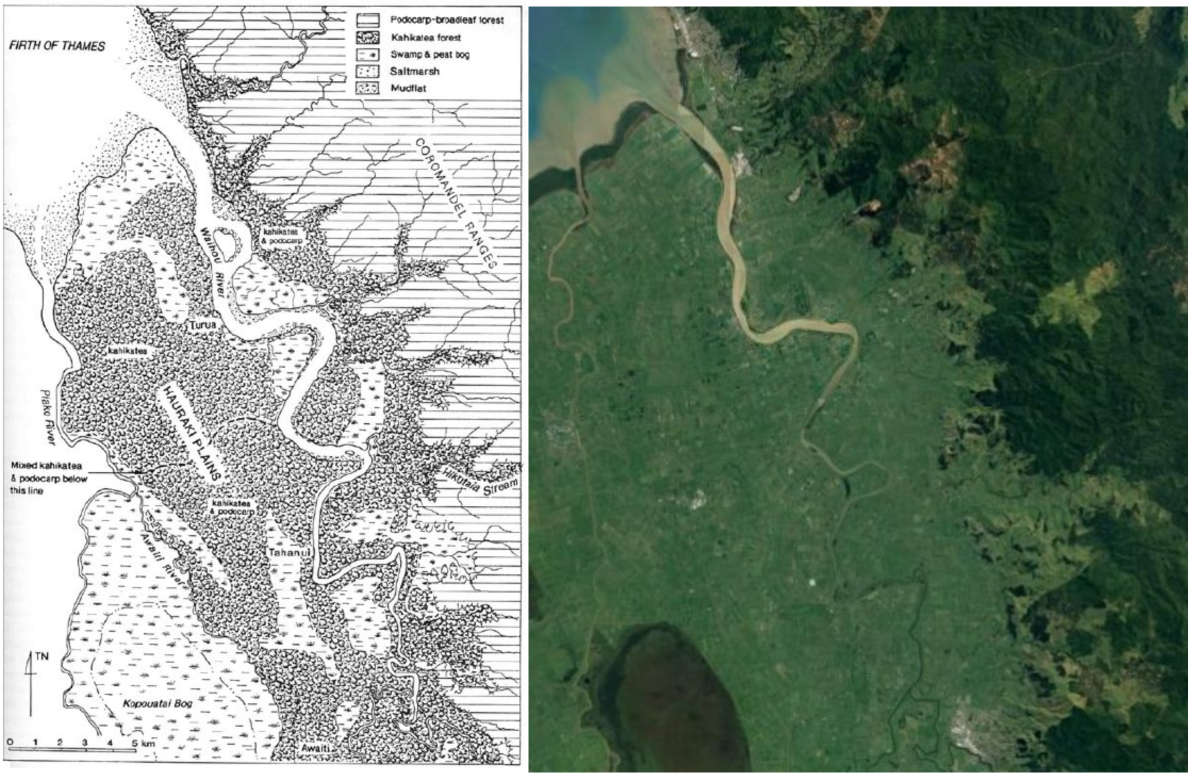


Figure 5: **Left) Extent of kahikatea forest and wetlands, east of the Piako River, northern Hauraki Plains, at beginning of European colonisation (figure 4 of Wilton 2019, originally sourced from Phillips, Waihou Journeys, 21); right) present day land cover in 2020. Note the patchwork of dairy farming fields. The Kopouatai peat dome is the oval brown object in the lower left of the image.**

1867–early 1900s Coromandel gold mining

Gold was first reported by Pākehā at Te Aroha in 1838 but, due to the 'Rotorua War' at that time and other matters, was not acted on. Gold was officially found in 1862. The first official gold workings started at Thames somewhere around June 1867. On 10th August 1867, a gold-bearing quartz reef was discovered, followed by others in later years, and the population of Thames expanded hugely and by mid-1868 had a peak population estimated at 18 000. Gold mining eventually dwindled out by the end of World War I. Ultimately, the Coromandel gold fields produced over £1½ million in gold, the Thames £7 million, and the Karangahake area another £4 or £5 million (Isdale 1964). Environmentally, it left a legacy of contaminated mine tailings, including deposits in Coromandel Harbour, and with much of present-day Thames built on mine tailings, some of which extend into the adjacent marine environment.

1840s – present: urbanisation of Auckland

The marine history of Auckland, spanning from first European settlement to today being New Zealand's largest city, is fascinating. Although it is well outside the domain of this review to cover it in detail, some interesting brief facets are given below for content. Initially focused on the southern side of the Waitematā Harbour, Auckland has always been an important trading port, and the development of its port and central business district has caused great changes to the shoreline and adjacent former seabed. With little flat land being present, reclamation has been key, with Figure 6 showing the extent of reclamation along the central Auckland waterfront and a reconstruction of its historical bathymetry (Wotherspoon & Lee 2016).

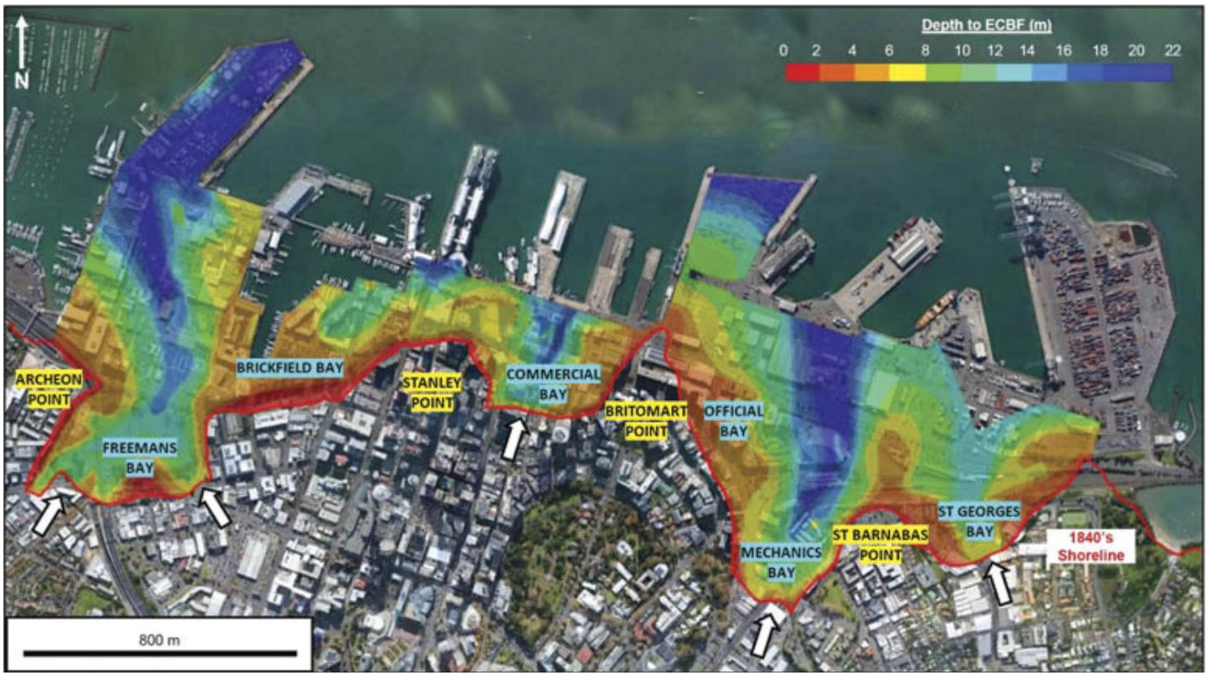


Figure 6: Reconstruction of natural bathymetry of Auckland Central Business District. Original 1840s shoreline (red), historic bays (light blue), headlands (yellow), paleo-river/stream channels (white arrows showing direction of flow). Source: Wotherspoon & Lee (2016).

To the west, the construction of the Westhaven Marina, and then the Auckland Harbour Bridge, has reclaimed further foreshore and seafloor (Figure 7).

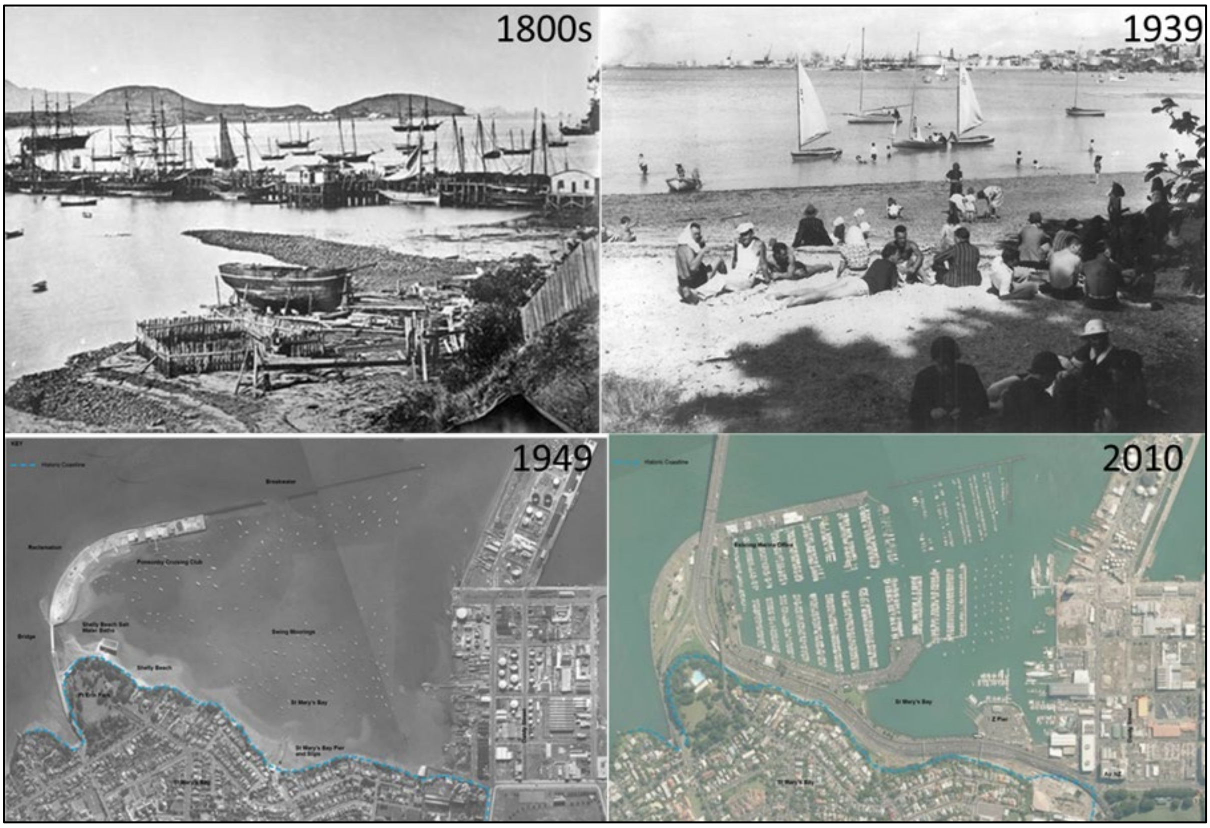


Figure 7: Changes in the Shelly Beach / Westhaven Marina / southern approach to the Auckland Harbour Bridge, 1800s to 2010. The blue dashed line in c and d shows the natural shoreline. Source: 2020 Panuku Development Auckland; https://www.westhaven.co.nz/westhaven/about/history/.

Where coastal features stood in the way of infrastructure development they were completely removed; this included Fort Britomart Point (see Figure 6 for its location) from 1972–1885, and the Bastion Point tor (Figure 8), where the Tamaki Yacht Club exists today on Tamaki Drive, sometime after 1920. An interesting narrative on the Fort Britomart Point removal is given by Truttman (2013).

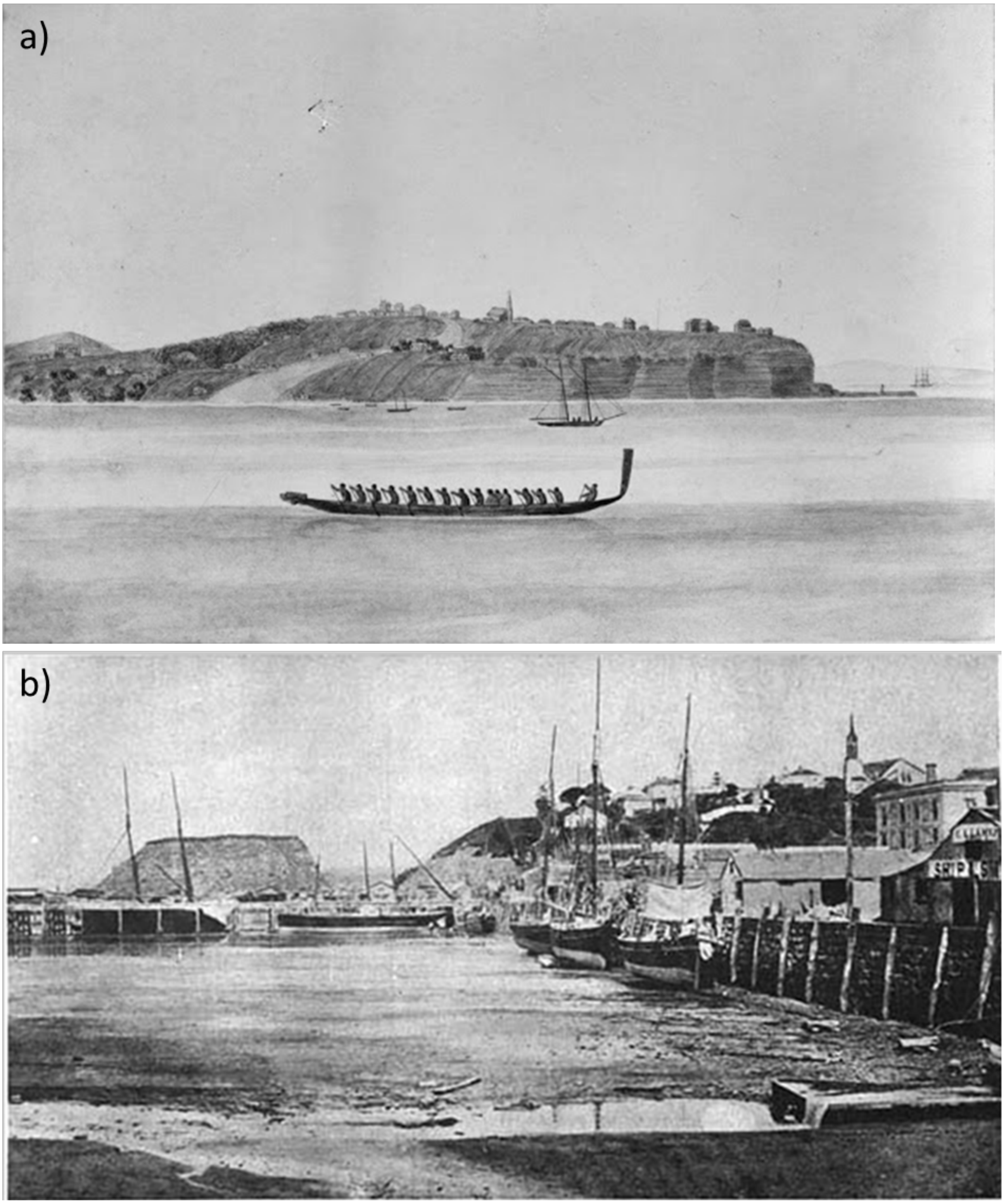


Figure 8: (continued on next page) a) Point Britomart, or ‘Soldiers' Point’, 1850s. Fort Britomart at the tip (right), St Paul's Church to be seen in the centre; b) Point Britomart partway through its removal, with earthworks, circa 1876–1878; c) Bastion Rock, before its removal around 1920; d) close-up of Bastion Rock, also showing the Auckland shoreline before the construction of the Tamaki Drive roadway. Source: a) Sir George Grey Special Collections, Auckland Library, Ref 4-576A; b) Sir George Grey Special Collections, Auckland Library Ref 4-7130; c) unknown; d) Sir George Grey Special Collections, Auckland Libraries, 4-3375.


c)
3


d)

3.2 The sea

Marine species exploitation and loss

MacDiarmid et al. (2016) provide a detailed review of human harvesting of marine biological resources from the greater Hauraki Gulf from the time of first human arrival, through to the present day. Most of the following historical fish, marine mammals and bird populations section is summarised from MacDiarmid et al. (2016). Archaeological reconstructions of pre-European food harvesting by Smith (2011, 2013) found Māori harvested a wide range of marine species from first settlement, with 101 taxa (46 shellfish, 28 fish, 22 shorebirds or seabirds and 5 marine mammals) identified from the Hauraki Gulf region. Around 1400 AD, about 30% of overall dietary energy of Māori came from terrestrial birds and mammals (the Polynesian rat *Rattus exulans* and dogs (kurī), falling to less than 10% by 1750 AD. Over the same period, reliance on marine dietary energy sources rose from about 70% to 90%. The total harvest of resources over this period increased, but with targeting of fewer marine taxa (down from 22 to 15 for fish, 88 to 38 for shellfish, 5 to 1 for marine mammals, and 22 to 10 for seabirds) (MacDiarmid et al. 2015). Annual harvests of sharks and rays increased from about 5 ± 1.45 t to 1318 ± 396 t, snapper from 72 ± 22 t to 997 ± 299 t, and cockles (*Austrovenus stutchburyi*) increasing from <1 t to 1358 ± 407 t (Smith 2011). Smith (2011) attributed these large increases to both an increasing human population and the loss of other food sources (seals, moa, and some marine and coastal birds).

With the arrival of Europeans, substantial local domestic market fisheries developed for rock oysters (*Saccostrea commercialis*), dredge mussels (*Perna canaliculus*), rock lobster (*Jasus edwardsii*), grey mullet (*Mugil cephalus*), flatfish, groper (*Polyprion oxygeneios*), and particularly snapper (Parsons et al. 2009, Paul 2012, 2014). As harvesting pressure increased over time, early concerns were raised about the sustainability of some fisheries species (grey mullet, rock oysters). Other species declined more slowly. In 1930, following 70 years of fishing and the introduction of Danish seining around a decade earlier, the Chief Inspector of Fisheries at the time (Mr Hefford) warned that "*This fishery is prosecuted at the expense of spawning aggregations of flounders and dabs, and therefore requires careful watching both from the economic and the biological aspect*" (MacDiarmid et al. 2016). Issues with declining availability of groper/hapukū were also raised around this time, with Mr Hefford noting in 1928 that '*It is the amount of fish that are extracted, not the method of fishing, which is the important factor*'. With the introduction of steam trawlers and purse seining in the early 20th century, localised snapper stock depletion became apparent in the inner Hauraki Gulf, and trawling was banned from these areas to protect the local longline fishery and the health of Auckland's recreational fishery (Parsons et al. 2009, Paul 2012).

Fisheries catches were first statistically documented in 1930; Paul (2012) characterised these fisheries in detail from this time through to the introduction of the Quota Management System in 1989. The nature of commercial fishing operations in the greater Hauraki Gulf study region since the start of the fisheries statistical period in 1930 is described in detail by Paul (2014). A complete history of the exploitation of snapper, compiled from archaeological, historical, and contemporary sources, is given in detail by MacDiarmid et al. (2012).

Modelling by McKenzie & MacDiarmid (2012) estimated that the greater Hauraki Gulf's combined biomass of all commercially fished finfish and lobster species declined by about 60% after 1930 (when fisheries catches were first statistically recorded). All 23 fished species declined in biomass, with snapper contributing half of the overall decline, and seven species including rock lobster, groper, and school shark (*Galeorhinus australis*) decreased by over 70%. Expressed as tons per square kilometre, standing combined biomass of these fished species dropped from around 21 t per km^2 in 1930 to 8.5 t per km^2 (note, these are gulf wide averages only; habitat, depth, and area density gradients occur at finer spatial scales).

Sea mammal and seabird populations

New Zealand fur seals (*Arctocephalus forsteri*) and sea lions (*Phocarctos hookeri*) were lost from the gulf ecosystem overall a fairly short period (MacDiarmid et al. 2016). Smith (2011) estimated that from 1250–1450, about 60% of the human population's marine diet came from seals, with around 4000 fur seals and 560 sealions harvested each year from the greater Hauraki Gulf. This exploitation level eliminated the gulf populations; at the national scale, seal populations disappeared southwards from North Cape to the bottom of the South Island over a 500-year period (Smith 2005). Late 18th and early 19th century European commercial sealing completed the extinction of the mainland seal populations (Lalas & Bradshaw 2001, Ling 2002).

New Zealand fur seal populations are now recovering at the national scale, concentrated in the Otago-Catlins region. This recovery is thought to be slowly moving northwards as breeding colonies become re-established, and small numbers of fur seals are now regularly being observed in the greater Hauraki Gulf (Clemens et al. 2011, Department of Conservation 2011, 2012). Female New Zealand sea lions have strong tendencies to return to their beach of birth to reproduce (Chilvers & Wilkinson 2008, Gales 2017), making new breeding colonies slow to establish. The main surviving populations persist at the sub-Antarctic Auckland and Campbell islands, the southern edge of its historical distribution. New Zealand sea lions are listed as 'Nationally Vulnerable' under the New Zealand Threat Classification System (Baker et al. 2019). Recent genetic work on sea lion DNA recovered from middens has shown that those once inhabiting mainland New Zealand were distinct (possibly at a species level) from the surviving sub-Antarctic populations of today (Collins et al. 2014). Re-population of northern New Zealand may never occur.

Larger whales were probably not actively hunted by pre-European Māori (Smith 2011), though stranded whales were utilised for bone and teeth for carving. Small dolphins were estimated to be taken at a rate of 21–47 individuals a year (species generally unknown) in the greater Hauraki Gulf (MacDiarmid et al. 2016). At the time of Caption Cook's first arrival in New Zealand (1769 AD), an estimated southern right whale (*Eubalaena australis*) population size of 27 500–31 500 individuals (95% CI; 22 000 to 38 000) occurred in New Zealand waters. With the commencement of commercial whaling by Europeans, around 35 860 to 40 650 individuals were harvested, with around 80% of these between 1830 and 1849 (Carroll et al. 2014). Although few were taken from the Hauraki Gulf region, intensive whaling in southern New Zealand took the overall population close to extinction in the early 20th century, with a low point of perhaps less than 100 individuals (Jackson et al. 2016).

Sperm (*Physeter macrocephalus*) and humpback (*Megaptera novaeangliae*) whales were also taken from New Zealand waters (Jackson et al. 2016, Smith et al. 2012), and pelagic whaling in the Southern Ocean/Antarctic removed large numbers of humpback, blue (*Balaenoptera musculus*), and fin (*Balaenoptera physalus*) whales (Baker & Clapman 2004, Clapman & Ivashchenko 2009, Clapman et al. 2009), some of which would have migrated through New Zealand waters during their seasonal migrations between southern waters and the tropics (MacDiarmid et al. 2015).

Pinkerton et al. (2015) estimated that collectively, historical whaling harvests have acted to reduce the present-day greater Hauraki Gulf whale biomass by about 70%. Combined with the loss of fur seals and sea lions, and some fish stocks, there may have been negative trophic effects on apex predators such as great white sharks (*Carcharodon carcharias*) and killer whales (*Orcinus orca*) (see references given by MacDiarmid et al. 2016), including the availability of marine mammal pups, calves, and placentas.

In the present day the greater Hauraki Gulf is viewed as a globally significant biodiversity hotspot for seabirds (State of the Gulf 2020). More than 70 species are found in the region (20% of the world's seabird species), with 27 species breeding there (with 59% of these endemic to New Zealand). Four species breed only in the greater Hauraki Gulf (Buller's shearwater *Ardenna bulleri*, Pycroft's petrel, *Pterodroma pycrofti*, black petrel *Procellaria parkinsoni*, and New Zealand storm petrel *Fregetta maoriana*) (Gaskin & Rayner 2013). Evidence is increasing that seabirds were once important in transferring nitrogen from the ocean to terrestrial and freshwater ecosystems at mainland breeding colonies (Hawke & Holdaway 2003, 2005, 2009; Harrow et al. 2006, Holdaway et al. 2007). Before human arrival, mainland New Zealand's hills and mountain ranges were densely populated with seabird breeding colonies (Worthy & Holdaway 2002); for the Hauraki Gulf, these would have been dominated by small burrowing seabirds (particularly petrels). The arrival of the Polynesian rat *Rattus exulans* (kiore) is likely to have swiftly depleted these by predation, along with direct exploitation by early Māori (Wilmshurst et al. 2010, MacDiarmid et al. 2016). Smith (2011) estimated that around 1400 AD, Māori were harvesting around 19 t of seabirds a year (about 18 000 individuals). Harvest was dominated (70.1% of harvested biomass) by little blue penguins (*Eudyptula minor*), spotted shags (*Stictocarbo punctatus*), pied shags (*Phalacrocorax varius*), and albatrosses. By around 1550 AD, the combined harvest had increased to about 28 t (16 000 individuals, with three of the above four species still dominating (90.2% of biomass); the exception being pied shags). By 1750, the annual harvest was 56 t (62 500 individuals) of the same three key species (now 71.6% of biomass). Marine and coastal bird harvest by Māori expanded at less than half the rate of human population growth, decreasing their overall contribution to human diets; this was considered to reflect reduced bird availability from both human and introduced mammalian predators (rats/dogs) (Holdaway 1999, Smith 2013, MacDiarmid et al. 2016).

With the arrival of European settlers, many new mammalian predators were introduced (Towns et al. 1997, Holdaway 1999, Nogales et al. 2004). Combined with harvesting and fisheries bycatch issues, although few seabird species became extinct (Tennyson & Martinson 2006), many species population numbers have declined (MacDiarmid et al. 2016). The magnitudes of the declines in seabird numbers from the combined effects of predation, harvesting, and fisheries bycatch are not well known (Pinkerton et al. 2015). In the present day, the black petrel is most at risk, through fishing bycatch mortalities in and adjacent to the greater Hauraki Gulf (Richard et al. 2020). Its only breeding locations in the present day are Great and Little Barrier islands in the Hauraki Gulf. This endemic species is classified as 'Nationally Vulnerable' by DOC (Robertson et al. 2017).

3.3 Marine species and habitat changes and loss – past to present

Changes to the seafloor and pelagic habitats of the greater Hauraki Gulf are far less well documented than changes in fished species populations, apart from a few fished species that are also major biogenic habitat formers (green-lipped mussels and, to a much lesser extent, scallops). In this section a range of historical accounts are summarised to provide some idea of the biological changes that have occurred. It does not cover all past human impacts, e.g., not covered are the extensive draining and channelisation of the Hauraki Plains following the passing of the 1908 Hauraki Plains Act; the large-scale removal of small stones/gravel deposits from many of the gulf island beaches for building aggregate; and quarrying of many of Auckland volcanic features, some to oblivion.

1902 Piako River fish

In March 1902 (Mair 1902), a Captain Muir was visiting the settlement of Pokatunawhenua about three miles up the Piako River (top of the Firth of Thames) and came across a Māori group using a fishing net known as a '*tarawa*'. It was described as a form of passive funnel net fastened across the channel using two sets of poles 6.1 to 7.6 m apart and deployed using long poles with forks in them to fasten it to the channel floor, and the top was attached to a cross-channel bar. It was deployed to fish passively with the tide running, every 15 mins the tapering end was lifted, and the catch emptied into a canoe. This net fished around 4.6 m of the 55-metre wide channel. From two lifts over a total of 45 minutes of fishing, they caught 581 eels (30–120 cm long), eight dozen flounders, large numbers of aua/kataha (yellow-eyed mullet *Aldrichetta forsteri*), 27 to 32 kg of the pilchard *Clupea sagax* (mohimohi, although he mentions "two varieties"), a few snapper, mullet, kahawai *Arripis trutta*, hundreds of young red cod 7–10 cm long, and juvenile 'rockcod' which he considered to be juvenile "kokopu"/"rawaru"/blue cod (*Parapercis colias*), 5–15 cm long. Local iwi called these fish "toitoi"/"panepane". "*Very large quantities*" of a whitebait species were also caught. At high tide the water was described as '*almost salt*', and '*only slightly brackish when low*'. He was certain about the red cod identification (mentioning '*the two feelers hanging from the throat*', and a little less so about the blue cod (Mair 1902). He also wrote that '*I have never heard before of either the young of the rock* (blue) *or red cod coming from freshwater before*.'

It is probable that his red cod were misidentified northern bastard red cod (*Pseudophycis breviuscula,* grows to 25 cm) or āhuru (*Auchenoceros punctatus*, grows to 13 cm), as all three are morid cods with barbells, and red cod, especially juveniles, are rare in the gulf (at least, in the present day). The fish thought to be juvenile blue cod were almost certainly one or more species of bully (*Gobiomorphus*) because the word 'toitoi' refers to these species, and possibly also juvenile torrent fish, known as 'panepane'. The larger sized fish (8–15 cm) may have been giant bully individuals, because this species lives in lowland waterways, sometimes including brackish waters. This fish assemblage from 120 years ago was an abundant one with a range of species. Today, the Piako River is heavily degraded, with its (lower plain) tidally influenced channels having mud-sides and a mangrove fringe, its waters mud-laden and turbid (see Figure 9) and surrounded by intensively farmed land. Present day fish assemblages are almost certainly impoverished; limited searching with beach seines in the Piako River for juvenile grey mullet in the mid-2010s found only a few small juvenile flatfish (M. Morrison, NIWA, unpubl. data).

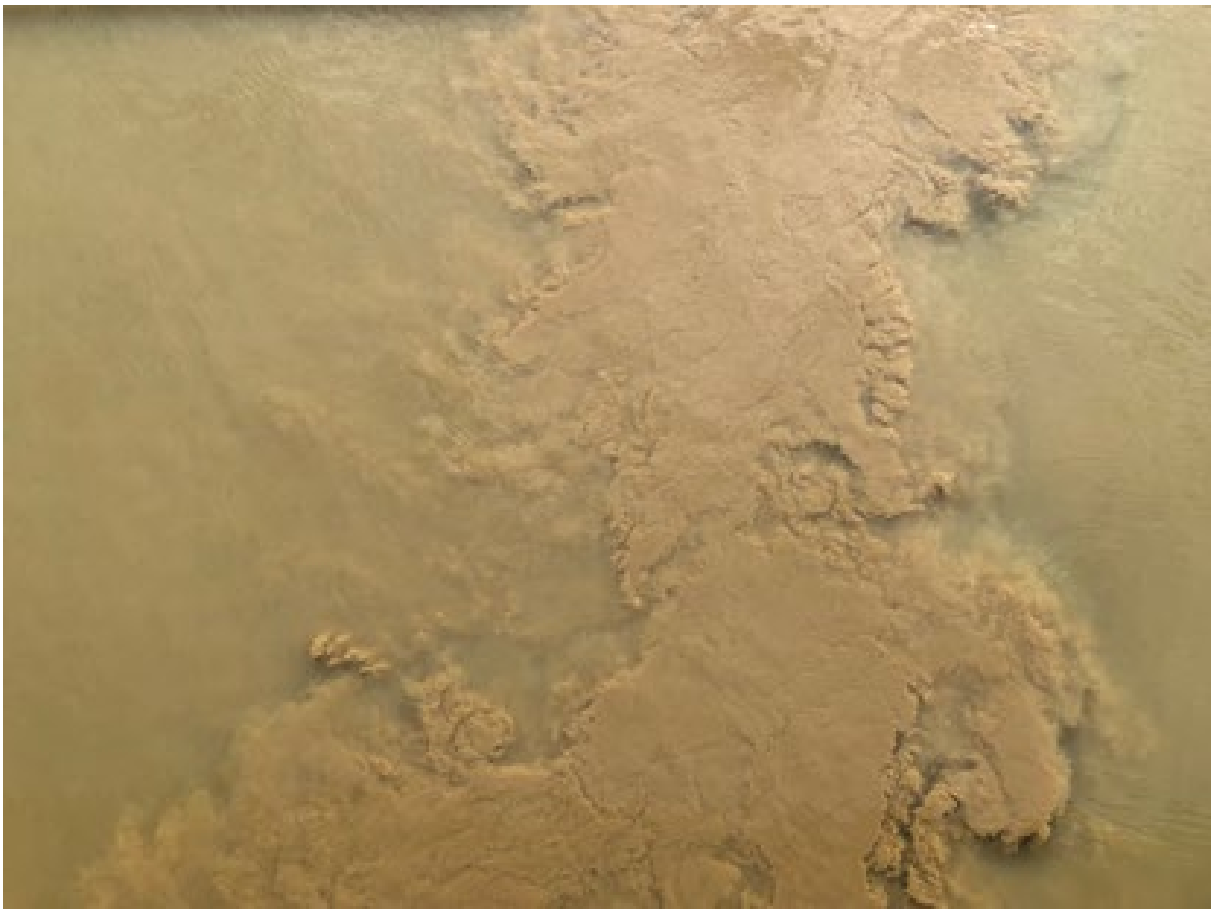


Figure 9: Current day water clarity under today's Kopu Bridge that crosses the Waihou River, which flows from the Hauraki Plains. Kopu Bridge is five kilometres downriver of the small settlement of Turua; which translates as 'twice seen', as one could historically "*simultaneously see the forest in the land and reflected in the river*" (Hatvany 2008). Today's turbid water quality, as well as the removal of the forest, makes this place name a reminder of the past. Source: Shane Kelly, Coast & Catchment Ltd.

1901 and 1907 exploratory trawl expeditions

The New Zealand Government funded exploratory trawl surveys of fisheries resources (finfish) around different coastal areas of New Zealand, including the Hauraki Gulf, in 1901 (Ayson 1901) and 1907 (Ayson 1908). Although focused on finfish, notes were kept on bottom types and some of the larger epifauna caught as bycatch, and these were summarised in map form (Figure 10) by Kelly et al. (2014). Biogenic habitat forming species included ‘bottom coral’ (more likely to be frame-building bryozoans rather than true corals, which only occur in mainland New Zealand coastal waters as small solitary cup corals), many (green-lipped) mussels (see following section), ‘horse mussels and sponges’ (mid-gulf in around 50 m water depth), and “*a net full of grass and weed*” in eastern Tamaki Strait. Based on present data observations in Te Rawhiti Strait in the eastern Bay of Islands (a small ‘coastal sea’ similar to Tamaki Strait, there is a high probability that the ‘grass’ may have actually been the seagrass species *Zostera muelleri* (either growing attached subtidally in situ (which would have been in around 7–10 m water depth, deeper than it grows in New Zealand today; or as drift from nearby intertidal/subtidal beds). Similarly, the ‘weed’ may have been from attached subtidal meadows of the fleshy green seaweed *Caulerpa* sp. (and/or several other soft sediment green/red/brown macro-algae species), which grows laterally using stolons and is associated with coarse shell and grit substrates. Within the present-day gulf, soft sediment *Caulerpa* beds are known to exist at Little and Great Barrier islands, and at Great Mercury Island, off East Coromandel.

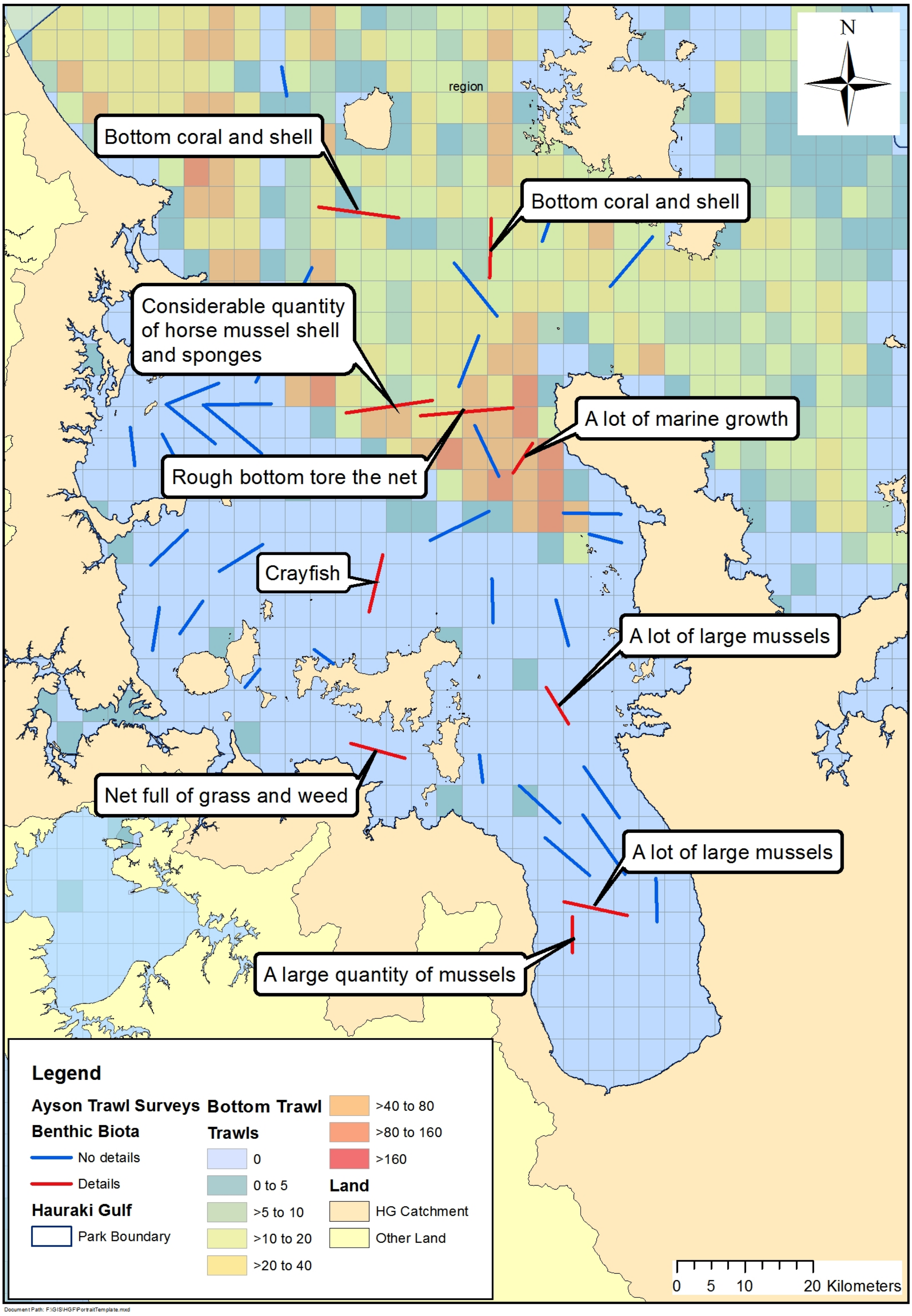


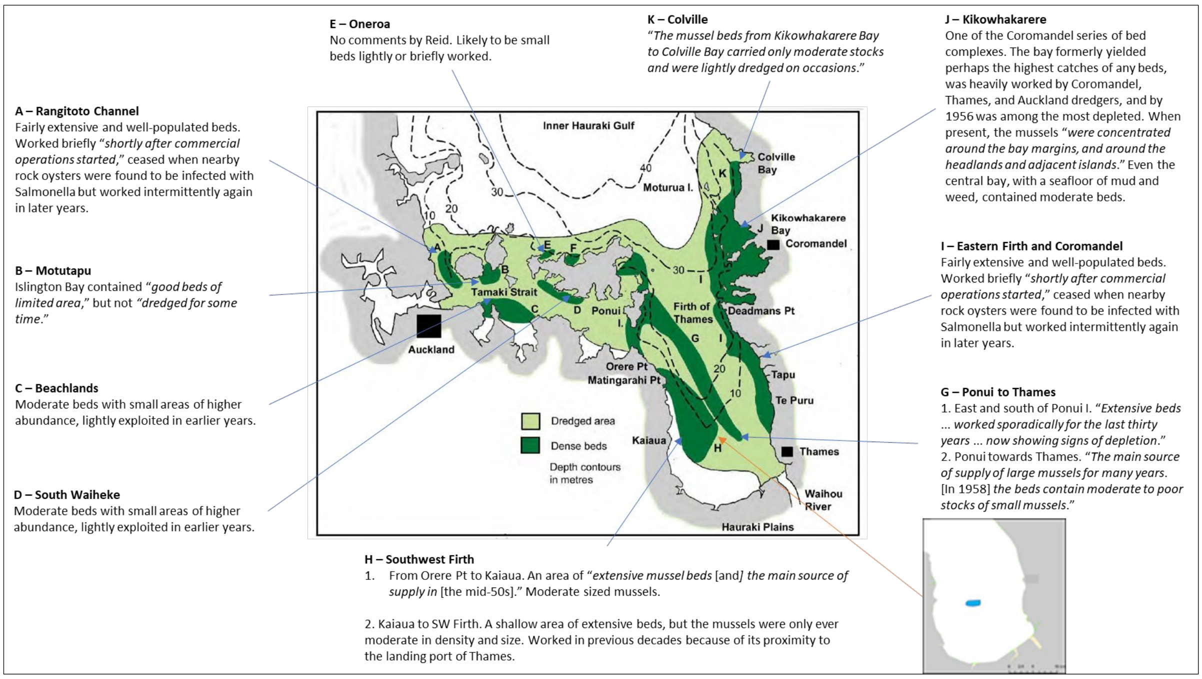
Figure 10: Stations with 'bycatch of note' from New Zealand Government trawling expeditions in 1901 and 1907 (from figure 6.8 of Kelly et al. 2014, see also Ayson 1901, 1908).

AD 1240–1800 archaeological sites, commercial wild fishery 1910s–1960s, green-lipped mussels

The inner Hauraki Gulf and the Firth of Thames once supported extensive green-lipped mussel beds growing on soft sediments, as well as on fringing rocky reef systems. Examination of 77 Hauraki Gulf middens over the period AD 1240–1800 found the exploitation of these species (measured as percentage occurrence) declined from 88% to 15% over this time. Over the same period, the percentage occurrence of pipi (*Paphies australis*) and cockles (*Austrovenus stutchburyi*) increased, and this was interpreted as a shift of Māori communities from rocky to estuarine shorelines, as crop cultivation developed (Smith 2011).

Mussel fishing as an industry was first noted in the New Zealand Marine Department Annual Report for 1913–14, where mussel curing and canning factories were noted as present at Tapu and Thames. By the 1920s a larger fishery started to develop, with the earliest beds worked inside Coromandel Harbour and in the island channels adjacent to the entrance (Paul 2012). Paul (2012) provides a comprehensive review of the life span of the fishery, building on earlier works including Reid (1969) and Greenway (1969). The fishery was dominated by the Strongman and Gundlock families. Throughout the 40 years of the fishery, 2–4 vessels operated full-time, with some additional casual boats, landing a high of 40 900 sacks of mussels in 1961 (estimated at 15 million mussels). Despite warning signs of over-exploitation and other stock issues over several decades, in 1960 2600 t were landed (green weight) and almost 2800 t in 1961 (the highest annual landing recorded). After this, annual catches started to drop away rapidly, even as searching time increased; with 1650 t in 1963, 850 t in 1964, and 100 t in 1966 (Paul 2012). By 1966 the fishery had collapsed (Greenway 1969). Full-time dredging ended in 1967, and landings in 1969. Beds were serially depleted; as the populations around Coromandel and the eastern firth shoreline declined, effort turned to the southern and western areas, then to Ponui Island, and finally to random searching over the whole area (Greenway 1969) (Figure 11). The prevailing theory at the time seems to have been that the populations were unsustainably fished down, but at the same time dredging had removed settlement surfaces for newly recruiting mussels, which were extracted as part of the mussel/byssus matrix. It was thought that a spatial refuge remained for populations on rocky reef areas, but that this was under threat from increasing recreational harvesting (Greenway 1969).

Acoustic and towed camera surveys in 2001 and 2002 of the Firth of Thames and the western inner gulf, respectively, using single beam acoustic (both surveys) and sidescan sonar (2001 survey only) methods, found no sign of bed recovery (only a very few mussel clumps seen all in the Firth of Thames, largest about 0.5 m^2) (Morrison et al. 2002, 2003). Investigation of an inner-mid firth location based on local knowledge, targeted with multibeam sonar, reported a larger bed (D. Immenga, University of Waikato, unpublished data, not sighted). Further direct searching by McLeod et al. (2012) found only a few scattered beds/remnants totalling about 0.64 km^2. Two of these were adjacent to mussel farms and were likely formed by live mussel drops: one in the outer west Firth of Thames off Waimango Point in 5 m water depth, the other in the Man Of War/ Ponui Passage off east Waiheke Island, in 20 m water depth. One other was in 0.5 m water depth in one of the Weiti River Estuary channels (an estuary adjacent to the Whangaparāoa Peninsula), and the final one in the Okiwa Estuary in 0.5 m water depth, on the northeastern coast of Great Barrier/Aotea Island.



E – Oneroa
No comments by Reid. Likely to be small beds lightly or briefly worked.

K – Colville
"The mussel beds from Kikowhakarere Bay to Colville Bay carried only moderate stocks and were lightly dredged on occasions."

J – Kikowhakarere
One of the Coromandel series of bed complexes. The bay formerly yielded perhaps the highest catches of any beds, was heavily worked by Coromandel, Thames, and Auckland dredgers, and by 1956 was among the most depleted. When present, the mussels "*were concentrated around the bay margins, and around the headlands and adjacent islands*." Even the central bay, with a seafloor of mud and weed, contained moderate beds.

A – Rangitoto Channel
Fairly extensive and well-populated beds. Worked briefly "*shortly after commercial operations started*," ceased when nearby rock oysters were found to be infected with Salmonella but worked intermittently again in later years.

I – Eastern Firth and Coromandel
Fairly extensive and well-populated beds. Worked briefly "*shortly after commercial operations started*," ceased when nearby rock oysters were found to be infected with Salmonella but worked intermittently again in later years.

B – Motutapu
Islington Bay contained "*good beds of limited area*," but not "*dredged for some time*."

C – Beachlands
Moderate beds with small areas of higher abundance, lightly exploited in earlier years.

G – Ponui to Thames
1. East and south of Ponui I. "*Extensive beds … worked sporadically for the last thirty years … now showing signs of depletion*."
2. Ponui towards Thames. "*The main source of supply of large mussels for many years.* [In 1958] *the beds contain moderate to poor stocks of small mussels*."

D – South Waiheke
Moderate beds with small areas of higher abundance, lightly exploited in earlier years.

H – Southwest Firth
1. From Orere Pt to Kaiaua. An area of "*extensive mussel beds* [and*] the main source of supply in* [the mid-50s]." Moderate sized mussels.

2. Kaiaua to SW Firth. A shallow area of extensive beds, but the mussels were only ever moderate in density and size. Worked in previous decades because of its proximity to the landing port of Thames.

Figure 11: Map of historical green-lipped mussel distribution, as created by Reid (1969), with associated notes on the fishery history of the (ten) individual main mussel beds. Also shown in the lower right smaller figure is the location of the mussel patch mapped by D. Immenga in about 2003. Source: adapted from figure 5 of Paul (2012), who redrew the map from Reid (1969), and added depth contours; the text is also from Reid (1969), as paraphrased in table 1 of Paul (2012); adapted from figure 2 2 of Thompson et al. (2020).

1930s versus 1990s inner Hauraki Gulf soft sediment invertebrate assemblages

Powell (1936) used a small 'naturalist's dredge' to sample stations located across the Waitematā Harbour, Rangitoto Channel, Tamaki Strait, Noises Islands, Whangaparāoa Passage, Kawau Bay, and Omaha Bay. He focused on sea-floor dwelling animal species; colonial organisms were left as general classes only (e.g., sponges), and macroalgae were excluded. Polychaetes were reported at variable taxonomic resolution. This work provides a rare insight into historical soft sediment animal assemblages of the gulf (sampled 1926–1935); with the caveat that by this time the Auckland isthmus had been cleared/logged of most of its native forest, the city of Auckland was well established, and commercial fishing had been operating for more than 70 years. Workers in the 21st century have found his work very useful, as effectively the only (semi-) quantitative account from earlier times. Five main species 'formations', and 12 species 'associations' within these were intuitively identified (based on numbers of individuals; without the statistical methods that exist today) (Figure 12, Table 5). Soft muds dominated by the heart urchin *Echinocardium cordatum* were the spatially more extensive foundation, with all other groupings being associated with channel areas with higher current speeds, and the Noises Islands. Powell commented that "*…the faunas for the soft and hard-bottom stations respectively are almost completely different. It is at once evident that the hard bottom supports the richer fauna both in respect to variety and to quantity. The average number of species and specimens for the soft-bottom series is very low…*". Table 5 describes the dominant species in each grouping.

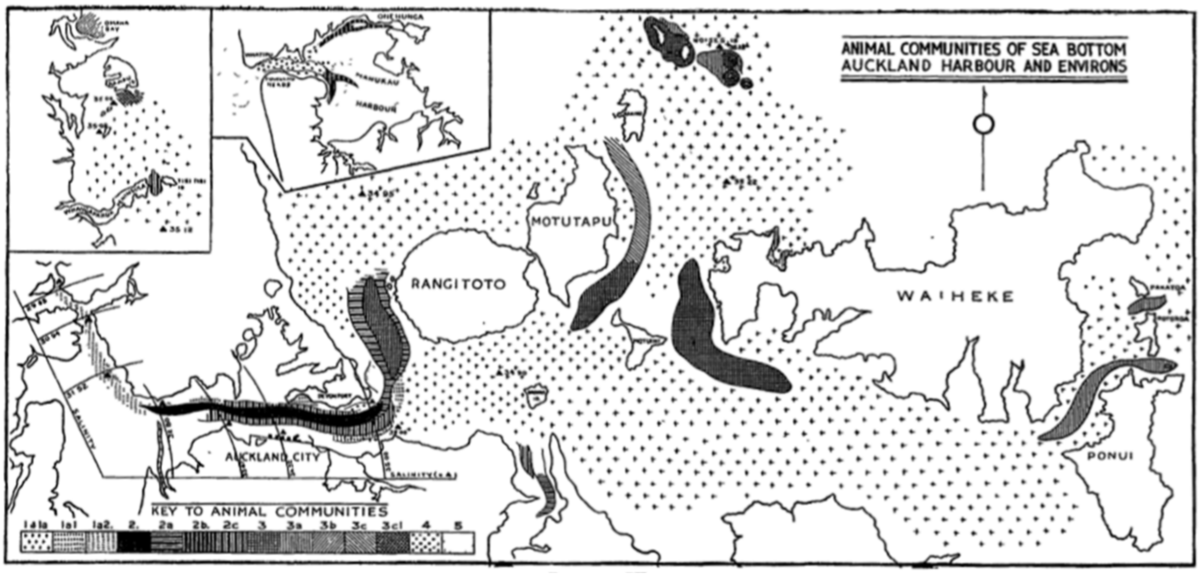


Figure 12: Soft sediment animal assemblages of the inner south-eastern Hauraki Gulf (figure 4 of Powell (1936); see Table 5 below). Note that Foundation 4, though shown in the key, was not found in the gulf by Powell. Key: Note that the numbers of stations sampled vary across formation/associations, and species/OTU richness is not corrected for this. Species names with revised taxonomic status (since 1937) are updated following Hayward et al. (1999) and Cook (2010, New Zealand Coastal Marine Invertebrates Volume I).

Table 5: Powell (1936) description of the faunal assemblages found in the inner south-eastern Hauraki Gulf. (Continued on next page)

Formation 1, *Echinocardium* formation (dominated by 3 dominant species – heart urchin *Echinocardium cordatum*, the bivalve *Dosinia lambata*, and the small brittle star *Amphiura rosea*), on soft muds/fine sands.

Association	1, 1a	(as above), & 1a (*E. cordatum* absent, 2 other dominants present). 59 species.
	1a1	(3 dominants absent, replaced by polychaetes and the bivalve (razor clam) *Zenatia acinaces*). 21 species.
	1a2	(3 dominants absent, polychaetes dominate). 12 species.

Formation 2, *Maoricolpus* formation (dominated by gastropod *Maoricolpus roseus* (turret shell) at very high densities, on hard bottom of coarse shell-debris. 62 species.

Association	2a	*Maoricolpus* and *Dosina* association (*Maoricolpus* less dominant, 5 other dominants are the bivalves *Dosina zelandica*, *Venerupis largillierti*, *Corbula zealandica*, and the gastropods *Trochus tiaratus* (top shell) (herbivorous) and *Cominella quoyana* (carnivorous whelk). 77 species.
	2b	Not relevant, only found in Manukau Harbour. 30 species.
	2c	*Venerupis*, *Dosina*, and *Maoricolpus* association. 14 species.

Formation 3, *Tawera* and *Tucetona* formation (dominated by the bivalves *Tawera spissa*, *Tucetona laticostata* (large dog cockle), the half-crab *Petrolisthes elongatus*, crab *Notomithrax minor*, green-lipped mussel *Perna canaliculus*, and the gastropods *T. tiaratus* and *Murexsul octogonus* (spiny murex); on hard bottom of worn whole bivalve shell and comminuted (fragmented) shell. 142 species.

Association		
	3a	*Tawera* and *Venericardia* association (dog cockle *T. laticostata* absent; dominated by the bivalves *T. spissa* and *Venericardia purpurata*, and the gastropod *C. quoyana*), increased silt and fine sand sediment fraction. 55 species.
	3b	*Tawera* and *Astropecten* association (dominated by bivalve *T. spissa*, the starfish *Astropecten polyacanthus*, and the gastropod *Cominella adspersa* (predatory whelk). 28 species.

3c *Flabellum* and *Notocorbula* association (live *T. spissa* and *T. laticostata* absent; dominated by cup-coral *Flabellum rugulosum* and the bivalve *N. zelandica*; seafloor of dead whole shells, mainly *T. spissa*. 46 species.

3c1 *Corbula* and *Pleuromeris* association (dominated by the bivalves *N. zelandica* and *Pleuromeris zelandica*). 14 species.

Foundation 4, *Fellaster* formation (dominated by the sand dollar *Fellaster zelandiae*)

NB This association was not recorded in the Hauraki Gulf by Powell (but was recorded from the Manukau Harbour). Hayward et al. (1997) recorded it northwest off Takapuna Beach, between Bean Rock and Rangitoto Island, and in narrow bands off Devonport and Hobson Bay. Also seen off Omaha Beach (Taylor & Morrison 2008), and across Buffalo Bay, East Coromandel (M. Morrison, NIWA, pers. obs.). Species = N/A (not seen in Hauraki Gulf in 1930s samples).

Foundation 5, *Amalda* formation (dominated by the olive *Amalda australis*, the spired gastropod *Euterebra tristis*, and the whelk *C. adspersa*, shallow seafloor of fine clean sand. 12 species.

5 As above

To assess faunal changes six decades after Powell's work, from 1993 to 1995 Hayward et al. (1997) sampled a range of sites extending from the Waitematā Harbour out into the Rangitoto Channel, using a small dredge like Powell's. They reported "*a similar gross pattern to the 1930s*". Fourteen molluscs (largely carnivorous gastropods) were no longer seen, and Powell's groupings of '*Tawera–Tucetona*' (Formation 3) and '*Amalda*' (Foundation 5) had disappeared from the outer harbour. They also noted reductions in the abundance and range of the turret shell *M. roseus*, and various shelly channel sediment associated species in the harbour's centre. Nine or more New Zealand mollusc species and a crab species were reported present for the first time, and nine others apparently increased in abundance. North-east of North Head, a new extensive bed of horse mussels (*Atrina novaezealandiae*) was discovered (Figure 13). Three new invasive bivalve species (the file-shell *Limaria orientalis*, Asian semele *Theora lubrica*, and Asian date mussel *Musculista senhousia*), which first appeared in the 1960s and 1970s, were found to have become so abundant that they were now co-dominant species in six of the eight associations still present (Hayward et al. 1997).

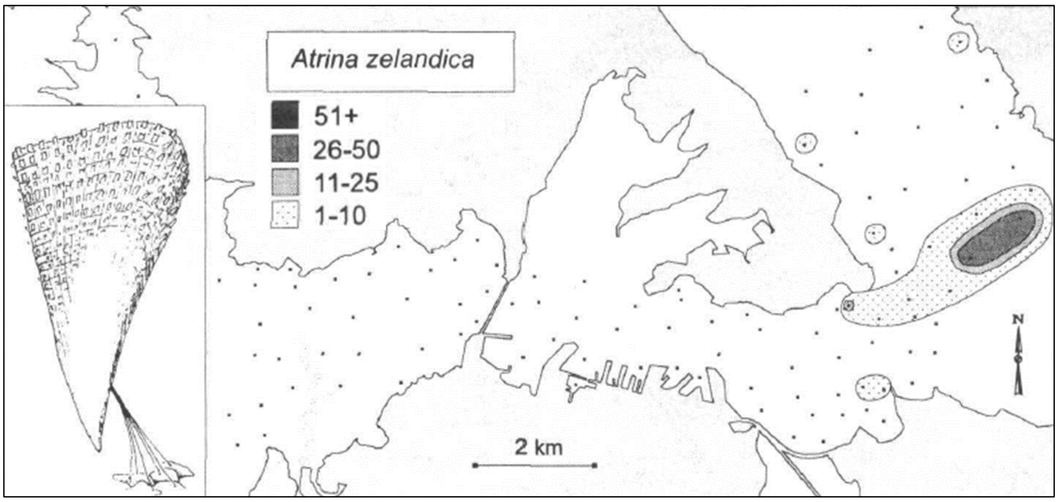


Figure 13: Contoured distribution and abundance map for the horse mussel *Atrina zelandica* (1993/94 sampling). Data standardised as number of live organisms per 20 litres of sediment (approx. per 0.2 m^2 of seafloor) (from figure 5 of Hayward et al. 1997).

1930s to present seagrass (especially subtidal) meadows

New Zealand has one species of seagrass, *Zostera muelleri*. Although primarily an intertidal plant, it can grow in the subtidal zone where water clarities permit, where it can grow down to depths of two to three metres below low tide. Although no historical maps of its distribution and abundance in the gulf exist, in the 1930s Powell made explicit notes of its decline in the Waitematā Harbour and adjacent areas (Powell 1936) (e.g., Figure 14). For Hobson Bay, he noted that the building of the waterfront road with two narrow openings for water flow had resulted in associated channel floors of hard shell being formed, with areas of soft mud being increasingly shelly, and "*the Zostera (seagrass), once abundant in the bay, has now almost entirely disappeared*". More broadly, he also commented that "*tide-deflectors and reclamation works elsewhere in the harbour have considerably reduced the area of Zostera*", and that this was likely to have reduced the food supply available to fish, including snapper.

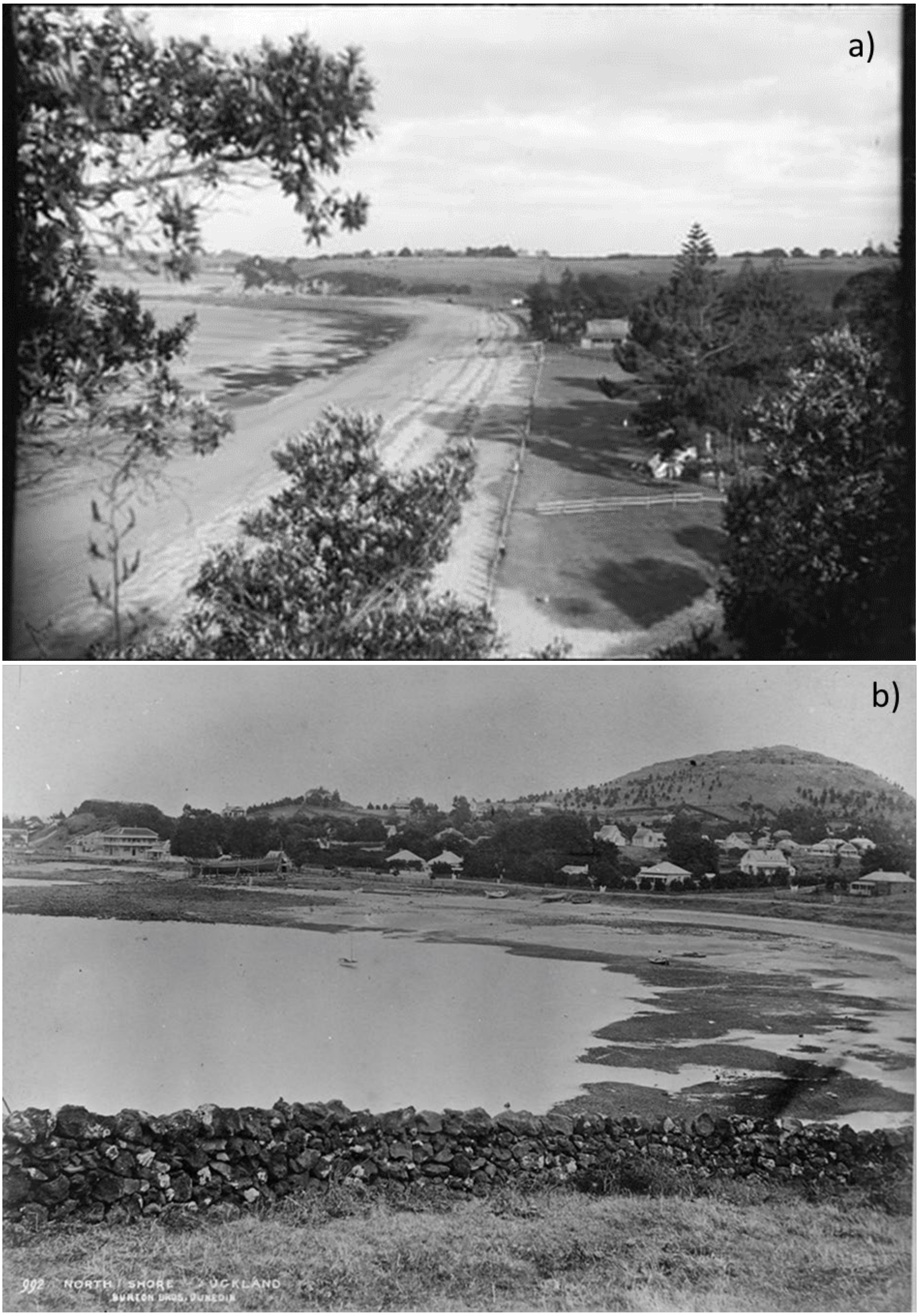


Figure 14: a) Mission Bay in 1908. What appears to be seagrass meadow is visible along the low-tide area, as well as possibly seagrass wrack (drift) along the upper tide mark of the beach. b) Torpedo Bay, Devonport; looking across Torpedo Bay to the Masonic Hotel, 1879. The dark patches on the right side are very likely to be seagrass and show linear marks characteristic of small boats being dragged over seagrass. Source: left) Sir George Grey Special Collections, Auckland Libraries 4–2879; right) Source: Bartley Family Archive Image SGGSC APL 4-2979.

Other smaller local scale seagrass losses have also been noted. Morley et al. (2001) reported that in the 1940s and 1950s there used to be extensive seagrass areas at low tide on Howick Beach (eastern Tamaki Strait), where sea horses and piper were common. In the 1950s, seagrass was reported to have disappeared from the entire Waitematā Harbour, following a fungal disease (Armiger 1964, 1965). In 2001, small patches were reported to be present at low tide (Morley et al. 2001).

Aerial photography has been used to track seagrass extent changes at Cox's Bay. Lundquist et al. (2018) used aerial photographs and satellite imagery to assess seagrass bed temporal change in Cox's Bay and surrounds (in the Waitematā Harbour) (Figure 15). In 1940, the extent of seagrass covered 7.29 ha. In 1996, seagrass was virtually absent (0.1 ha). From 2001 to 2005, seagrass cover varied between 3 and 4 ha, increasing to 6.41 in 2007, and continued to increase in extent until the most recent 2015 imagery, where it covered 43.51 ha. Concurrent benthic sampling (2000–2015) found an associated increase in macrofaunal richness and abundance, as well as of mud content and organic matter material. The authors suggested that "*the changes over time in sediment characteristics and macrofaunal community structure and abundance following colonisation by seagrass suggest the potential for temporal dynamics of seagrass meadows to affect estuarine ecosystem services*".

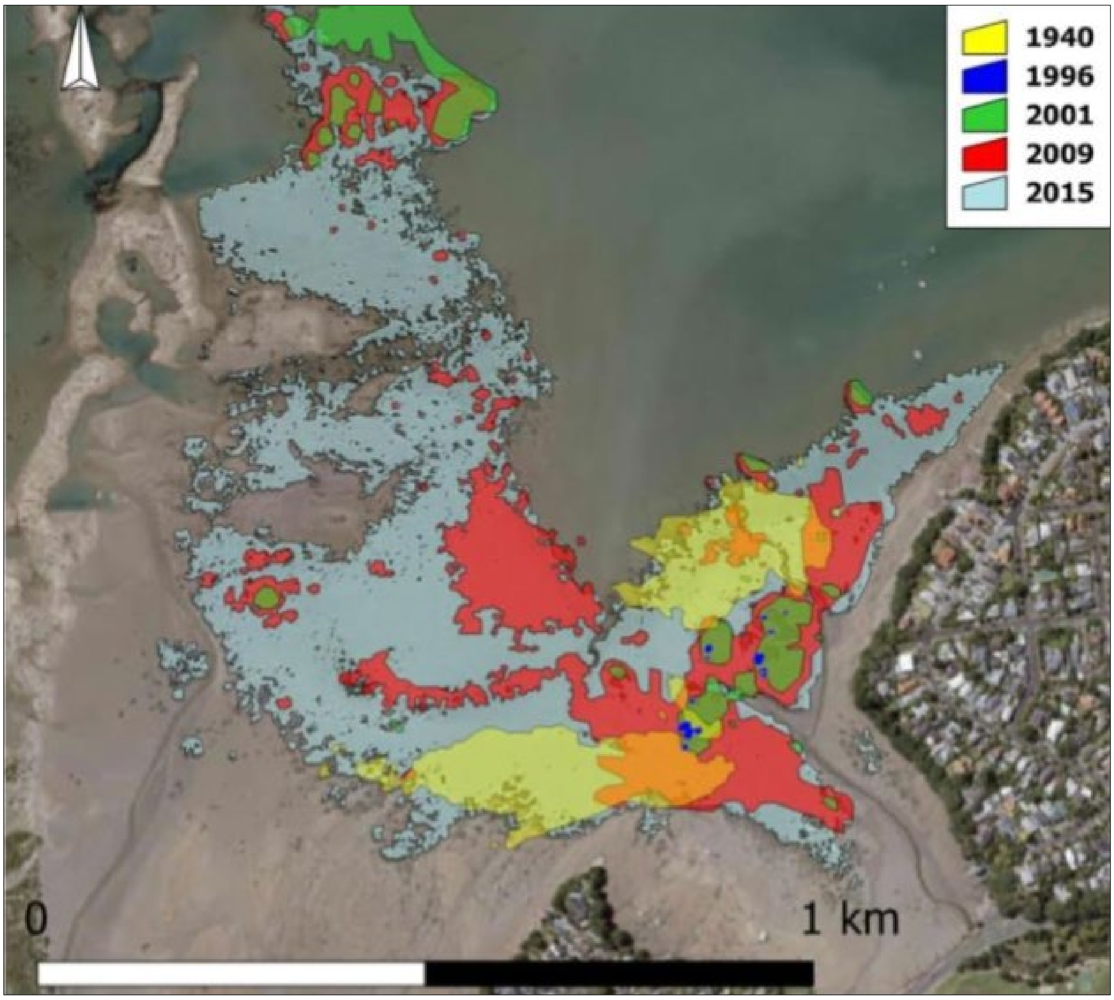


Figure 15: Temporal changes in seagrass cover from 1940 to 2015 at Cox's Bay, Waitematā Harbour. Source: figure 1 of Lundquist et al. (2018).

Loss of 'live' ground (mixed biogenic habitat species assemblages)

A term often used by commercial fishers to describe seafloor that holds a great diversity and abundance of invertebrate life is 'live ground'. The species group composition of such areas is not well known, but is likely to include assemblages of bryozoans, sponges, hydroids, horse mussels, and other epifauna that are caught in nets as bycatch. In general discussions with commercial fishers, active and retired, there appears to be a general feeling that the extent of live ground over the gulf has greatly reduced over the last several decades or more. Anecdotal oral stories from the 1970s and 1980s help visualise some of the changes that may have been brought about (noting that older oral accounts do not exist, because generations have passed on). These accounts are from all around New Zealand. These accounts are not given here to direct blame to activities from a different era and generation/s with different world views; but rather to suggest that some areas of the gulf are now very different from the not-so-distant past, that these changes have never been documented in recorded form, and that we would remain unaware of such large changes without knowing these stories. By knowing (even broadly) of these changes, society now has the potential option to work towards restoring these areas back towards more natural and productive habitats.

Activities described during conversations with retired/older fishers (not necessarily those who did the activities) by the author included the following:

- Horse mussel beds were hard on older trawl nets (pre-synthetic netting) due to their sharp edges that cut and damaged the nets, as well as the fish catch. Steel wires, bobbins, and water filled drums were dragged by vessels across areas of seafloor to eliminate horse mussel beds, ‘condition the ground’, and make them less damaging to fishing gear.
- Towed wires and other items were used to remove the kelp forests of North-West Reef (west of Little Barrier/Hauturu Island), to allow subsequent trawling over the low ground that much of the system is composed of (Figure 16). Limited multibeam work (2011) over part of this reef shows that although there are several large rock stacks, much of the ground surrounding these stacks is low undulating ground, which limited towed video (in 2020) suggests is hard compacted coarse sediments and shells that form topographically diverse seafloor mound arrays and other features. Epifaunal species appear to be sparse on these features (as observed in February 2020). This general area still returns some of the highest combined catch weights of the Hauraki Gulf (Figure 16).
- Old trawl nets were used to clear the rich biogenic habitat ground between Channel Island and the end of Coromandel Peninsula (Figure 16), followed by standard trawling to extract the fish biomass now open to harvest (including 20 to 30 tons of large snapper).

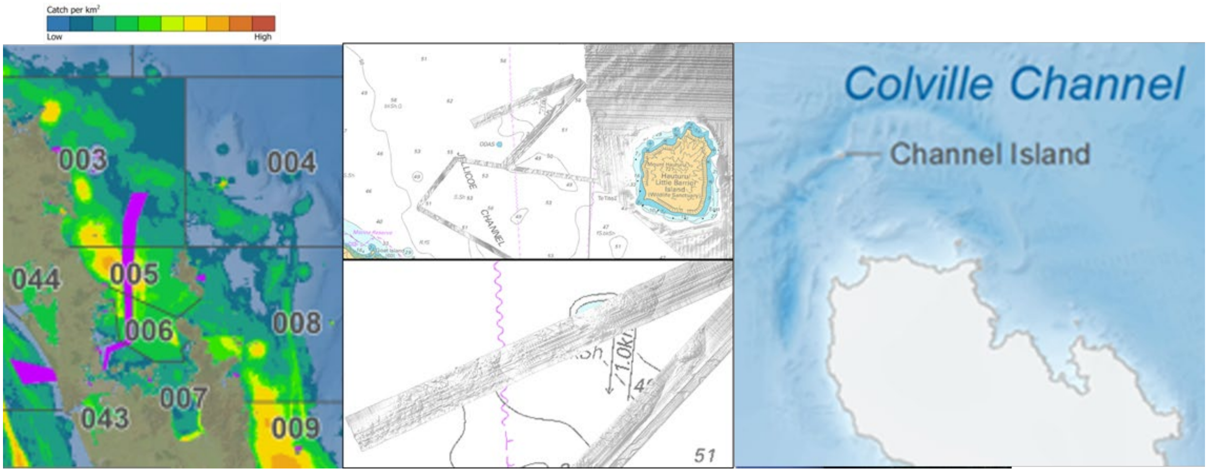


Figure 16: **Left) New Zealand commercial fishing intensity (all fishing methods October 2007–September 2018); centre, NIWA multibeam sonar bathymetry transects over North-West Reef, due west of Great Barrier Island, showing seafloor variability to the west of the rock stacks of the reef proper, in and outside the Cable Protection Zone (purple wavy line); right) Channel Island to Cape Colville area, top of Coromandel Peninsula.**

Work commissioned by Sea Change in 2016 used a towed video camera to assess whether there is recovering ‘live ground’ inside the Hauraki Gulf Cable Protection Zone (CPZ), which is protected from all forms of bottom contact from human activities. There has been a designated ‘prohibited cable area’ within the Hauraki Gulf from about 1920 but the current day cable protection areas as defined in the Submarine Cables and Pipelines Protection Order 2009, 2006 (SCPPO) and associated legislation—the Submarine Cables and Pipelines Protection Act 1996 (SCPPA)—have been in place since the early 1990s. Initially there were no cable patrols (sea or air), but during the survey of the CPZ for the introduction of the Southern Cross Cable Network in late 1998, a considerable number of trawling scars were detected on the seabed within the CPZ. The discovery of these trawl scars prompted the cable owners to fund and introduce both sea and air patrols in 1999, which are still operational today (Mike McGrath, Spark New Zealand, pers. comm.).

Five blocks along the cableway were assessed using individual 200-m long transects going east and west from the cableway centre, where the cable was presumed to sit (Figure 17) (Morrison et al. 2016). Sampling was done during the hours of darkness, to capture diurnal fish species sleeping on the seafloor as well as nocturnal species that come out at night to forage.

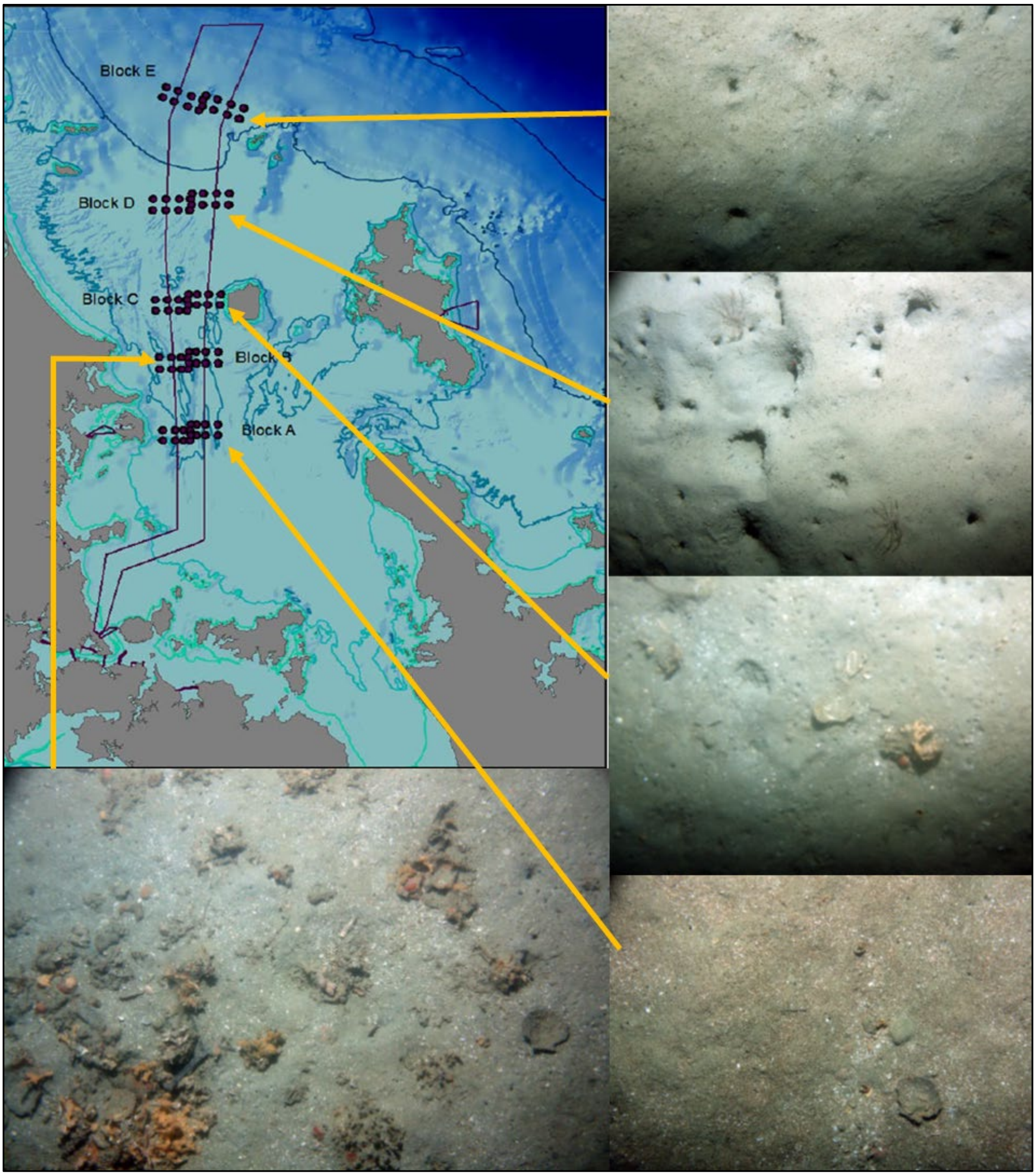


Figure 17: Seafloor images from 5 areas of the Hauraki Gulf, taken in 2016. Blocks A and B show coarser seafloor sediments, block C has muddier sediments, and blocks D and E have soft mud with burrows out on the continental shelf. Depth ranges from 35 to 129 metres. Block B shows small discrete 'clumps' of biogenic material (cemented shell, bryozoans, hydroids, sponges, and other contributions), scored as 'indeterminate biogenic clump' by Morrison et al. (2016).

Univariate and multivariate statistical models were used to assess whether the species assemblages varied inside and outside the CPZ. For univariate measures, many factors were significant for assemblage level response variables (e.g., species richness and abundance), as well as for many individual species. However, the effects of block and depth were much stronger than that of CPZ status, and in general effects were relatively modest. Overall, the effect of the CPZ was negligible.

However, of interest was the presence of indeterminate biogenic clumps. No direct physical sampling was possible, but visual estimations suggested these to be a mixture of dead shell and other biogenic debris, along with some modest levels of living fauna including sponges, bryozoans, and possibly hydroids. This is likely to represent some of the once live ground that fishers refer to. It is not known whether these represent a stable state, or part of a trajectory of change, either increasing or decreasing over time. In Foveaux Strait, Cranfield et al. (2004) suggested a succession model of recovery after fishing (oyster dredging) ceased, in which the macrofauna (epifauna) passed through a series of compositions, each increasing in biological complexity. Its progress was dependent on both the time since fishing ceased and on the proximity of the sources of re-colonising propagules.

It is possible that these indeterminate biogenic clumps in the Hauraki Gulf represent a stage in succession, but equally they might simply represent relic material from the past, with low levels of present-day fauna utilising the hard surfaces available. The value of less disturbed seafloors was quantified by Handley et al. (2014) for the closed area (since 1980) off Separation Point, between Tasman and Golden bays at the top of the South Island. Sites inside (unfished) and outside (fished) were compared for their benthic assemblages. Fished sites were dominated by fine mud, with little to no shell-gravel, fewer species, and a loss of large bodied animals, with an associated reduction in biomass and productivity. Conversely, at the unfished sites, larger rarer molluscs were more abundant and contributed the most to size based estimates of productivity and biomass. Fished sites had increased scavengers, predators, and deposit feeders, whereas the unfished sites favoured filter-feeders and grazers. They proposed that the colonisation of biogenic species in protected sites was contingent on the presence of shell-gravel atop the soft sediments, and that fishing removed this shell-gravel (Handley et al. 2014). Put another way, unfished sites were more productive, held more potential prey for fish feeding, and supported higher densities of biogenic habitat formers (in this case molluscs). Of note, the Separation Point area was actually closed to protect large field of bryozoans (dominated by the large frame-building species *Celleporaria agglutinans*), which were considered to be critical habitats for juvenile fish (snapper, tarakihi, red cod, blue cod, and others) (Saxton 1980a & b, Mace 1981).

Handley et al. (2014) did not work on the bryozoan fields, which occur deeper and further from shore than where their sites were placed. As part of NIWA's MBIE-funded Juvenile fish habitat bottlenecks research programme, in September 2020 almost the entire Separation Point closed area (140 km^2), as well as a one-kilometre buffer, was mapped using multibeam sonar; this was followed by quantification of the bryozoans fields importance as important fish nurseries in February 2021. Initial observations are that the bryozoan fields of Separation Point no longer exist as of 2020/21, with sedimentation being the likely cause. Formal analyses of these data are scheduled for 2021/22. Work on the secondary productivity (invertebrate assemblages) of bryozoan beds (and several other juvenile blue cod nursery habitat types) is also underway for a diverse bryozoan bed dominated by *C. agglutinans* occurring on the west side of D'Urville Island, Marlborough Sounds. Initial observations show a diverse and high-density secondary fauna, with the likelihood that such habitats will also turn out to be highly productive and valuable, as shown for unfished mollusc shell seafloors (Handley et al. 2014) and subtidal soft sediment green-lipped mussel beds (McLeod et al. 2012).

Possible loss in carrying capacity—reduced growth rates in fished snapper from 1990s to 2010s

Although the snapper fishery of the Hauraki Gulf has been operating for more than a hundred years, sampling of commercial landings in port (known as market sampling) for their length and age compositions first started in 1963 and was undertaken intermittently until a more structured sampling programme was established in the 1988–89 fishing year (Davies et al. 1993). Sampling was undertaken every year until 2009–10, when the sampling intensity was reduced to once every three years, with the most recent published sampling being from the 2017–18 fishing year (Walsh et al. 2019a). Within each sampling year's survey (23 in all), a random series of fish are selected from the fisheries landings and processed for age (using otolith/ear bone annual rings), length, and sometimes weight. These data are used to monitor the fishery over time and provide an important data input for stock assessments and quota setting.

Figure 18a shows the weight-at-age estimates for each of the 23 surveys during the spring-summer period for longline caught snapper (plotted against their sampling year), for fish age classes ranging from 3 to 19 years old (fish less than 3 years of age and, since 1995, 4 years of age are below the minimum legal length allowed for landing). These data show a gradual long-term decrease in the mean weight-at-age for snapper for most of the age classes, i.e., a temporal decline in growth rates, with fish in more recent years taking more time to grow to a given weight than in earlier years (Walsh et al. 2019a).

Figure 18b shows mean weight-at-age data from the two most recent surveys (2012–13; and 2017–18) and compares it against the average weight-at-age from combined surveys from previous decades (1990s and 2000s). The mean weight-at-age of the common age classes drops over the three decades of survey, indicating reductions in average growth rates over time. The difference in mean weight-at-age between the first decade (1990s) and 2017–18 indicates the overall net weight loss to be estimated conservatively at around -34% for most of the common age classes (i.e., 5- to 15-year-old snapper) in the Hauraki Gulf. Similar declining growth rates have also been documented for snapper in the Bay of Plenty (-24%) and East Northland (-10%) which together with the Hauraki Gulf comprise the SNA 1 stock; as well as for the west coast SNA 8 stock (Walsh et al. 2014a, 2017, 2019b).

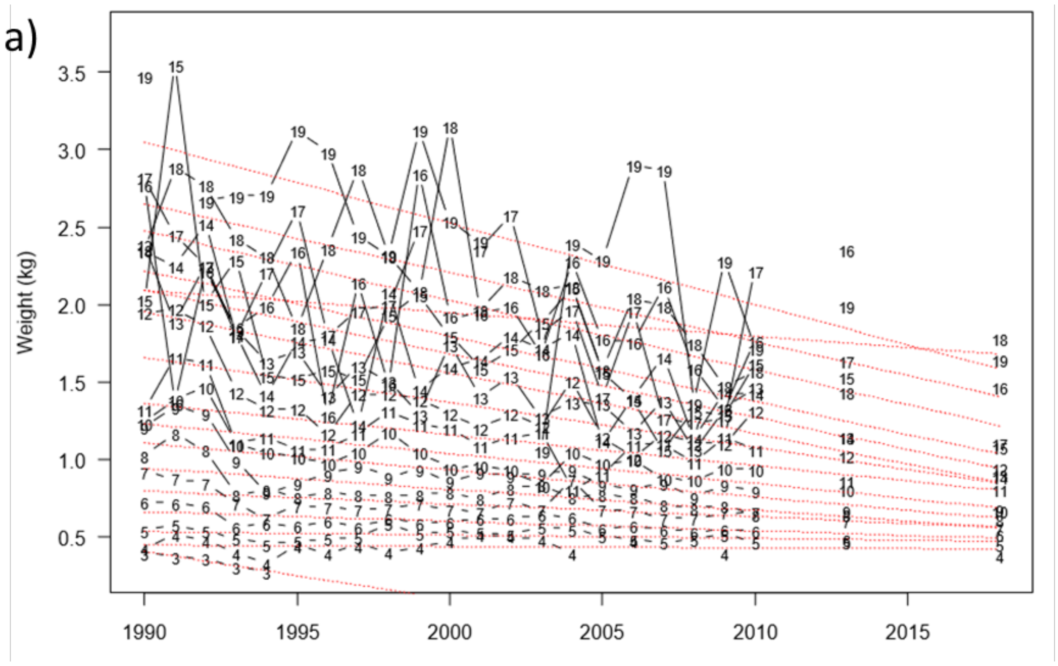


Figure 18a: **Mean fish weight-at-age estimates for 3- to 19-year-old snapper sampled from the Hauraki Gulf bottom longline fishery (during spring–summer) between 1989–90 and 2017–18 with fitted trend lines (dotted) for each age class number (given as fish age, e.g. 5 = 5+ years old) depicting long-term changes in growth rates over the 29 year period. Source: modified from figure 22 of Walsh et al. (2019a).**

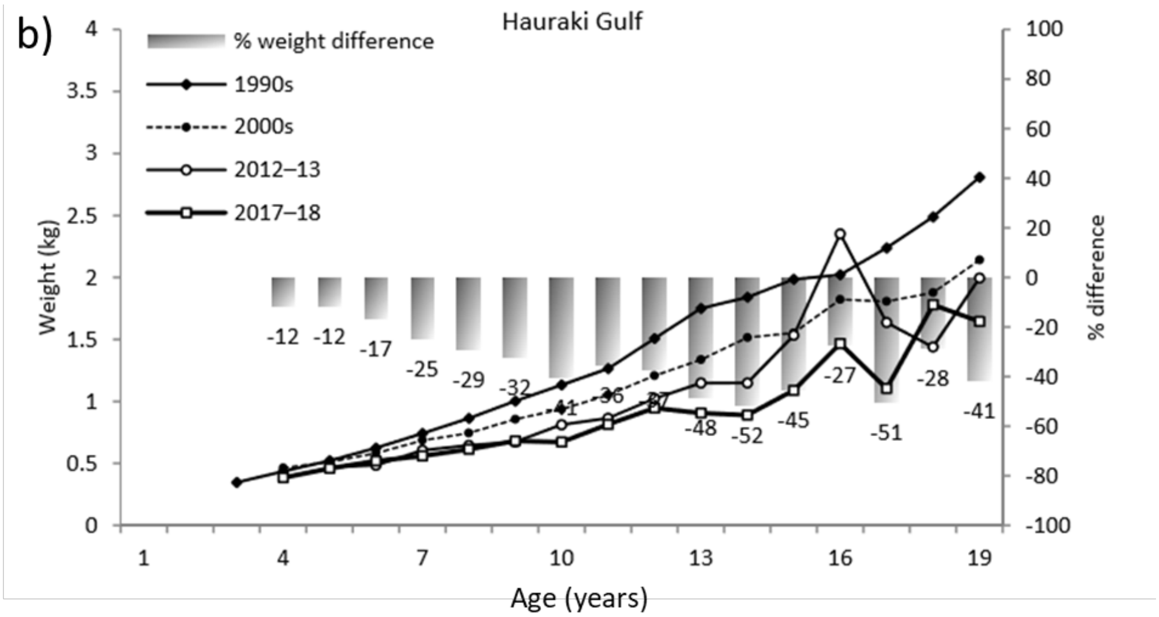


Figure 18b: Mean weight-at-age estimates for Hauraki Gulf snapper sampled from the bottom longline fishery from two distinct decadal time periods and from the two most recent sampling years 2012–13 and 2017–18, and where each period reflects the average mean weight-at-age for those years. The percentage weight difference for each age class (positive or negative) is the difference between the first decade (1990s) and 2017–18, and indicative of a net weight gain or loss in mean weight-at-age through time. Note: for comparative purposes over time, only spring-summer samples have been presented. Note that the annual mean weight-at-age estimates for many of the older age classes (i.e., over 13 years of age) appear highly variable from year to year and are unlikely to provide realistic estimates due to the low number of individuals present in those older age classes. Source: modified from figure 20 of Walsh et al. (2019a).

This decline in growth rates has been suggested to be likely attributable to compensatory density dependence (Rose et al. 2001), because recent stock assessments suggest biomass has increased through this period (Francis & McKenzie 2015), rather than temperature related effects; or bias in the sampling design or fishing method selectivity. In the past, the Hauraki Gulf contributed most of the SNA 1 fishery catch (Walsh et al. 2014b) but now contributes the lowest catch of the three SNA 1 sub-stocks (e.g., Hauraki Gulf landings dropped by more than 30% between 2013–14 and 2017–18). Commercial fishing effect has shifted towards East Northland and the Bay of Plenty, where snapper are of larger average size and better quality (colour/condition) than in the gulf, and (anecdotally) valuable bycatch species are more readily available (C. Walsh, Stock Monitoring Services, pers. comm.).

Collectively, these findings suggest that as snapper stocks recover from past overfishing and increase in fish biomass and density, food limitation may be occurring, with the carrying capacity of the Hauraki Gulf having significantly declined from what it was back when snapper had yet to be heavily fished. Though yet to be tested, this may be directly linked to the extensive loss of 'live ground', the 500 km^2 of green-lipped mussel beds, kelp beds, horse mussel beds, and other biogenic habitats likely to provide higher levels of prey for snapper and other predatory fish, as well as declines in the prey assemblages of bare sediment seafloors (e.g., filter-feeding bivalve species including cockles, pipis, and other species forming dense infaunal beds). Snapper also prey on pelagic species such as anchovies, which have been reported by fishers to have had bad spawning years and have been speculatively linked to other issues such as skinny kahawai and mass mortalities of little blue penguins. There may be less food available in the Hauraki Gulf at all levels of the food chain than historically.

4. HABITATS OF THE HAURAKI GULF

4.1 Seafloor mapping of the Hauraki Gulf's habitats

Restoration of marine habitats and ecosystems requires knowledge of what was present in the past before significant degradation started, and what exists in the present day. Without this information, it is not possible to set restoration goals and objectives. Such knowledge is inherently spatial and involves mapping. Many approaches and technologies are available for mapping marine systems, including aerial -based platforms (satellites/planes/drones) with photography, LIDAR, and multispectral sensors (e.g., CASI) for shallower systems with good water clarity, and vessel-based multibeam, sidescan, and single-beam sonars for deeper waters. Most of these technologies are relatively recent in their creation and use, with respect to human impacts on marine ecosystems; for example, intensive aerial photography from planes was largely introduced to New Zealand when many pilots returned to New Zealand after World War II. Mapping further back in time requires more reliance on other approaches, such as fisheries catch records, and the death assemblages sometimes preserved in seafloor sediments from harder-bodied animals. Internationally, many countries are putting significant resourcing into mapping their marine estates, for resource inventorying of biological and non-biological resources, and as part of the infrastructure needed to build 'blue economies'.

For example, Gerovasileiou et al. (2019) assembled data from a wide variety of sources for Europe, to provide a census of available map resources at the European scale focusing on: a) key marine habitats; b) degraded habitats; c) human activities and pressures acting on degraded habitats; and d) the restoration potential of degraded habitats. Of 580 map records, around half were from the grey literature/web resources and provided no georeferenced files (maps) for download. Habitat degradation was assessed in only 28% of map records and was largely qualitative. Less than half of the map records included assessments on degraded habitats recovery/restoration potential, and passive restoration by removal of human activities was the most recommended measure. They concluded that "*comprehensive mapping of habitats using ground-truthed high-resolution techniques and covering all European Seas should be the ultimate target in marine habitat mapping in Europe. This will serve as a baseline to monitor changes, and as a tool to ensure spatial planning initiatives and conservation actions to be undertaken using the best available knowledge to act beyond the 2020 headline target and enables meeting the 2050 vision of the EU Biodiversity Strategy: "European Union biodiversity and the ecosystem services it provides — its natural capital — are protected, valued <u>and appropriately restored</u> for biodiversity's intrinsic value and for their essential contribution to human wellbeing and economic prosperity*".

They recommended that future key marine and degraded habitat mapping initiatives focus on:

- Making geo-referenced spatial data freely available (supplementary files in peer-reviewed papers; online repositories)
- Enabling free and open access to grey literature (e.g., through online repositories)
- Production of high resolution and fine-scale habitat maps based on comparable or harmonised methodologies
- Ground-truthing of habitat maps and reporting model uncertainties, especially in cases of habitat modelling
- Filling thematic gaps concerning specific habitats (e.g., hard substrate and deep-sea habitats)
- Filling geographical gaps regarding specific (sub-) regions
- Filling temporal gaps through the digitisation of old/historical maps

New Zealand has also recognised the value of seafloor mapping, e.g., through the Oceans2020 initiative and other programmes. However, most of its marine estate remains to be mapped (see DOC's 'SeaSketch' GIS project for some data displays). The Hauraki Gulf is no exception. Although navigational charting has been carried out since the early days of Europeans arriving in New Zealand, specific seafloor habitat mapping is a much more recent endeavour. Work in the 1990s and early 2000s has involved mainly aerial photography, single beam acoustic sounders interpreted by software such as

QTC View, and sidescan sonar for small localised areas such as estuaries and shallow bays. More recent work has utilised multibeam sonar, which maps both high-resolution bathymetry and backscatter (reflectivity/hardness of the seafloor) (e.g., Morrison et al. in review, a–c). Each mapping approach has its own strength and weaknesses, but generally multibeam sonar is seen as the state of the art. However, because it maps four times the water depth from the vessel it is deployed, it becomes ineffective in very shallow waters, where aerial photography/multispectral sensors and LIDAR become most effective. All remote sensing methods generally required some level of direct ground-truthing at finer scales to assign identities to different the different classes mapped (e.g., by visual inspection on foot over low tides or drop cameras).

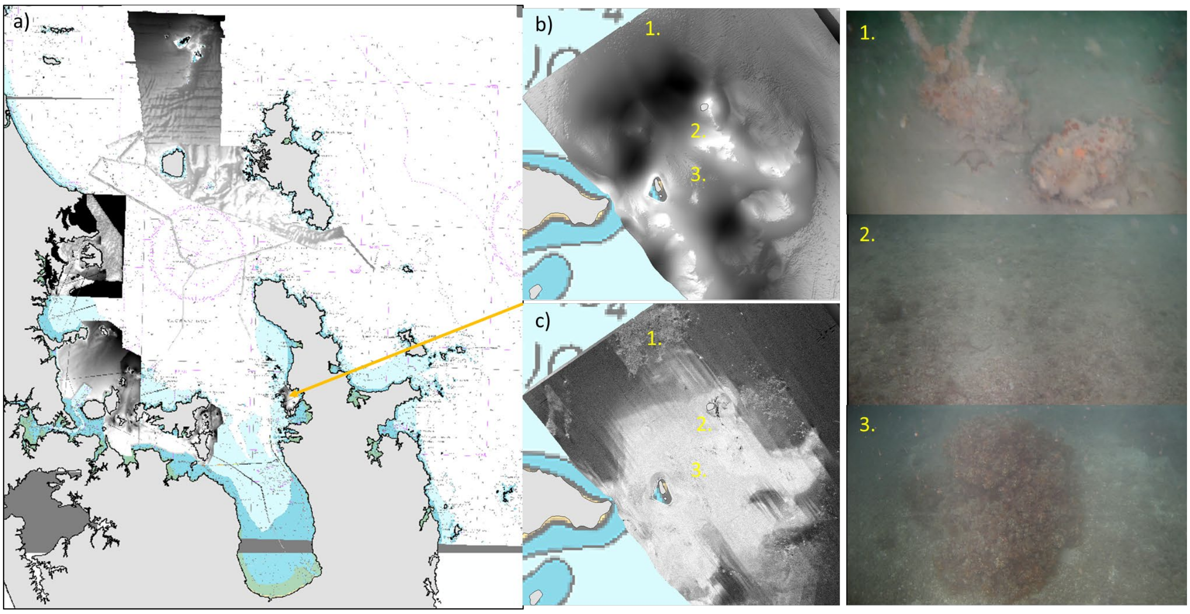


Figure 19: a) Multibeam sonar mapping in the Hauraki Gulf, as grey-scale bathymetry (note that different blocks have different intensity ramps, to avoid whiteout of shallower areas). The line features are single pass (shallow water) or double pass (deep water) transects, with some shallow water ones not visible at the spatial resolution shown. Not displayed is a low-resolution large-scale swathe through the mid/central Gulf by the New Zealand Navy (the sonar unit used trades off spatial extent for finer-scale seabed resolution); nor various small blocks mapped by private mapping companies, and the University of Waikato (data not publicly available). Figures b and c show a recently multibeam sonar mapped area off the west Coromandel as bathymetry (top), darker areas are deeper, maximum depth around 37 metres, and backscatter (bottom), a measure of seafloor acoustic reflectivity/hardness (white is shell-dominated area, dark grey is muds). Different biogenic habitats found in this area include horse mussel beds (1.), shell gravel with low densities of epifauna (2., several scallops), and *Galeolaria* tubeworm mounds (3.) (images vary in scale). Horse mussel beds appear in the backscatter data (c) as hardness mosaics (see position of 1.). Sources: multibeam data, Omaha Bay across to Ponui Channel (LINZ); Tarakihi Island to south-east Ponui Island, West Coromandel, and Port Fitzroy, Great Barrier Island (Foundation North GIFT Fund, AC, WRC, NIWA SIFF Fund); inshore transects (Ministry of Business, Innovation and Employment (MBIE) Juvenile fish habitat bottlenecks programme C01X1618); Mokohinau Islands and Barrier islands (RV *Tangaroa* Fund, NIWA).

Mapping can be laborious and expensive, and with the vagaries of economy and government priorities, may take a long time to acquire for large areas. One approach advanced by Townsend et al. (2014) for broad-scale mapping of ecological services was "*defining services from a series of principles based on current ecological understanding and linking these to marine biophysical parameters*" to develop ecosystem service maps. Using the Hauraki Gulf as an example, they used modelling techniques to predict and map putative biogenic habitat, nutrient recycling, and ecosystem productivity (Figure 20). These model predictions were partially verified using data from several well studied locations.

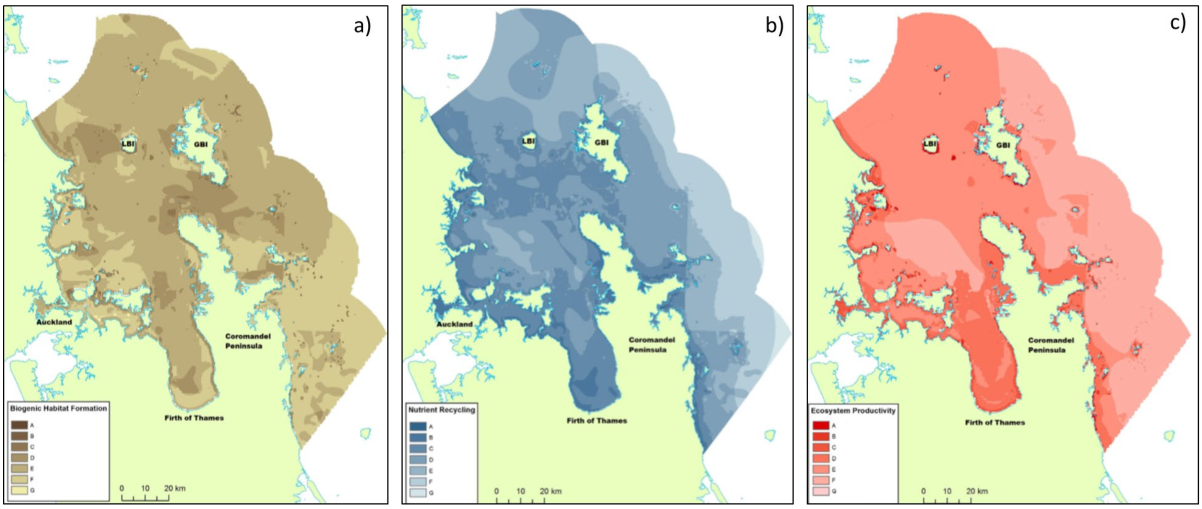


Figure 20: **Service maps produced using the ecosystem principles approach, illustrating biogenic habitat provision (a), nutrient recycling (b) and ecosystem productivity (c) for the Hauraki Gulf, New Zealand. Ranked scores indicate relative importance for service provision from high (darkest shading) to low (lightest shading). Locations are shown for orientation, including Little Barrier Island (LBI) and Great Barrier Island (GBI). Source: figure 3 of Townsend et al. (2014).**

Townsend & Lohrer (2019) further explored this approach, targeting biogenic structure across 56 sites west of Great Barrier Island, in water depths of less than 50 metres. Using a biogenic ranking (1 through 5) they found high agreement between direct empirical seafloor observations, and the model predictions; in areas predicted by the approach to have the highest levels of biogenic habitat complexity, the habitat was typified by complex rocky reef communities and macroalgal forests. However, it should be noted that the main input data for the model prediction were bathymetry and sediment type, which by their very nature are highly likely to clearly identify rocky reef habitats, which in clear shallow waters are usually dominated by macroalgal forests (Anderson et al. 2019).

Another approach, to assist in current data mapping efforts where decisions need to be made about where to place effort within larger areas, as well as learning more about what has been historically lost, is the use of Local Ecological Knowledge (LEK). This is the first-hand experience of people who have had a long-term direct association with the sea (e.g., fishers, divers, food-gatherers), that might extend back generations. For indigenous peoples, such knowledge is sometimes referred to internationally as Traditional Ecological Knowledge (TEK), and as components of Mātauranga Māori in New Zealand/Aotearoa.

Some national scale, continental shelf LEK work was done with retired trawlermen and other commercial fishers (Jones et al. 2016), who fished the mid to outer Hauraki Gulf. Sampling intensity (number of interviews) was low, but nonetheless identified areas of coral (likely to include and/or be bryozoans), horse mussels, dog cockles, tubeworms, sponges, and kelp/macroalgae (Figure 21, Table 6). Where science information was available, there was general agreement between these and LEK (Jones et al. 2016). More recent LEK work for the shallow water environment of the Hauraki Gulf, from recreational fisher and diver interviews, are currently being written up in the MBIE Juvenile fish habitat bottlenecks programme (Handley et al. unpubl. data).

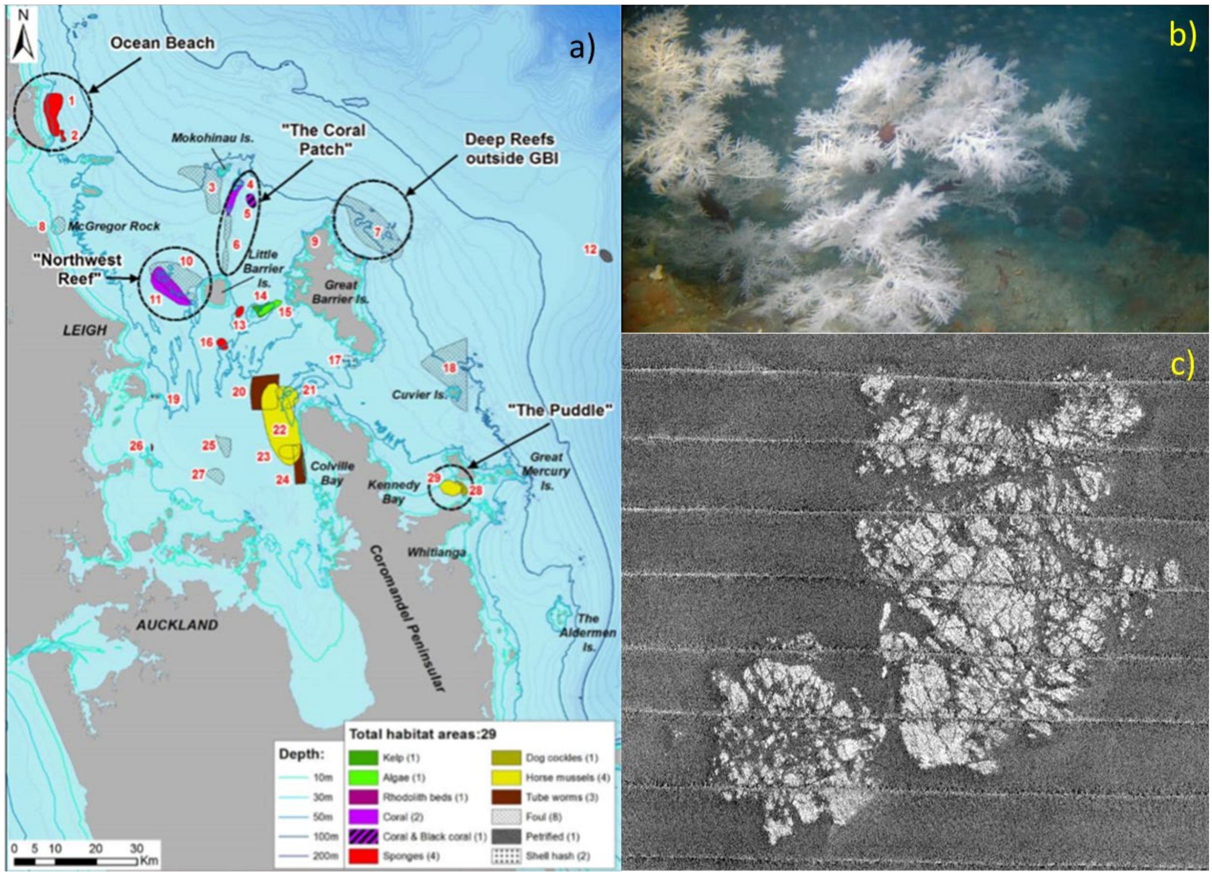


Figure 21: a) LEK identified habitat areas in the Hauraki Gulf. Each fisher-drawn area has a unique number (red). Key sites are circled and labelled as black text on white background. There was no fisher knowledge of the inner Hauraki Gulf, where trawling has been banned since the 1930s. b) black coral tree with ophiuroids (brittle stars) on the actual 'Coral Patch' reef (southern end of site 6). c) 'Coral Patch' reef (several hundred metres across) as mapped in backscatter by multibeam sonar (circa 70 m water depth). Source: a) figure 5 of Jones et al. (2016), b) NIWA 2011 multibeam mapping; c) towed video camera imagery, MBIE Juvenile fish habitat bottlenecks (CO1X1618, 2020).

Table 6: Summary table of sites described by fishers in the Hauraki Gulf region with the area identification numbers, brief description, fishing impacts where mentioned, and the number of fishers who described verbally or identified overlapping or very close areas. Key sites in bold. 'Foul' refers to seafloor too rough to trawl (usually reef). (Continued on next page)

Sites	Area ID No.	Description	Frequency of ID
Ocean Beach	1, 2	Rugged terrain with perceived high fish abundance, that was avoided due to the bycatch of "*pumpkin and cauliflower sponges*".	2
"***The Coral Patch***" (south of Mokohinau Islands, Simpson Rock and north of Little Barrier Island)	3, 4, 5, 6	A narrow strip from south of the Mokohinau Islands, to north of Little Barrier, including around the pinnacle "*Simpson Rock*" (4 and 6); a series of mounds sitting 6 m above surrounding seabed. High snapper catches, presence of small fish noted, and a bycatch of coral. Patches of foul, coral, and black coral also reported either side of this strip (3, 5).	3
Deep reefs, **Great Barrier** and Cuvier Island	7, 18	Large areas of foul northeast of Great Barrier Island and north of Cuvier Island in 100 m + of water. Some clear tows closer inshore targeting snapper, hāpuku, gemfish, and bluenose. No bycatch described. G.B.I. foul known of by 2 fishers but not marked.	GBI (1), Cuvier (3)

Sites	Area ID No.	Description	Frequency of ID
The "*Petrified Forest*" and other deep-water environments off G.B.I.	12	Located about 60 miles east of Great Barrier Island, in 400 m depth a "*Petrified Forest*" with shell and rock embedded together in unusual formations was described. The site was targeted for hāpuku. "*Pinnacles*" in depths of 250–300 m where "*black spikey corals and orange sponges*" were snagged were also mentioned in this area, and patches of "*slimies*" (pink-coloured sea pens), "*slimey*" soft corals and sponges on muddy grounds also mentioned (neither marked on the chart).	2 (Petrified forest)
North-west Reef (west of Little Barrier Island)	10, 11	Described as "*Foul*", "*Reef*" and with corals resembling deep-water stony corals. Targeted mainly for snapper, associated with catches of small fish.	3
Craddock Channel, "*The Pumpkin patch*" and Horn Rock	13, 14, 15, 16	An area of sponge on the edge of the Craddock Channel (13) and a pinnacle further south (16) were both close to trawl tows targeting snapper, with a bycatch of "*pumpkin sponges*" / "*black pumpkin sponges*" reported. On the eastern side, tows close to these areas (14 and 15) targeting spawning snapper could come up clogged with kelp.	4
Inshore Reefs	8, 25, 17	McGregor's Rock off Bream Tail had been heavily fished for snapper but was previously an area where "*sponges and weed*" were caught as a bycatch. Several other patches of "*Foul*" were located by a second fisher in under 50 m depth.	1
North-west coast of Coromandel Peninsula	20,21,22, 23,24	Horse mussel beds and tubeworm patches along the Coromandel coast from Colville Bay north. Described as "*workable*" when inshore trawling was permitted, being targeted for snapper, but had been "*fished down*". A current fisher mentioned only tube worms in two distinct patches to the north and south of the horse mussel area.	5
"***The Puddle***", Mercury Islands	28, 29	Overlapping areas, one described as horse mussels with undersized snapper, the other as dog cockles with sponges growing on them. Kennedy Bay and the Aldermen Islands were also mentioned as fish nursery grounds, but not marked.	2

4.2 Habitats

A range of habitats are discussed in the following sections. A summary of their past and present distributions is given in Section 4.14, along with a summary table of relative ecosystem goods and services (from Geange et al. 2019).

4.2.1 Salt marsh

Salt marsh refers to a mixture of salt tolerant plants which occur in estuaries and coastal areas, at or above the high tide mark (e.g., Figure 22). Across New Zealand, around 74 vascular plant species contribute to salt marsh, including seven endemic species (sea sedge *Carex litorosa*, *Lachnagrostis littoralis*, Batchelors buttons/shore cotula *Leptinella dioica*, oioi/or jointed wire rush *Apodasmia similis*, salt marsh ribbon wood *Plagianthus divaricatus*, silver tussock *Poa cita*, and Walkers salt grass *Puccinellia walker*) (Haacks & Thannheiser 2003). Species richness increased from north to south, with mangrove forest limiting the area available for salt marsh in more northern areas, along with largely monospecific stands of the species poor habitats of *Juncus maritimus* (sea rush) and *A. similis*. Around 49% of the species present are introduced, including the cord grass species *Spartina angelica* and *Spartina alterniflora*, introduced for foreshore protection and estuarine reclamation, but now viewed as noxious species. Figure 23 shows an idealised salt marsh plant species profile. Haacks & Thannheiser (2003) identified four salt marsh vegetation complexes in the North Island, and three in the South Island; the reader is directed to that paper for in-depth species lists and descriptions.

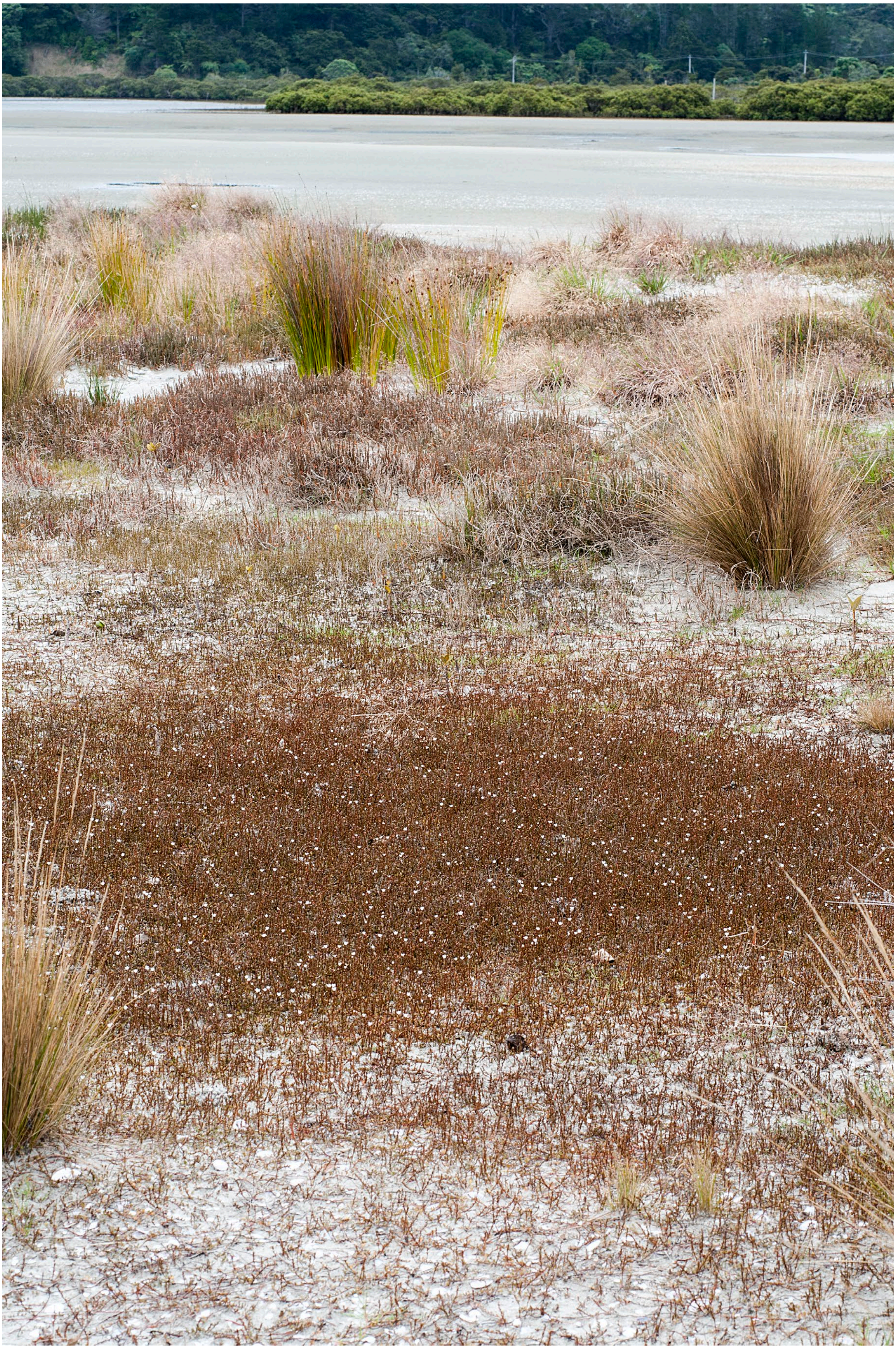


Figure 22: Salt marsh assemblage at Horseshoe Island, Whangateau Estuary, north-western Hauraki Gulf (Source: Richard Taylor, Institute of Marine Science, University of Auckland).

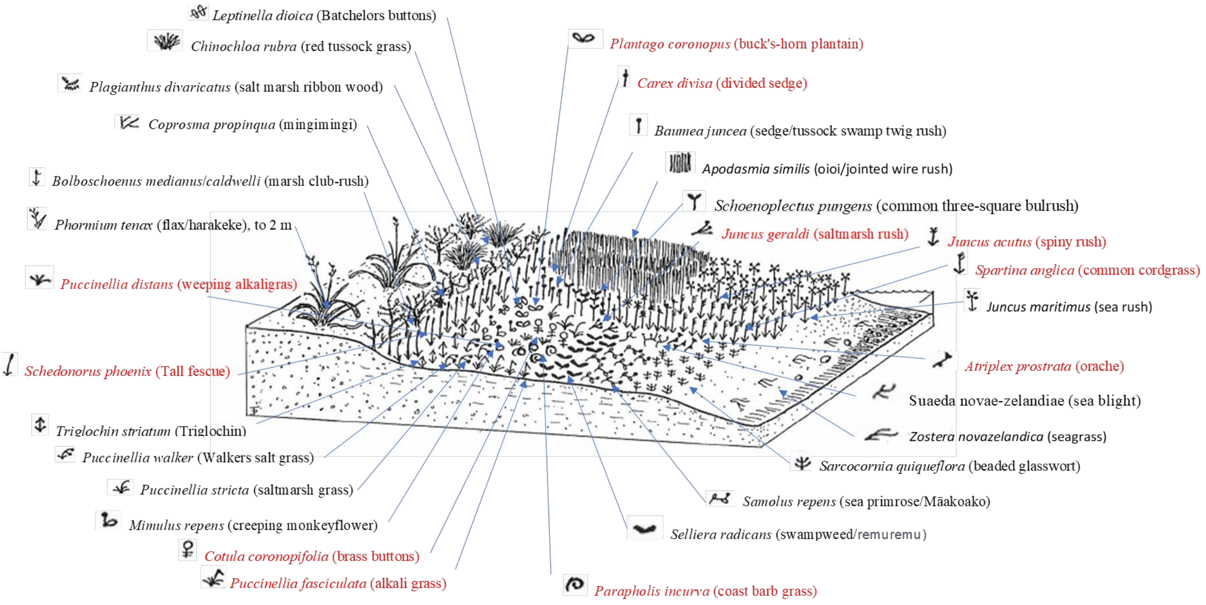


Figure 23: Idealised profile of species occurrence in salt marshes of New Zealand, including introduced species (shown as red text). Source: modified with permission from figure 3 of Haacks & Thannheiser (2003), www.borntraeger-cramer.de/journals/phyto.

Historical distribution

No estimates of salt marsh loss for the full Hauraki Gulf could be found in the literature. For the adjacent Bay of Plenty, Thomsen et al. (2009) used data from Park (2000) and the New Zealand Land Cover Base Map to estimate a loss of 56%. This was noted as being less than that of intensively altered North American coastal areas (e.g., 80% of New England marshes). Thomsen et al. (2009) also noted that the New Zealand wide loss of salt marsh was estimated at more than 90% by Marsden & Heremaia (1998); but that the 1998 study provided no supporting data or references. Lundquist et al. (2020) used the Ministry for the Environment's website (https://data.mfe.govt.nz) data on 'swamp' and 'marsh' wetland to estimate that, within the Waikato region (including part of the west coast), wetland habitats declined 93%, from 356 516 to 28 226 ha, between pre-human times and the present day. This estimate may be inflated through the unavoidable inclusion of some terrestrial (non-salt marsh) wetland habitat.

Loss of this habitat in the western and inner Hauraki Gulf has probably been even more extensive and widespread than for the Waikato region. This will have included reclamation for the city of Auckland (e.g., Figure 24) and, more broadly, for live-stock farming.

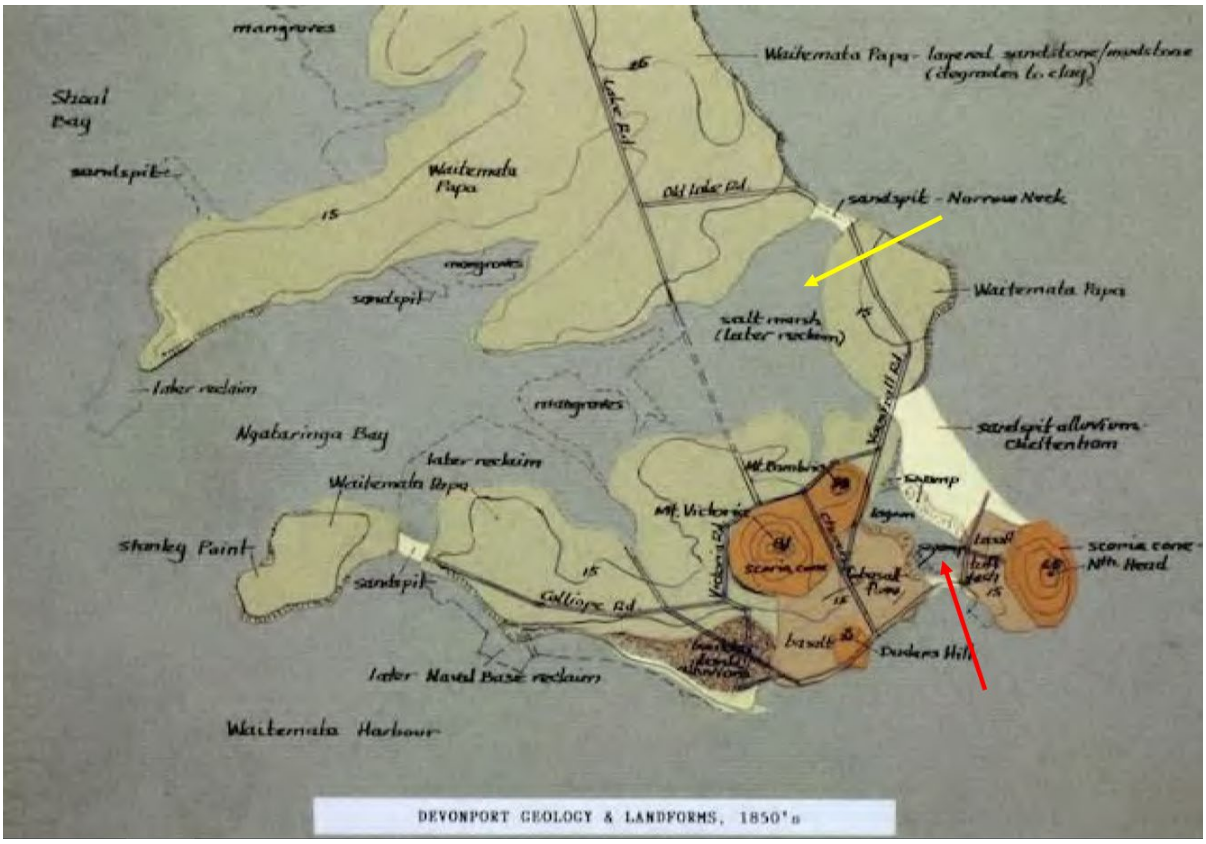


Figure 24: Devonport peninsula in the 1850s. A large salt marsh area was subsequently reclaimed (position shown by yellow arrow), which today holds the maim road, a reserve, and a golf course. Other reclamations include a lagoon (position shown by red arrow) adjacent to North Head (now sports grounds), area along the south Waitematā Harbour for a naval base, and mangrove areas for more naval grounds and housing. North Head was once covered by pohutukawa that grew down to the shore, and Ngataringa Bay and Stanley Point hold evidence of past mixed forest of karaka, mahoe, toitoi, clematis, and kauri. Source: figure 5 of Auckland Council (2011).

Present distribution

Services and value of habitat

The goods and services that salt marshes provide are not well known for the Hauraki Gulf/New Zealand, with little/no data on animals utilising salt marshes (Thomsen et al. 2009). Fish usage is probably constrained to short periods of foraging during spring tides when they are accessible. These are probably semi-pelagic species, such as grey (*Mugil cephalus*) and yellow-eyed (*Aldrichetta forsteri*) mullets, smelt (*Retropinna retropinna*), and the common galaxiid *Galaxias maculatus* (inanga) (Morrison et al. 2014). *Galaxias maculatus* is the only galaxiid species which spawns in estuarine systems (C. Baker, NIWA, pers. comm.), and, as with other galaxiids, it deposits its eggs among riparian vegetation and other substrates supra-tidally (at extreme upper tidal elevation) (McDowall & Charteris 2006). Hickford et al. (2010) sampled sites around Banks Peninsula and recorded *M. maculatus* eggs on eight plant species which prefer damp conditions; however, most of those plants were not salt marsh species. There was a strong positive correlation between the density of riparian vegetation and initial egg densities from spawning. They also found higher egg survival in riparian vegetation with dense stems and a thick aerial root-mat (*Juncus edgariae*, *Schedonorus phoenix,* and *Holcus lanatus*) which provided a cooler and more humid micro-environment, as well as probably protection to the eggs from lethal ultra-violet light (Hickford et al. 2010). *Galaxias maculatus* juveniles contribute 90% of the New Zealand whitebait catch (four other species contribute the other 10%) (McDowall 1965), and there is a general consensus that catches have declined since the late 1950s, although little direct statistical evidence is available to attribute this to overfishing (Hickford et al. 2010). The destruction of spawning and rearing habitat is seen as a contributing factor (Taylor 1996). A significant negative relationship between the occurrence and abundance of adult inanga *G. maculata* (the common galaxiid) and increasing catchment development has been reported from a national wide estuarine fish survey (69 estuaries, including in the Hauraki Gulf, Francis et al. 2011). Presumably, loss of salt marsh and other semi-aquatic brackish water vegetation in the gulf region has substantially reduced inanga adult populations (and the associated highly valued whitebait catch of juveniles returning from the sea as part of their catadromous life history).

Recovery and restoration potential

The greatest threat for salt marsh habitats is from drainage and reclamation, followed by livestock grazing, and trampling. Sea-level rise is a new threat. Thomsen et al. (2009) felt that one potentially positive aspect of rising sea levels, which they quoted as being an expected 0.14 to 0.18 metre by 2050 and 0.31 to 0.49 metre by 2100, was that increased land salinisation around estuaries will transform terrestrial land into salt meadows (including land reclaimed in the past), and eventually estuary mudflats (Bell et al. 2001). Seawalls, coastal roads, and other human infrastructure will limit such advances in some areas of the Hauraki Gulf, through ‘coastal squeeze’.

Some habitat restoration projects have been undertaken by councils and local trusts (Bergin 1994, Auckland Regional Council 2000a, 2000b), with seeds of the most common salt marsh species available from commercial nurseries (e.g., sea rush *Juncus maritimus* and oioi/jointed rush reed *Leptocarpus* (*Apodasmia*) *similis*, New Zealand Tree Seeds), local voluntary groups/trusts (e.g., Guardians of Pāuatahanui Inlet), and council nurseries (Thomsen et al. 2005). Transplantation success of sea rush and oioi has been shown to depend on initial transplant size (increases with size), but not with transplant spacing or nutrient additions (Bergin 1994). Further work by Thomsen et al. (2005) also confirmed these two dominant species as suitable for restoration work; however, there was no success with the common three-square bulrush *Schoenoplectus pungens*. They also found that for sea rush and oioi, survival and growth was not affected by soil type (dredged estuarine soils vs. marsh soils), but that plants from natural populations generated more biomass than plants sourced from a local nursery. This was attributed to nursery plants being ‘softer’ than wild stock, i.e., less resistant to wind, currents, and waves; and/or more exposed to rabbit grazing.

Knowledge gaps

- The use of salt marsh by whitebait (juveniles of *Galaxiid* species) in the Hauraki Gulf region would provide useful knowledge when quantifying the value of these habitats.

4.2.2 Mangroves (mānawa)

New Zealand has only one species of mangrove, *Avicennia marina australasica* (Figure 25). In strong contrast to most other countries, New Zealand’s mangrove forests are expanding at a significant rate, replacing other habitats such as open sand and mud flats, causing strong and often acrimonious societal debate as to their value and what management actions are appropriate. There is strong anecdotal evidence that significant mangrove loss has occurred historically at some locations, largely to create farmland, but true losses are unknown (Morrisey et al. 2010). From 1977, mangroves have been fully nationally protected from infilling (e.g., to create farmland), and clearance of mangroves has been a controlled activity (i.e., consents are required). Overall, mangroves can be viewed as one of the very few (if not the only) marine biogenic habitats that is expanding in spatial extent in New Zealand. A comprehensive review of temperate mangroves in New Zealand is provided by Morrisey et al. (2007), and at the global scale by Morrisey et al. (2010). Subsequent to those publications: Morrison et al. (2014) reviewed their relative role in helping underpin New Zealand fisheries productivity; Horstman et al. (2018) explored the dynamics of New Zealand’s expanding mangrove forests; Dencer-Brown et al. (2019) assessed perceptions and attitudes towards mangrove and their removal in New Zealand; Anderson et al. (2019) completed a ‘stock-take/inventory of mangroves at the national scale’; and

Dencer-Brown et al. (2020) completed an 'integrated biodiversity assessment of urban mangroves', focused on forests in the Manukau Harbour, Auckland.

Figure 25: Australasian bittern/matuku (*Botaurus poiciloptilus*) foraging on the edge of mangrove fringe, Pakiri Estuary, north-western Hauraki Gulf. Abundant when Europeans arrived, less than 1000 remain in New Zealand. Also found in Australia and New Caledonia, where populations have also declined dramatically. They are now classed globally as endangered. Source: Richard Taylor, Institute of Marine Science, Auckland University.

Historical distribution

Following earlier historical losses from land reclamation and livestock grazing, mangroves have been increasing in spatial extent across many northern New Zealand estuaries over the last 50 to 70 years (Burns & Ogden 1985, Young & Harvey 1996, Lovelock et al. 2007, Swales et al. 2007, 2015, 2016; Harty 2009, Morrisey et al. 2007, 2010; Horstman et al. 2018). For the Hauraki Gulf, this has included most of the estuaries on the Coromandel Peninsula (Jones 2008, Morrison et al. 2009). Of note is the large (>1 km wide) mangrove forest that covers the upper Firth of Thames, which was not present 50 years ago (Lovelock et al. 2010, Swales et al. 2007, 2016). This area was once an extensive and long shelly beach along which horse races were run.

Present distribution

Analysis of aerial imagery across multiple decades for some Auckland estuaries indicates expansion rates of up to 20% per year (average of 4.1%), since the late 1940s (Swales et al. 2009). Further analysis across all Auckland region estuaries suggests a rate of 3.4% per year between 1940 and 2014, and, since 1974, differential expansion rates for dwarf (23.2% p.a.) and tall mangroves (1.2%) (Suyadi et al. 2019) (Figure 26). Modelling found that catchment forest cover, sea level, and mean annual air temperature explained about 85% of the variation in mangrove area, and that gains in mangrove forest area were correlated with periods of catchment forest clearance.

A range of estuaries have been mapped using aerial imagery in the Auckland region, the largest and most standardised set being the 38 estuaries (both east and west coasts) covered by Suyadi et al. (2019) (Figure 26). More broadly, DOC has compiled a national-scale inventory of mangrove extent in a SeaSketch project, which includes 103 mangrove datasets. This inventory, published in November 2018, includes temporal changes in extent and/or density of mangroves for most estuaries, but is limited by the same metadata issues of past comparisons (e.g., Land Cover Data Base (LCDB) areal polygons without metadata, or missing the year of data collection). In November 2018, the SeaSketch database represented 215 km^2 of mangroves in the 'most recent' layers (including those with incomplete metadata). Metadata required for each survey includes the date of survey, name of the estuarine system (e.g., Tauranga Harbour), density, area (ha), Intellectual Property, and a bibliographic reference with a link to a copy of the report. If density data are included, they are represented visually by three classes (low <20%, medium 20–50%, or high >50%, or simply as 'present').

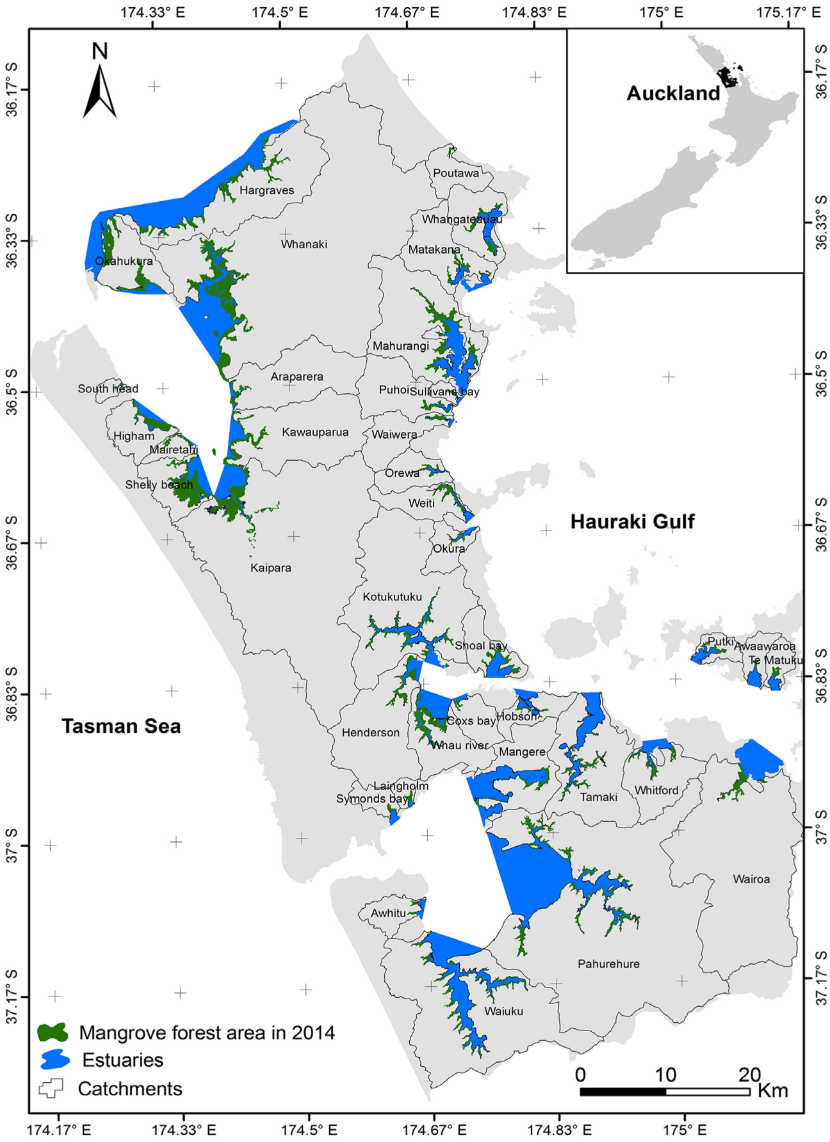


Figure 26: Summary map of coverage of mangrove mapping in the Auckland region as completed by Suyadi et al. (2019). Source: figure 1 of Suyadi et al. (2019). Maps are also available for mangrove distributions in the Waikato region.

Services and value of habitat

The goods and services that New Zealand's temperate mangroves provide are not well quantified but are thought to be moderate compared to tropical mangroves (Morrisey et al. 2007, 2010). Infaunal seafloor assemblages are neither diverse nor abundant (Morrisey et al. 2007, Lowe 2013), with one study suggesting that as forests grow older, the associated infaunal richness and abundance of some species possibly reduces; with a potential shift in faunal diversity from the benthos to the mangrove trucks and canopy (Morrisey et al. 2003). Recent work on mangrove forests in the Manukau Harbour (west coast of Auckland) using a beat sheet and sweep net recorded 49 species of arboreal arthropods from 2699 individuals (Dencer-Brown et al. 2020). This included 15 spider (order Araneae) and 12 beetle species (order Coleoptera), with the most abundant species being the ant *Technomyrmex jocosus*, then the common woodlouse *Oniscus asellus* (an isopod). None of the species found were unique to mangrove ecosystems; and the authors noted that the two known endemic mangrove insect species of the mangrove leafroller *Planotortrix avicenniae* and the eriophyid mite *Aceria avicenniae* were not encountered in their study. Dencer-Brown et al. (2020) also deployed a range of other methods to sample the fauna at their mangrove sites, including inkpads, housing units, Holden traps, acoustic bat detectors, timed bird counts, trail cameras, and underwater Go-Pro cameras. They recorded 15 bird species, of which five were native (sacred kingfisher, welcome swallow, fantail, pukeko, banded rail), and six mammal species (including two passes of a presumed long-tailed bat, and the introduced species of the Norway rat (*Rattus norvegicus*), mice (*Mus musculus*), a possum, a hedgehog, and a domestic cat). Of the 619 birds observed (not all necessarily unique individuals, the green finch (*Chloris chloris*, introduced species) was the most abundant, at 324 observations. Banded rail (*Gallirallus philippensis*), a threatened endemic bird species, were detected by trail cameras well inside the mangrove forests. No Australasian bittern (*Botaurus poiciloptilus*) were seen (see Figure 25).

No skinks or geckos were found in the mangroves. Although skinks (species not given) were observed at all three sites in the landward salt marsh, the authors noted that the adjacent industrial and urban land use, and associated human disturbance, made the presence of reptiles unlikely. These new data/observations match the review findings of Morrisey et al. (2007) and add new information on the insect fauna of New Zealand mangroves. Possible use of mangroves by reptiles remains very poorly known.

For fish, of note was the observation of "*a permanent channel was found at Waimahia which contained a resident short-finned eel population of approximately five eels per metre squared for a length of 20 m (abundance estimated at low tide)*" (Dencer-Brown et al. 2020). Such low tide refuges for adult short-finned eels have also been observed in Hauraki Gulf estuarine mangrove forests, where eels occur in high densities lying in very shallow channels at low tides, sometimes with their backs breaking the water surface. Examples include both mangrove channels in the Te Makutu Marine Reserve (Waiheke Island) and at Shoal Bay (Waitematā Harbour) (M. Morrison, NIWA, pers. obs.). This may represent a behaviour only seen in places where fishing pressure is low/absent, and adult short-finned eels can occur at higher local densities.

In general, New Zealand's mangrove forests are not particularly important as fish habitat (Morrisey et al. 2007, 2010, Morrison et al. 2014a). Fyke net sampling has shown them to provide nursery habitats for short-finned eels, parore (east coast only), and grey mullet (west coast). Although other species such as juvenile yellow-eyed mullet can occur in, or be found adjacent to them, in large numbers, these also occur in similar densities across a range of other estuarine and coastal habitats, including estuaries that do not contain mangrove forests. No fish species are obligate habitat users of mangrove forests in their life histories, being found across other habitats also (Morrison et al. 2014a).

A recent literature review of the use by threatened or at-risk birds of mangrove habitat by Bell & Blayney (2017) identified a range of species (Table 7), though noted that many information gaps remained, and the quality of the supporting data was variable. For each of these bird species, they provided a summary of degree of dependence (on mangroves), usage (e.g., foraging, breeding, and roosting), important characteristics (of mangrove architecture), seasonal use, and connectivity and habitat patch size (with other habitats). Only banded rail (moho pererū, *Gallirallus philippensis*) was rated as dependent on mangrove habitat for its continued survival (Beauchamp 2012, Baird 2015), including for providing cover from aerial predators while foraging close to their preferred salt marsh roosting and breeding habitats (Botha 2011, Beauchamp 2012, Bellingham 2013). Caspian tern (taranui, *Hydroprogne caspia*) and North Island fernbird (kōtātā/mātātā, *Poodytes punctatus*) were thought to have a moderate dependency on mangroves, with the other threatened/at risk bird species having low or no dependence on mangroves.

Bell & Blayney (2017) more broadly noted that as many as 48 different bird species (native and introduced) have been observed using mangrove habitat (Crisp et al. 1990, Morrisey et al. 2007, 2010, O'Donnell 2011). Species breeding in mangroves include grey warbler (riroriro, *Gerygone igata*), silvereye (tauhou, *Zosterops lateralis*), fantail (pīwakawaka / tīwakawaka / piwaiwaka *Rhipidura fuliginosa*), house sparrow (*Passer domesticus*), shining cuckoo (pīpīwharauroa, *Chrysococcyx lucidus*), and New Zealand kingfisher (kōtare, *Halycon sancta vagans*) (Cox 1977, Morrisey et al. 2007, 2010). Species observed roosting within mangroves include little black shag (kawau tūi, *Phalacrocorax sulcirostris*), pied shag (kāruhiruhi, *Phalacrocorax varius*), white-faced heron (matuku, *Egretta novaehollandiae*), royal spoonbill (kōtuku ngutupapa, *Platalea regia*), starling (*Sturnus vulgaris*), house sparrow, and chaffinch (pahirini, *Fringilla coelabs*) (Beauchamp & Parish 1999).

Table 7: Threatened or at-risk species of birds that use mangroves. 1, use mangroves or habitat fringing mangroves for foraging; 2, observed roosting within or directly adjacent to mangroves; 3, observed using mangroves as refuge. Source: modified from figure 1 of Bell & Blayney (2017).

Threat Classification (Robertson et al. 2017)		Common name	Scientific name
Threatened	Nationally critical	New Zealand fairy tern[1]	*Sterna nereis davisae*
		White heron[1]	*Ardea modesta*
	Nationally endangered	Australasian bittern[1]	*Botaurus poiciloptilus*
	Nationally vulnerable	Lesser knot	*Calidris canutus rogersi*
		Caspian tern	*Hydroprogne caspia*
At risk		Pied shag[1,2]	*Phalacrocorax varius*
		Banded rail[1]	*Gallirallus philippensis assimilis*
		New Zealand pied oystercatcher[1,3]	*Haematopus finschi*
		Pied stilt[1,2]	*Himantopus leucocephalus*
		Eastern bar-tailed godwit	*Limosa lapponica baueri*
		North Island fernbird	*Poodytes punctatus*

No summary insect, bird, or fish species/assemblage metrics or rankings exist that would allow for numerical values to be assigned to different mangrove forest settings, architecture, or regional locations in New Zealand.

In terms of carbon storage in mangroves, there is increasing interest in how carbon is sequestered and stored in different marine habitats, their relative contributions to global carbon budgets and climate change, and as protection against coastal erosion and storm surges. Recent work in the Tairua Estuary (eastern Coromandel Peninsula) on blue carbon stocks measured the contributions from a range of habitats, including above ground vegetation, and down to 100 cm below the sediment surface. Values varied significantly across habitats, from saltmarsh (90 $t\ ha^{-1}$) to mangrove (46 $t\ ha^{-1}$), seagrass (27 $t\ ha^{-1}$), and to unvegetated habitats (26 $t\ ha^{-1}$). Using δ13C and δ15N mixing models, 'cross-habitat subsidies' (detritus) were also identified as flowing on to unvegetated sediments, contributing an estimated 41% of the organic carbon within unvegetated sediments (Bulmer et al. 2020). Mangroves also provide protection against coastal erosion, and storm flooding.

Recovery and restoration potential

Work assessing the impacts of clearing mangrove areas has found that the associated mudflats do not easily revert to sandy substrate, especially in sheltered areas or areas with high sediment inputs. For example, three years on from large scale mangrove removals in Whangamata Harbour, in the cleared area the sediment properties and benthic community assemblage were closer to those of nearby uncleared mangroves, rather than nearby sandy areas (Horstman et al. 2018).

Although mangrove restoration has become a focus for several species in tropical regions, in New Zealand no work has been done, because mangrove forests have significantly expanded since the arrival of Europeans. Rather, there has been often acrimonious societal debate on their relative habitat value, and what management actions are most appropriate, focused on mangrove control around both removals, and limiting further expansions (Morrisey et al. 2007, Horstman et al. 2018, Dencer-Brown et al. 2019). There is unlikely to be any demand for mangrove restoration in the Hauraki Gulf, given the complete reclamation of some coastal fringes that probably once held mangroves but which are now heavily used high-value urban/industrial areas (e.g., Auckland city on the Waitematā Harbour), and, perhaps more importantly, that there has been extensive mangrove forest expansion into areas where they were historically absent.

Knowledge gaps

None of direct relevance.

4.2.3 Seagrass

New Zealand has only one species of seagrass (*Zostera muelleri*), which is indigenous to New Zealand and southern parts of Australia (previous taxonomic names have included *Z. novazelandica* and *Z. capricorni*). The conservation status of this species in New Zealand is At Risk – declining (de Lange et al. 2017). Inglis (2003) collated and summarised data on seagrass extent in New Zealand and concluded that there had been substantial declines in many of New Zealand harbours and estuaries. Matheson et al. (2011) also concluded that there has been substantial seagrass habitat loss, particularly for subtidal seagrass habitat, within the last 50–80 years. This matches global declines in many seagrass species, which are viewed as a plant group in global crisis (Orth et al. 2006, Waycott et al. 2009). Turner & Schwarz (2006) provide a comprehensive review of what is known about seagrass in New Zealand; subsequent to that review, a range of further research has been completed, including its value in supporting fisheries species (Morrison et al. 2014a–c) and a national-scale inventory of its occurrence (Anderson et al. 2019).

Although present as intertidal patches or small meadows in many estuaries, as well as on some rock platforms and off some sheltered coastal islands, extensive healthy and extensive meadows in the present day tend to be in remoter areas away from high density human populations (e.g., Pārengarenga, Rangaunu, and Kaipara harbours, Northland; Farewell Spit and Whanganui Inlet, top of South Island; Bluff/Awarua Harbour, Southland).

Historical distribution

Many regions have had their stocks of seagrass distribution and abundance substantially reduced by human activities on the land, largely through those that have generated large scale increases in sediment (and to a lesser extent nutrient) inputs into estuarine and coastal receiving waters (Morrison et al. 2009). Reduced water clarity and increased seafloor muddiness, along with other stressors, have adversely affected the ability of seagrasses to grow and maintain populations through time. Examples include large and fundamental historical seagrass losses from Whangarei, Waitematā, Tauranga, and Avon-Heathcote estuaries, as well as many other smaller estuaries. Of these, only Whangarei has shown any significant recovery, with large scale return of seagrass in the harbour starting around 2013, although it is not yet back to historical levels. However, loss of seagrass in that harbour was a result of five million metric tons of fine sediments being released in the harbour from port development works and a large cement factory (for the latter, ‘fines’ less than 10 microns were dumped into the harbour adjacent to the factory) in the 1960s (Morrison 2003). That is quite different from the more diffuse land-use inputs to other harbours around New Zealand. Fifty years passed before the seagrass started to return, for reasons that are still unclear—although improving environmental conditions are probably the most likely driver.

For all seagrass (intertidal and subtidal) in the Waikato region, Lundquist et al. (2020) suggest that less than 20% of the original seagrass extent remains.

Present distribution

The Hauraki Gulf almost certainty also had its historically widespread subtidal seagrass meadows (e.g., Powell 1936) supporting high densities of juvenile fish (including snapper and trevally). Today, very little subtidal seagrass can be found in the gulf. Very limited small subtidal fringes (metres to 10s metres scale) or discrete patches can still be seen occasionally (e.g., Figure 27a), though their collective area is very small. They also seem to come and go, never expanding beyond their initial ‘foothold’. Their presence does suggest that the recovery of larger (and ecologically) meaningful subtidal seagrass meadows is a possibility. In recent years (5–10 years) there has been a general increase in intertidal seagrass across the Auckland region, at multiple locations (Table 8). This appears to be part of some poorly understood longer-time abundance cycles, rather than attributable to locally improving environmental conditions. Locations include Snell’s Beach (Kawau Bay), Little Shoal and Shoal Bay, west Ngataringa Bay, and Cox’s Beach (Waitematā Harbour), Orakei Bay, Kohimarama Bay, Mission Bay (Tamaki Strait), the Tahuna Torea Sandspit (Tamaki Estuary), and the eastern side of Whitford Bay (Tamaki Strait) (Figure 28). Examination of a number of these beds shows a small subtidal fringe to sometimes be present, but of very limited extent, and in very shallow water. Edges extend to a uniform water depth, suggesting that the limiting factor for seaward expansion may be water clarity or turbidity, rather than a lack of colonising plants through lateral rhizome extensions.

Table 8: Observations of subtidal seagrass occurrence in the greater Hauraki Gulf, from the 1990s to the present. (Continued on next page)

Location	Date	Observation	Source
Motuora Island, south of Kawau Bay	Early 1990s	Very sparse small linear bed on western side of Motuora Is north of wharf. Rotenone sampling (T. Willis) returned numbers of gobies but no other fish.	M. Morrison, NIWA, pers. obs.; unpubl. data
Huawai Bay, and Pukapuka Bay, Mahurangi Estuary	Late 1990s	Huawai, very shallow (<15 cm water depth) fringe on edge of intertidal bed, around 15 m wide, numbers of just settled juvenile parore (beach seine, sampled informally, no data collected). Pukapuka, narrow subtidal fringe <1 m wide, < 30 cm water depth, along edge of rectangular intertidal bed on channel edge.	M. Morrison, NIWA, pers. obs.
Whangapoua Estuary, East Coromandel	2003	Subtidal fringe of intertidal meadow and patches at site formally for fish sampled 2003, abundant juvenile fish assemblage especially of snapper, parore, and trevally. Subtidal seagrass gone by 2004, seagrass strongly retracted up shore into intertidal. Historical aerial photography/accounts suggest that in late 1940s most subtidal channels were covered by seagrass meadow/s.	M. Morrison, NIWA, pers. obs.; unpubl. data

Location	Date	Observation	Source
Te Huruhi Bay, Great Mercury Island, Coromandel	2004	Small subtidal meadow on the northern entrance of the bay, going to intertidal to the west. Beach seine tow on subtidal seagrass, from low tide 1 m water depth into the intertidal, recorded density of 45 0+ snapper per 100 m^2.	Schwarz et al. (2006)
Tairua Estuary, East Coromandel	2005	Very narrow (1 m wide) subtidal seagrass fringe on channel edge, to 1 m water depth, about 180 m in length. Sampled for small fish, held high densities of juvenile parore (40 ± 23 s.e., per 100 m^2).	Morrison et al. (2014)
Manaia Harbour, West Coromandel	2012	Large discrete (circa 1000 m^2) subtidal patch in 20–30 cm water depth low tide, holding moderate densities of juvenile parore (10s per 100 m^2; sampled informally (no data available) during development of NIWA’s Iwi Estuarine Monitoring Kit.	M. Morrison, NIWA, pers. obs.
Matheson’s Bay, Leigh	2016	Single circa four m^2 patch (Figure 8), around one metre deep at low tide.	Richard Taylor & Nick Shears, IMS, University of Auckland, pers. comm.
Tiritiri Island	2017	Sparse subtidal seagrass seen north of the wharf, circa 2 m depth, extent unknown.	Jon Stead, NIWA, pers. comm.
St Heliers Bay, Tamaki Strait	Circa 2017.	Very poor-quality sparse cover fringe on intertidal meadow, <10 cm water depth at low tide), 0–2 m width.	M. Morrison, NIWA, pers. obs.
North side of Whangaparoa Peninsula	Circa 2017	Evidence on Google Earth of some shallow water, small low-blade-density subtidal beds in several bays (not verified by site visits).	M. Morrison, NIWA, pers. obs.
End of Point Chevalier, Waitematā	Circa 2018	Small subtidal patches from Point Chevalier Beach and around the point across to Meola Reef.	M. Morrison, NIWA, pers. obs.
Cox’s Bay and towards Auckland Harbour bridge, Waitematā Harbour	Circa 2017	Sparse cover seaward fringe of intertidal bed, out to circa 30 cm water depth low tide, no significant associated fish populations from informal (catch not processed) beach seine sampling.	M. Morrison, NIWA, pers. obs.
Okoromai Bay, south Whangaparoa Peninsula	Circa 2018	Sparse cover 50 m seaward fringe of intertidal bed, to circa 50 cm water depth low tide.	M. Morrison, NIWA, pers. obs.
Clark Bay, Waiheke Island	2019	A limited subtidal fringe (<10 m wide) on edge of a large intertidal meadow.	Fleur Matheson, NIWA, pers. comm.

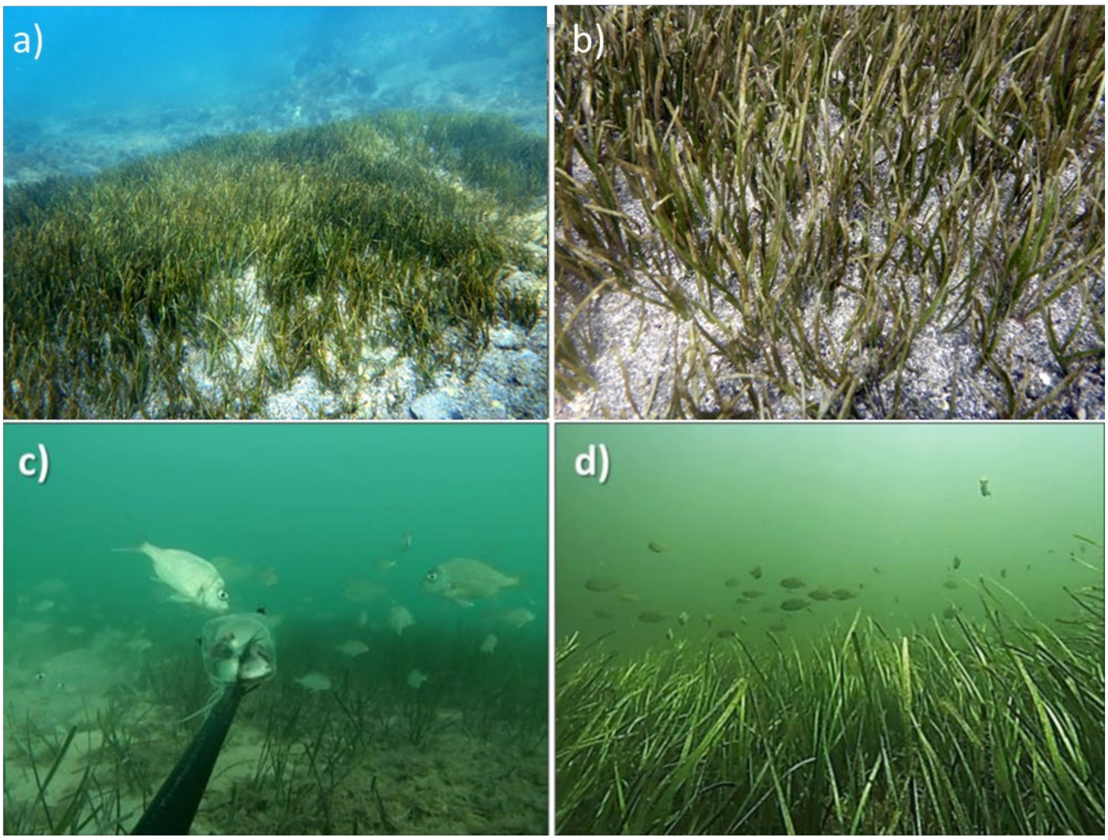


Figure 27: a-b) Circa 4 m^2 patch of subtidal seagrass discovered in 2016 at Matheson's Bay, Leigh, inside the tor, in circa 1 m of water at low tide; c-d) subtidal seagrass meadows supporting juvenile snapper nurseries (eastern Bay of Islands and Rangaunu Harbour, respectively). Source: a-b) Nick Shears, IMS, University of Auckland; c–d, Meredith Lowe, independent.

Figure 28: Examples of re-emerging intertidal seagrass meadows in the Waitematā Harbour, Tamaki Strait, and Whangaparāoa areas (western and inner Hauraki Gulf). Images vary in scale. See Figure 3 for where these locations are in the Gulf. Note the tide or wind generated sediment Source: Google Earth, Maxar Technologies, accessed 21/06/2020; image capture dates Snell's Beach 4/9/2020, Whitford 18/4/2019, all others 29/4/2019.

Services and value of habitat

Seagrass meadows provide a range of ecological goods and services, including:

- High primary productivity contribution to detrital and grazing food webs (Keough & Jenkins 1995, Turner & Schwarz 2004, 2006, Connolly et al. 2005)
- Nutrient recycling (Turner & Schwarz 2006) and providing a significant repository for 'blue carbon' (Matheson & Wadhwa 2012, Bulmer et al. 2020), oxygen release, and denitrification (Turner & Schwarz 2006)
- Attenuating water flow (Eckman 1987, Fonseca & Koehl 2006, Widdows et al. 2008), and trapping and stabilising sediment (Fonseca et al. 1983, Gacia & Duarte 2001)
- Increasing associated invertebrate biodiversity (van Houte-Howes et al. 2004, Alfaro 2006, Schwarz et al. 2006, Mills & Berkenbusch 2009)
- Providing meiofaunal and macrofaunal assemblages refugia from predation, while also providing prey availability for fish (Attrill et al. 2000, Hindell et al. 2000, 2001)
- Acting as critical nursery habitats for many species, including those valued as fisheries species (Orth et al. 2006, Grech et al. 2012, Morrison et al. 2014a–c, Morrison et al. 2019).

Although *Z. muelleri* is largely an intertidal plant, and not well adapted to low light conditions where the water clarity is high and the seafloor suitable, it can extend down into the subtidal zone to several metres below the low tide. Subtidal seagrass may support higher macrofaunal density/biomass/productivity than intertidal meadows in both northern (van Houte Howes et al. 2004, Alfaro 2006, Schwarz et al. 2006) and southern New Zealand (e.g., Mills and Berkenbusch 2009). This may be a result of the large fluctuations in intertidal environmental conditions (periodic desiccation, fluctuating temperatures), resulting in reduced growth (shorter blade lengths), and lower overall diversity and productivity (Schwarz et al. 2006). In contrast, subtidal environmental conditions are more environmentally benign and stable (reduced effects of currents or waves, shelter from predation, higher larval settlement) and are characterised by more complex structure, with higher density and longer stems providing up to 20 times more surface area for epifaunal animals to graze (Schwarz et al. 2006). Subtidal seagrass is also the first component of seagrass meadows to be lost as environmental conditions decline, retracting to the intertidal, and becoming sparser, and eventually disappearing; whereas the intertidal meadows persist and remain relatively healthy (until ultimately even they are lost). Experimental work using artificial seagrass (Whangapoua Harbour, East Coromandel) has shown that subtidal seagrass blade density is an important driver of juvenile and small fish abundances, with the numbers of some fish species increasing with blade density (e.g., juvenile snapper, trevally, triplefins); others were highest at intermediate blade densities (sand and exquisite gobies) and others were neutral to blade density (clingfish). Invertebrates also show very large positive responses to increasing seagrass density, including groups such as shrimps and crabs (Morrison et al. unpubl. data).

Where subtidal seagrass meadows still exist in northern New Zealand, they provide a very high value habitat to a range of juvenile and small fish species (Morrison et al. 2014a–c, Morrison et al 2019). In East Northland, where large subtidal seagrass meadows still exist, these habitats support very high densities of juvenile snapper (<70 mm) (which can exceed average densities of 159 individuals per 100 m^2), as well as juvenile trevally, parore, piper, spotties, leatherjackets, pipefish, and other fish species (Morrison et al. 2014a–c, 2019) as well as distinct invertebrate assemblages (Hayward et al. 2001, Morrison et al. 2014). Quasi-random surveys in 2013 returned juvenile snapper population estimates of 1.081 million fish (CV 17%) for Pārengarenga Harbour and 1.886 million fish (CV 30%) for Rangaunu Harbour, the two East Northland harbours still holding significant subtidal seagrass areas (Morrison et al. 2019). In 2018, nursery habitat surveys in the three East Northland key estuaries still holding extensive subtidal seagrass meadows (Pārengarenga, Rangaunu, Whangarei harbours, the latter holding partially recovered meadows) returned 0+ juvenile snapper densities per km^2 of subtidal seagrass channel edges of 73 000, 107 222, and 17 056 individuals respectively (Morrison et al. 2019). Otolith chemistry work is currently underway using the year 2018 fish, along with those from alternative coastal fish nursery habitats surveyed at the same time, to assess whether relative contributions to the adult fished stock can be generated for the different habitat types and areas (NIWA's MBIE programme 'Juvenile fish habitat bottlenecks', CO1X1618). Other work in this programme has also shown that juvenile snapper growth in subtidal seagrass is significantly faster than in other un-vegetated habitat types. For example, at 40 days of age post-settlement, juvenile snapper from subtidal seagrass areas weighed 1.45 times more than non-seagrass fish; this increased to 1.87 times by 70 days age (Stewart 2018). Year 2020/21 work is quantifying whether juvenile snapper also experience much higher survivorship rates in subtidal seagrass relative to other habitats. Aside from snapper, subtidal seagrass also holds high densities of juvenile parore, which migrate with increasing size/age from estuarine nursery habitats (subtidal seagrass, Morrison et al. 2014a–c), and to a lesser degree, mangroves (Morrisey et al. 2007) and shallow sandstone reefs (Morrison 1990) out to shallow coastal reefs, where they provide an important browsing role as one of New Zealand's few herbivorous fish species. Large numbers of juvenile piper and spotties are also found in subtidal seagrass and presumably also migrate to join semi-pelagic and reef associated populations, respectively, where they are a prey source to larger fish, birds, and mammals.

These high fish nursery values are also seen in the Bay of Plenty (e.g., Whangapoua, Tauranga, and Ohiwa estuaries (Francis et al. 2005, 2011), and Te Huruhi Bay, Great Mercury Island (Schwarz et al. 2006), though the first three estuaries have lost some, and continue to lose, subtidal seagrass elements. For example, using aerial photography, Park (1999) showed that intertidal seagrass in Tauranga Harbour declined 34% between 1959 and 1996, but that subtidal seagrass declined by 90%. Earlier losses before 1959 were also thought highly likely. Land-based sedimentation was strongly inferred as the cause. Broad scale national beach seine sampling in the early 2000s for juvenile and small fish around New Zealand recorded juvenile fish (snapper included)-subtidal seagrass habitat associations at several Tauranga Harbour sites, as well as in Ohiwa and Whangapoua estuaries (Francis et al. 2006, 2011). However, by 2004, no subtidal seagrass was able to be found anywhere in Whangapoua Estuary despite a harbour-wide search. Wide-ranging searches in 2018 for subtidal seagrass throughout Tauranga Harbour, to catch 0+ snapper for otolith chemistry, found only one limited subtidal area with low density seagrass and, overall, few snapper were caught across four days of searching. Similarly, searching for subtidal seagrass in Ohiwa Harbour found only two sites, which returned relatively low snapper numbers for the areas covered (M. Morrison, NIWA, pers. obs.). At face value, these findings suggest that subtidal seagrass continues to be in serious decline (towards ecological extinction) in the estuaries of the Bay of Plenty.

Recovery and restoration potential

Seagrass as a marine plant growing often immediately adjacent to land run-off, is particularly vulnerable to human driven stressors and threats. Threats include:

- Increased sedimentation, both through new inputs and resuspension, which reduces water column light levels as well as covering seagrass blades and therefore reducing photosynthesis (Inglis 2003, Turner & Schwarz 2006)
- Increased nutrient levels from land run-off, which can promote overgrowth by epiphytic algae which reduces photosynthesis and encourages excessive growth of other species such as *Ulva* spp. As well as agriculture, other sources include septic tank leachate, urban stream and stormwater inflows, boat effluent, and grey water discharge [which often contains phosphorus, which promotes algal growth] (Matheson et al. 2010)
- Coastal development, vehicles on tidal flats, swing moorings, propeller scarring, and, in the past, dredging and spoil dumping (Whangarei Harbour, Morrison 2003), and in-water harbour infrastructure (Powell 1936)
- Intensive grazing by black swans (Matheson et al. 2008)
- Disease, e.g., the slime mould (*Labyrinthula zosterae*) was implicated (but not proven) in the loss of seagrass in the 1960s in the Waitematā Harbour and Avon-Heathcote estuary (Armiger 1964, 1965, Inglis 2003)

- Climate change effects, including coastal squeeze and increasing temperatures and water turbidity (Kettles & Bell 2016).

Substantial restoration efforts have been made internationally for seagrass.

In New Zealand, seagrass has been experimentally transplanted with success in Whangarei Harbour, using seagrass ‘sods’ and ‘sprigs’ (Matheson et al. 2017). Transplants from a remnant bed were placed in a mid-intertidal experimental block (18 x 0.25 m^2 plots) and monitored for survival for two years. Both methods were equally effective with plant cover increasing from < 1% to 63%. Cover across the wider transplant site increased from 10% to 46%, and the associated biomass from 58 to 321 g m^{-2}. The donor plots regenerated within 9 months. It was concluded that *Z. muelleri* can be rehabilitated by transplant if former sites have suitable growing conditions, i.e., the stressor/s that originally caused seagrass loss have been removed (Matheson et al. 2017).

Subsequent to the above study, the transplant plots of Matheson et al. (2017), an on-transplant of some of these plants to a second transplant sites, and a separate transplant experiment in Pāuatahanui Inlet, Wellington, were collectively monitored for 1.5 to 4 years, along with photosynthetic light availability. The Whangarei Harbour transplant sites were successful, and the donor sites also recovered quickly (9–10 months); in Pāuatahanui Inlet the seagrass did not survive >14 months. Monitoring of light levels found that the failed site had seasonally lower light levels, with more than 25 days falling below 3.9 mol m^{-2} d^{-1} in autumn and winter (versus <10 at the successful site). Other mud related stress mechanisms also interacted with seasonal low light levels to negatively affect seagrass; combined, these findings emphasize the critical importance of limiting mud loading to estuaries to protect and restore seagrasses (Matheson et al. unpubl. data, pers. comm).

Work on intertidal seagrass in Tauranga Harbour has also provided further insight into how seagrass adapts to different environmental conditions in the intertidal zone (Kohlmeier et al. 2014a, b).

Knowledge gaps

- A reconstruction of the historical extent of seagrass—subtidal seagrass focused—is a key knowledge gap to address. Historical aerial imagery of Auckland is available back to at least the 1940s, and although not taken for this purpose, a methodical exploration of this imagery and associated spatial ‘capture’ of past distributions would be very valuable. Local and Ecological Knowledge (LEK) and Matauranga Maori knowledge will also exist across a range of local communities; this could be captured through structured interviews (see Morrison et al. 2014d for Kaipara Harbour LEK examples).

- Current NIWA work by Fleur Matheson, Andrew Swales, and others is looking at the large seagrass meadow in Anzac Bay, Waiheke Island. This seagrass meadow is the closest remnant to the Wairoa River mouth (east end of Tamaki Strait); it is suspected that historically seagrass beds may have occurred in Waiheke Island bays that are closer to the river. Light, suspended sediment, and other biophysical monitoring equipment are being deployed to help model the light climate for the Wairoa River receiving environment (*sensu* Gall et al. 2019), predict where seagrass should occur, and deploy transplants into those places as appropriate (F. Matheson, pers. comm.). These new current predictions could be combined with the historical distribution reconstruction, to target areas where the environmental conditions could be improved, for subsequent seagrass reconstruction.

4.2.4 Green-lipped mussels

This species is endemic to New Zealand, and once supported regionally important fisheries in the Hauraki Gulf, Marlborough Sounds, and the Kaipara Harbour. Today, it makes an important contribution to New Zealand economy through an extensive aquaculture industry.

Historical distribution

The Firth of Thames and the inner gulf across to Rangitoto Channel once supported extensive green-lipped mussel beds (Figure 11) (Reid 1969, Greenway 1969), which were fished to commercial and ecological extinction in the Hauraki Gulf from the 1920s to the mid-1960s. Greenway (1969) listed the main areas as Rangitoto Channel, Islington Bay, Tamaki Strait, Ponui Island, Ponui-Thames, Orere Point-New Brighton, and New Brighton to Thames. Mussel beds occurred along the entire Coromandel coast, from Te Puru to Colville. Only one general map of their historical distribution exists, created by Greenway (1969) from fisheries catch records and fisher knowledge. This shows ten high density areas (some quite large) which Greenway (1969) called dense beds, which covered around 500 km^2 of seafloor surrounded by a larger area of around 1000 km^2 referred to as dredged area. These observations are from fishing by dredging, with individual tows that may extend 100s of metres, which 'smooths' over finer scale density variations (including mussel clumping matrices/mosaics on a background of 'bare' soft sediment seafloor). Figure 29a shows the Okiwa bed at Great Barrier Island, as sampled by McLeod et al. (2014). This was a denser part of the bed, with seafloor cover of around 50% mussels and 50% coarse sand (there were also large pipis present, some of which the mussel clumps were attached to (M. Morrison. NIWA, pers. obs.). The dense beds of Greenway (1969) were probably similar (Figure 11), with spatial variability in mussel densities and seafloor coverage at a range of spatial scales; whereas the wider area fished by dredge may have held lower densities and fewer mussel clumps. The two Gundlock dredges seen by Reid were 2.2 m and 2.7 m wide and were deployed with a tow speed of half a knot (0.93 km hr^{-1}). A tow of 20 minutes would have swept from 1354 to 1662 m^2, depending on dredge width. As with scallop dredging, tow distances vary by the skipper, depending on how rapidly the dredge fills with catch, bycatch, and general debris. The terms 'dense' and 'bed', as determined by dredge catch, do not automatically translate to what these terms might mean as used by a diver who is visually observing the same bed (unfortunately SCUBA diving only became popular and widespread after the demise of the gulf's soft sediment mussel beds). This definition issue needs to be kept in mind when discussing Reid's historical distribution map.

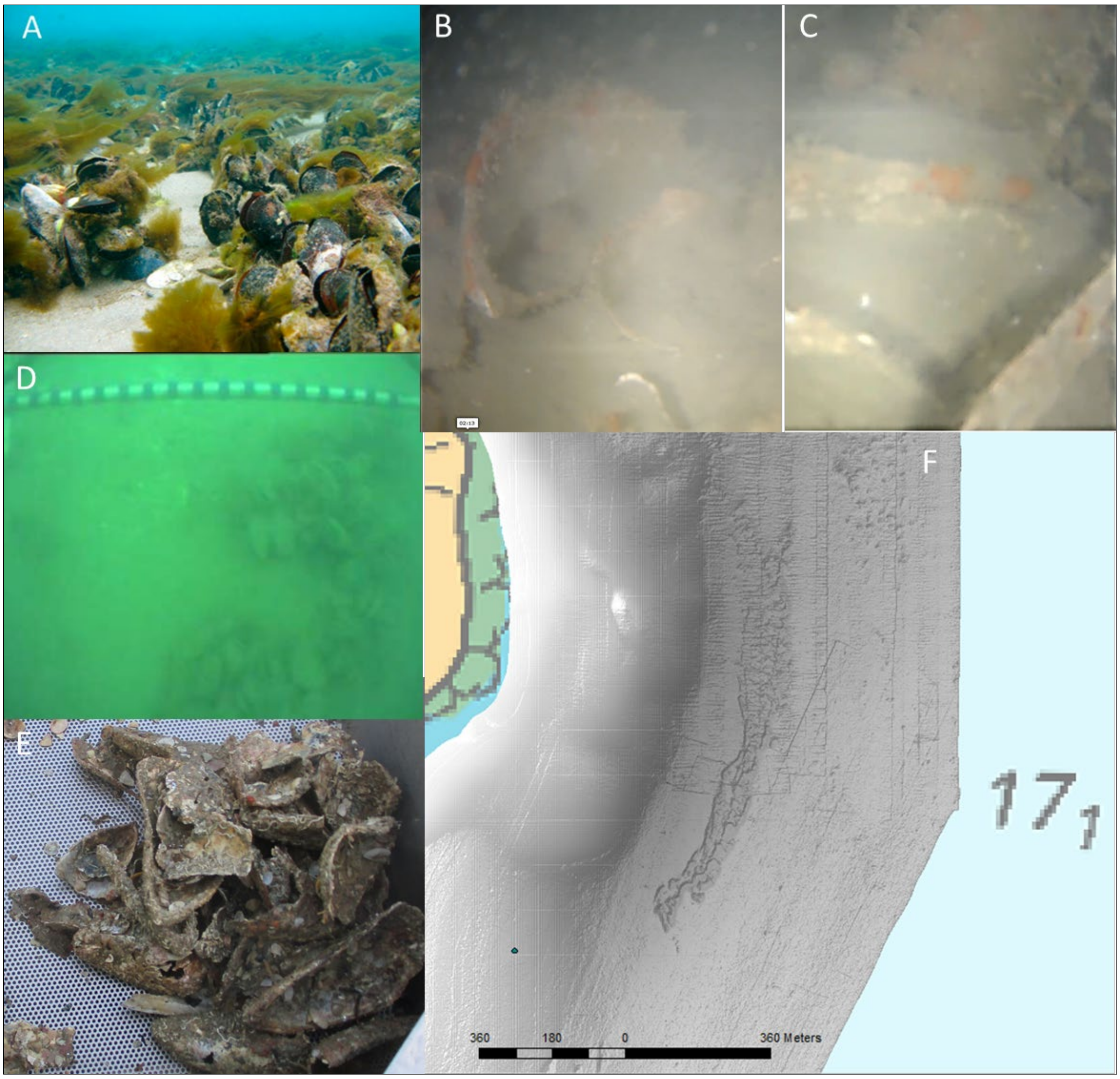


Figure 29: a) present day (2008) green-lipped mussel bed in Okiwa Estuary, Great Barrier Island; b, c) dead relic mussel shell (part of F bedform), as seen by towed video close to the seafloor in low visibility conditions; d) largest live green-lipped mussel patch seen in 2002 survey (Morrison et al. 2002); e) dead mussel shell as beam trawl bycatch, from a central Firth of Thames site (10 m water depth); f) multibeam sonar seafloor bedform feature of dead mussel shell in 20 m of water, east Ponui Island. Source: a, frontispiece of McLeod (2009); b, image from appendix 1 of Morrison et al. (2002); c–f, images collected as part of NIWA MBIE research programme 'Juvenile fish habitat bottlenecks CO1X1618, 2019–2020 period.

Present distribution

From 2017 to 2020, the NIWA MBIE research programme 'Juvenile fish habitat bottlenecks', and a related spin-off MPI project in 2019 (Morrison et al. 2019), used a small research beam trawl to undertake extensive and widespread sampling of the Hauraki Gulf soft sediment seafloor, from 3 to 30 m water depth. This small fine-meshed trawl net was slung from a 3 m wide bar and fished a width of around 2 m. Tows were a standard length of 400 m and took about 4 minutes on the bottom to complete. The main target was juvenile snapper (< 1 year old, < 100 mm long), but it also caught a range of other fish species, plant, and invertebrate bycatch. Although its efficiency in retaining bycatch was relatively low, biogenic habitat forming species were caught and duly recorded and measured for either volume or weight. From more than 600 tows, from Omaha Bay down to Whangamata and further south, and including the Firth of Thames, Little Barrier/Hauturu and Great Barrier/Aotea islands, catches of live green-lipped mussels were very rare and, where present, consisted of only 1–2 individuals at most in a tow (all adults). Dead relic mussel shell was far more common (Figure 29) and widespread across the Firth of Thames, as well as along the west Coromandel coast. In some places, some seabed forms seen on multibeam sonar bathymetry and backscatter, were found to be relic dead mussel shell accumulations when ground-truthed by towed video (Figure 29); although Smith-McIntyre grab samples also found dead mussel shell on or below the sediment surface (this device sampled 0.11 m^2 of seafloor, down to a depth of around 30 cm in soft muds and to lesser depths in coarser more compacted sediments). How dead shells accumulated (in situ, and/or through post-death shell transport processes) was not readily apparent; many shell deposits may lie more deeply buried under mud. Regardless, dead relic green-lipped mussel shell was widespread. Surface dead relic shell surface accumulations appeared more common in the middle of the firth, being found in a range of beam trawl stations ranging from 10 to 30 m depth along the firth's mid-axis (no stations deeper than 30 m were sampled). Mixed with this were relic shells from other bivalve species, especially scallops, as well as larger infaunal bivalves (not dog cockles).

Services and value of habitat

Green-lipped mussels, as with other epifaunal and infaunal bivalve species, provide a range of goods and services, especially when they occur at high densities in beds. A summary of the goods and services, also relevant for other bivalve species covered in this report (scallops, horse mussels, pipi, and cockles—large infaunal bivalve species) includes:

- Cultural, social, and economic value, as food, for those species that are consumed by humans, e.g., the historical fishery for green-lipped mussels in the gulf, as well as the fishery for scallops (see scallop section). Provision of kai moana for iwi, both for everyday diet, and for special events, is a particularly important role in the New Zealand context.
- Provision of harder surfaces for settlement and growth of other sessile plant and animal species (Gutierrez et al. 2003), as live epifaunal shellfish and dead epifaunal/infaunal shell, e.g., barnacles, ascidians, tubeworms, macroalgae, and *Ecklonia* kelp growing on horse mussels; dead dog cockle shell drifts holding various encrusting species, especially sponges, ascidians, bryozoans, and live attached *Ecklonia* and other kelps (in some contexts).
- Proving shelter, as well as prey assemblages, for a range of mobile animal species beds (e.g., live horse mussels, Cummings et al. 1998; and dead dog cockle shell drifts, Dewas & O'Shea 2012, Beaumont et al. 2015). Large mobile animal species such as fish, birds, and marine mammals may also preferentially forage over such habitats.
- Acting as nursery habitats for many species, including those valued as fisheries species by humans (e.g., Morrison et al. 2014a).
- Maintaining water clarity and quality by their filter feeding actions in large numbers. This includes the removal of suspended sediment from the water column, which is bound in pseudo-faeces and dropped to the seafloor.
- The accumulation and sequestering of carbon. Dead shells can persist in the marine environment for very long time periods, especially when buried, and probably represent one of the most common fossil forms observed. Dead disarticulated dog cockle shells (*Tucetona. laticostata*) collected from the Wanganui Shelf have been carbon dated at 9170 ± 210 years BP (Gillespie et al. 1998) and relict surface deposits shells from the Bay of Plenty at 35 800 ± 2250 years (Beu 2004). As an example, Smith et al. (2010) calculated a carbonate sediment budget (including bivalve contributions) for Otago Harbour. Carbonate was added to the system by in-situ production (~10 000 tonnes $CaCO_3$ y^{-1}) and transport though the harbour entrance from the longshore system (~24 000 tonnes $CaCO^3$ y^{-1}). Removals were calculated as fisheries shell harvest (~2 tonnes $CaCO_3$ y^{-1}), dredging (~18 000 tonnes $CaCO_3$ y^{-1}), and early seafloor processes such as abrasion and dissolution (~2000 tonnes $CaCO_3$ y^{-1}). This results in an annual net accumulation of ~14 000 tonnes $CaCO_3$ y^{-1} in sediment storage—equivalent to ~0.14 mm y^{-1}. Calculation of the budget for 2000 years ago suggested a storage rate twice that of the present (no human-driven removals), and, projecting human impacts into the future, the

authors suggested that carbonate storage may end within 100 years. No similar studies are available for the gulf.

- Removal of excess nutrients, including through denitrification (Cloern 2001). In the Firth of Thames, on average the riverine supply of organic and inorganic nitrogen is greater than the supply coming from mixing across the boundary between the firth and the wider Hauraki Gulf (J. Zeldis, NIWA, unpublished results). During periods of down-welling domination over the adjacent continental shelf, rivers contribute about 70% of the dissolved inorganic nitrogen load, dropping to about 50% when upwelling is active. This present-day water quality is significantly enriched, and it is likely that its productivity is substantially higher now than before the arrival of humans. The system is highly 'net-heterotrophic', consuming substantial organic matter and producing inorganic nutrients and dissolved inorganic carbon. It has been suggested that pastoral catchment development has resulted in strong effects on the Firth of Thames ecosystem, and that in turn the firth will respond to changes in catchment management to the extent that it affects nutrient loading (J. Zeldis, NIWA, unpublished results). Large-scale re-establishment of green-lipped mussel beds may help with balancing the system back to more natural levels. Suspension feeders are a key biological component of the 'nutrient filter'; if rates of particle filtering are high enough, they can balance out the rate of phytoplankton primary production, as a 'top-down' control process (grazing) (Cloern 1999, 2001). However, it is recognised that if the suspended sediment issue of the firth is significantly improved, but not matched by large dissolved nutrient reductions, there is the potential for large-scale micro-algal blooms to occur, which could be disastrous, ecologically, and for the aquaculture and fishing industries. Mussel restoration might help mitigate this scenario by removing high nutrient levels from the water column.

There is a detailed and increasingly substantial science literature on the goods and services that bivalves provide, in far greater detail than can be covered here. Some key review papers, reports, and book chapters include:

- Molluscs as system engineers (Gutierrez et al. 2003).
- Goods and services of marine bivalves (book) (Smaal et al. 2019) – see chapters on water clarification services (Cranford 2019), bivalve assemblages as hotspots for biodiversity (Craeymeersch & Jansen 2019), and enhanced production of finfish and large crustaceans by bivalve reefs (Hancock & zu Ermgassen 2019), along with many others.
- Ecological mechanisms underpinning ecosystem service bundles in marine environments—a case study for shellfish (Rullens et al. 2019).
- The importance of habitat created by molluscan shellfish to managed species along the Atlantic Coast of the United States (Coen & Grizzle 2007).
- The role of bivalves (and other biogenic habitat formers) in supporting fisheries production in New Zealand (Morrison et al. 2014a–c), as well as land-based impacts on those fisheries (and biodiversity) roles (Morrison et al. 2009).

One role often brought up in discussions of green-lipped mussels in the Hauraki Gulf is the water filtering role that extensive dense seafloor beds once provided (and might provide again with restoration). Simple calculations of possible historical water turnover times for the Firth of Thames by filtering mussels were made by McLeod (2009). Assuming each mussel filters 350 l d^{-1} (Broekhuizen et al. 2002), and using mussel densities of 1.5 ind. m^{-2} and 120 ind. m^{-2} (as calculated towards the very end of the fishery by Greenway 1969 and from a current day Waimangu Point mussel bed), the pre-1958 abundance of mussels could in theory filter a volume of water equal to the volume of water in the Firth of Thames in 57 and 0.7 days, respectively.

Such calculations, however, ignore some critical density dependent and spatial structure processes. Recent field observations have confirmed food depletion by bivalve aggregations through their large collective filtering capacity can improve water clarity. However, as the standing stock of bivalves increases, inefficiencies increase through increased flow reduction from structure drag, which creates an increase in water re-filtration (Figure 30). Combined, these processes ultimately constrain the maximum water clarification capacity of populations, at levels that can be substantially less than previously predicted (Cranford 2019). The role of neighbour suspended particle depletion in the local feeding zone by conspecifics (and potentially other filter-feeders) is also largely overlooked when scaling up from individual feeding rates to entire populations (Cranford 2019). Bivalve distributions at finer spatial scales can range from flat homogenous layers (infaunal species) through to mosaics of mounds (e.g., green-lipped mussels, Figure 29), as well as having density variations within beds. These patterns will interact to determine what level of food supply any given individual shellfish can access.

The morphology of bivalve species will also affect access, with those with short siphons (e.g., mussels) experiencing the highest water re-filtration rates (Monismith et al. 1990, O'Riordan et al. 1993). In addition, if the water column has any form of stratification, and does not fully mix, then benthic shellfish beds will be unable to access the full water volume of a region, with associated reduced particle clearance rates.

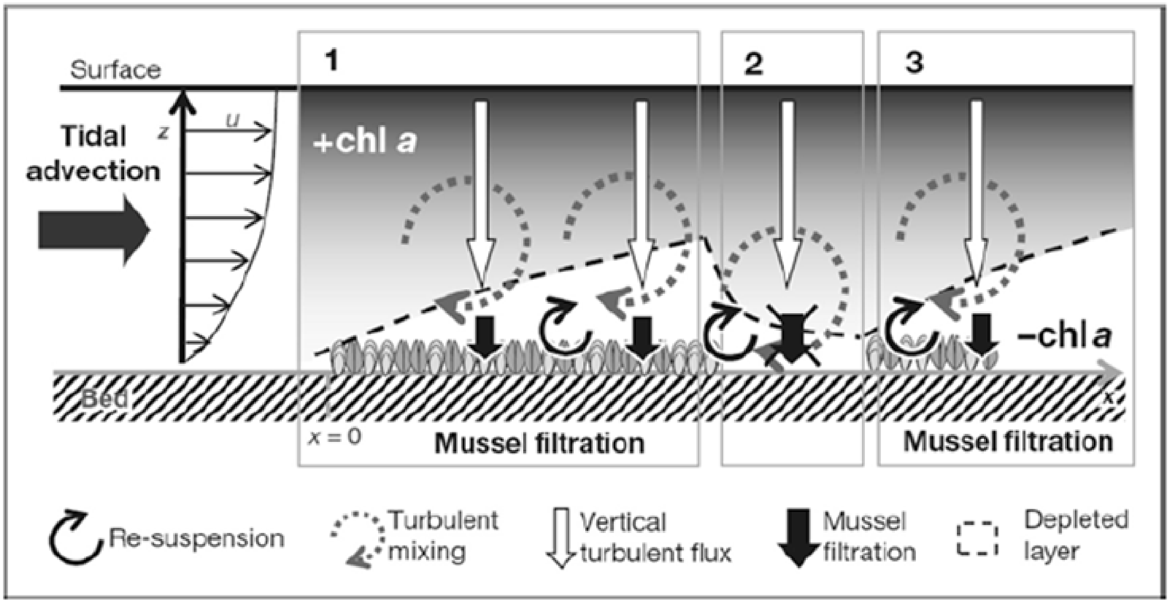


Figure 30: Graphic representation of the particle concentration boundary layer over a mussel bed, with a unidirectional water flow, showing some processes that control the magnitude of particle depletion (e.g., of phytoplankton, or suspended sediments). This includes a bare patch (Zone 2) between two mussel beds, to illustrate the effects of a habitat mosaic. The generation and area of the depleted zone is shown as the clear white polygons in the diagram. x = mussel bed length; z = water depth; u = water current velocity. Source: reproduced from Saurel et al. (2003) by Cranford (2019).

Another key factor to consider is that although filter feeding removes non-organic particles such as suspended sediment from the water column, and often binds them up in pseudo-faeces that are then released to the seafloor, these sediments are not removed from the ecosystem. Rather, they are transitioned from a pelagic state to a benthic state. Once on the seafloor, they can be resuspended by disturbances such as storm events and mechanical seafloor fishing gears (nets, trawls), because they are not 'locked' into the seafloor through chemical/mechanical binding processes, nor capped under layers of coarser seafloor material. Their negative impacts on the seafloor may increase through siltation over the key seafloor surfaces, including fine silt layering over plants blocking light (leaves, stems) and the smothering of surfaces that larvae need for settlement (Morrison et al. 2009).

McLeod et al. (2014) looked at the role of subtidal green-lipped mussel beds on soft sediments as a biogenic habitat for other species in the Hauraki Gulf. Using cores, the invertebrate fauna on and under mussel patches, and for adjacent bare sediment controls, was quantified. Invertebrate densities of the mussel patches (epifaunal and infaunal) were 2 to 8 times greater than the adjacent bare sediments.

Associated biomass values were 7 times higher. Species richness was also higher, especially of small crustaceans (0.5–5.6 mm) including calanoid copepods, and caridean, cumacean, and mysid shrimps, paguroidea (hermit crabs), porcellanidae (half crabs), pycnogonidia (sea spiders), and tanaidecea—all of which were absent in adjacent bare areas. Larger crustaceans (8.0–22.4 mm) were also much more abundant in mussel habitats, with high densities of *Petrolishes elongates* (blue half-crab) and *Halicarcinus innominatus* (pill box crab) at some sites. Such strong associations between mussels and high densities of crustaceans have also been documented in a number of international studies (e.g., Dittmann 1990, Moksnes et al. 1998, Ragnarsson & Raffaelli 1999, Moksnes 2002, Beadman et al. 2004, Lindsey et al. 2006), as well as the broader assemblage contrasts between mussel and adjacent non-mussel habitats (Ragnarsson & Raffaelli 1999, Duarte et al. 2006, Commito et al. 2008).

McLeod et al. (2014) calculated the potential loss of macrofaunal invertebrate productivity associated with the historical loss of 500 km^2 of Hauraki Gulf soft sediment mussel beds (Figure 10, pre-1958 estimate, Reid 1969). Using two estimates of historical mussel densities of 1.5 per m^2 and 120 per m^2 (see above), the total small mobile invertebrate loss estimates were 370- and 33 000-tons Ash-Free-Dry-Weight y^{-1} (note: this does not include larger mobile invertebrates such as predatory starfish and wandering anemones).

MacLeod et al. (2014) also sampled the small fish inside and outside using mussel patches using small 1 m^2 drop nets operated by divers. The fish catch was dominated by mottled triplefins (*Grahamina capito*), clingfish (*Trachelochismus melobesia*), and small spotties (*N. celidotus*). Overall small fish densities ranged from 2 to 10 individuals per square metre across the three locations sampled and were around ten times greater than densities on adjacent bare sediments. It is important to note the limitations of such diver-operated small drop nets in sampling juvenile/small fishes; more mobile species such as juvenile snapper and trevally will flee with the approach of divers, and, in this instance, the timing of sampling (October, May) did not coincide with when very small juveniles of these species are present in northeastern New Zealand. Other small relatively abundant fish species associated with epifaunal habitat-forming species that may grow on established mussel beds (e.g., bastard red cod and *Callyspongia* sponge species; juvenile leatherjackets with sponges and macroalgae) were also not included.

Given the large secondary productivity declines, changes in infaunal size distributions, and the loss of crustacean components from mussel beds to bare sediments, a strong cascading effect into epibenthic carnivores including fish (e.g., snapper) was highly likely. Using the two lost productivity estimates given above (370- and 33 000-tons Ash-Free-Dry-Weight y^{-1}), McLeod (2009) estimated that the small mobile invertebrate productivity associated with the pre-1958 mussel reefs could have supported an additional biomass of between 200 and 16 000 metric T y^{-1} of predatory fish above those able to be supported by ‘bare’ sediment areas, which replaced the reefs from the late 1960s onwards. The extra production supported by the current day Waimangu Point mussel bed (640 000 m^2) was estimated to be 20 metric T y^{-1}.

Recovery and restoration potential

Considerable efforts are being made to restore green-lipped mussel beds in the gulf. Initial pioneering work was done by McLeod (2009) as a Master of Science thesis and subsequently published by McLeod et al. (2012). Adult mussels were transplanted into small cages on the seafloor at three locations along the western side of the Firth of Thames that fell along a turbidity gradient (average visibility 0.8–4.7 m). After 500 days, 68% of the mussels were still alive and had grown an average of 19 mm in length. Mussels at the least turbid site were in better condition than those from the most turbid site. Deployment of settlement surfaces (the live mussels, adjacent dead mussel shell, and spat collecting materials) did not return any new mussel recruits. They concluded that the lack of recovery of mussel reefs in the firth was due to low recruitment and survivorship of mussels, and that the “*restoration of mussel reefs and the ecosystem services that they provide may therefore be possible*” (McLeod et al. 2012).

Subsequently, a group known as Revive Our Gulf / The Mussel Restoration Trust was set up to work to restore the green-lipped mussel beds of the Hauraki Gulf (http://www.reviveourgulf.org.nz/), and several reports and science papers have been published. Sim-Smith & Kelly (2015) assessed the seafloor habitats and species assemblages around Rotoroa Island on the outer west side of the Firth of Thames, looking for suitable trial mussel restoration sites using a combination of dropped and towed cameras. They used a range of criteria for selecting potential sites, including seeking sites that had the following:

- moderate or dense shell/gravel
- reasonable current flows
- no or few sensitive species present
- at least one drop camera site to verify the substrate
- historic mussel beds
- infrequent use as anchorages
- not sited near electrical cables and navigational markers
- in more than 5 m of water to provide a buffer against wave and storm action
- in less than 20 m water depth to allow for diving access
- not sited near areas of high predator abundance.

They found that the seafloor 500 m or more to the west and east of Rotorua Island was too muddy, with suitable substrates being found to the north, south, and east. However, these more suitable areas were already occupied by sensitive species. They commented that "*the ecological benefits of establishing a new mussel bed in these areas must be carefully considered against the possible adverse impacts that mussel bed deployment may have on the existing benthic community*". Additional sites around Ponui Island were assessed by Sim-Smith (2015) to determine their suitability for experiments testing the use of waste mussel shell to make muddy habitats more suitable for mussel bed restoration.

It was noted that "*the potential main ecological change of the proposed activity on the marine environment will be a change of habitat and benthic community from a muddy soft sediment habitat that is dominated by polychaetes, burrowing crustaceans, heart urchins and introduced bivalves, to a biogenic, hard substrate that is likely to support a diverse community of sponges, anemones, echinoderms, crabs, fish and gastropods. Although the current infaunal community will most likely be smothered by the deployment of the mussel and shell beds, the replacement community that is likely to colonize the area is considered to be of greater ecological benefit to the marine environment*".

The first larger scale restoration trial commenced in September 2014, with an estimated 70 t of farmed mussels (around 2.4 million individuals) dropped to the seafloor over a 10.5 ha area near Rotorua Island. These mussels were monitored at 3-month intervals from July 2015 to April 2016 using drop cameras and divers. At twenty months post-deployment, 22% of the mussels remained, with most of the losses stated to occur in the first four months of the trial, after which the losses slowed, although monitoring only started 9 months after deployment (Sim-Smith & Kelly 2016). Most of the loss was attributed to mussels being buried in the soft sediment, with predation and wider dispersal outside the surveyed area being unknown factors. In April 2016, 29.9 t were estimated to remain, but with a very large 95% confidence limit (± 20.8 t) due to the patchy distribution of the remaining mussels. The growth of mussels was very slow, growing an average of 9 mm in shell length over 10 months. No evidence of juvenile recruitment to the adult mussels was observed.

Wilcox et al. (2018) used four experimental beds established in the Rotorua Island area, monitoring them every 6 months over a two-year period. Only 26.2% of the mussels survived until the end of the study. No significant juvenile recruitment was observed, with only three new recruits being found during the study.

Wilcox & Jeffs (2017) looked at the potential influence of attachment substrates on mussel ability to establish on soft sediments. Field experiments where shell material was added to soft sediments showed no subsequent effect on the retention of adult mussels versus un-altered soft sediments. Laboratory experiments with juvenile mussels found that juveniles preferentially attached to adult mussels rather than unaltered soft sediments; and that juvenile survival in the presence of a common starfish predator was higher when attached to adult mussels, rather than un-altered soft sediments or dead mussel shell. The authors concluded that "*establishment of mussel beds on soft-sediment requires only adult mussels, which receive sufficient anchorage through attachment to neighbouring adults and in so doing, providing a stable, complex substrate suitable for improving the survival of establishing juvenile mussels by protecting them from sea star predators*".

Wilcox et al. (2020) used mussel spat collectors to look at patterns of settlement on transplanted beds of farmed mussels (see Wilcox et al. 2018). Settlement was found to be low and combined with a lack of recruitment to the transplanted beds, it was suggested that larval supply may be limited in the Hauraki Gulf and that further examinations into larval dispersal and retention was warranted.

Wilcox & Jeffs (2019) looked at the impacts of sea star predation on mussel bed restoration, using the same transplanted mussel beds as in the studies described above. Within two months of deployment, high abundance of the eleven-armed starfish *C. muricata* were attracted to the transplant plots, with their density continuing to increase for around eleven months. After that, it was thought that the beds had probably attracted most of the starfish within attraction distance (via odour plumes) of the beds. However, the further deployment of another 63 t of mussels about 100–500 m away may have also resulted in some starfish leaving the plot and moving away. It was estimated that over the 25 months of the study, 73.8% of the transplanted mussels died (Wilcox et al. 2018), with predation by the starfish being a major mortality contributor (although Wilcox et al. 2018 stated that one of the more evident signs of mortality was the burial of mussels). Mean estimated predation was estimated as accounting for 30.1% of the original population (40.4% of total mortality) and was identified as a potential significant issue for restoration success.

Issues have also been encountered with the transport of live adult mussels from mussel farms around the Hauraki Gulf for restoration/transplant trials. These may inadvertently also transport Non-Indigenous-Species (NIS), and such movements are currently banned under the Biosecurity Act by MPI. Although the argument has been advanced that many of the NIS organisms are already widespread in the Hauraki Gulf, including at both the donor (mussel farm) and transplant (seafloor seeding) areas, the issue remains that new and potentially very damaging NIS organisms might appear at any time, and such large-scale mussel transfers (multiple tens of metric tons) represent a real potential vector of spread. Mitigation of this issue might be achieved through selecting seafloor seeding sites that are close to existing mussel farms, to minimise required transport distances.

Another recent issue has been the transplanting of mussels into the Mahurangi Harbour. This harbour has never been recorded as supporting subtidal beds of green-lipped mussels on its soft sediments, including no oral histories of such beds in the past, nor any relic shell being found on the seafloor, as in the inner Hauraki Gulf and the Firth of Thames. Indeed, the historical large-scale fishing of soft-sediment green-lipped mussel beds has only been recorded as far north as the Rangitoto Channel on the western side of the gulf. As discussed earlier in this review, to clearly distinguish between habitat creation and habitat restoration, Elliott et al. (2007) recommended that "*producing new habitat can be termed creation or enhancement whereas re-creating habitat that was present within historical records, no matter how old, should be termed restoration*", and that marine habitat creation is "*an anthropogenic intervention which produces a habitat not previously there*". With no evidence that green-lipped mussel, soft sediment beds ever occurred naturally in the Mahurangi Harbour, such transplants are clearly habitat creation, rather than habitat restoration.

Marine habitat creation inevitably comes at the loss of another habitat, and whether it is an enhancement of the overall system is a value judgement. The past habitats of the Mahurangi Harbour were at least partly dominated by dense horse mussel beds and attached epifauna (e.g., sponges and ascidians), as well as scallop beds. The creation of extensive green-lipped mussel beds here may/will move the system to a very different state than that which historically naturally occurred (habitat creation, rather than habitat restoration), and is of questionable value.

Finally, as discussed by Elliot et al. (2007), the use of a standardised restoration terminology is a key component in maximising the effectiveness and success of restoring coastal ecosystems. Successful restoration is inferred to be a system that is "*self-sustaining and resilient; that is, it has the capacity for recovery from expected change and stress*" (SER 2004). Transplanting farmed mussels to the seafloor where a proportion of them survive for a while, but where there is no new mussel settlement and recruitment and the mussels die out over time and eventually become locally extinct, with no replacement by juveniles, cannot be claimed to be a successfully restored system (e.g., "*patterns of settlement within a <u>restored</u> mussel bed site*", Wilcox et al. 2020; "*the mean percent mortality of green-lipped mussels,* Perna canaliculus*, within <u>restored</u> mussel beds*", Wilcox & Jeffs 2019). Inappropriate use of terminology can be misleading and lead to false assumptions by individuals and agencies not directly involved with the work.

This also extends to statements of the historical area that once existed and may be used as a possible restoration target. The work of Reid (1969) and Greenway (1969) found that around 500 km^2 of dense mussel beds were commercially fished, with a further 1000 km^2 or so occasionally dredged. Dredges can be very efficient at catching objects sitting on the seafloor and so can still return good catches of objects such as mussels in lower density situations which would not qualify as beds. Recent authors have over-stated the likely historical extent of soft sediment mussel beds (e.g., "*these mussels once covered more than 1,300 km^2 of soft-sediment*", Wilcox et al. 2018; "*the green-lipped mussel,* Perna canaliculus*, which once covered extensive areas of the soft-substrate sea floor of the Hauraki Gulf, New Zealand (>1,300 km^2)*, Wilcox et al 2020). Such over-statements may lead to inappropriate restoration targets.

Knowledge gaps

Many gaps are noted above. Key gaps include:

- Why do adult mussel transplant plots not attract new settling mussel larvae?
- What role do mussel beds play as habitat in supporting fisheries production (as nurseries and feeding areas) and other trophic groups such as birds and marine mammals?
- Where did dense and extensive mussel beds once occur? Potentially addressed by the quantification of in situ death assemblages, which would also be useful for other species such as scallops and large clams.

4.2.5 Scallops

Scallops are epifaunal bivalves that can form dense beds ranging from tens of metres to kilometres in extent, as well as occurring in low densities as part of general soft sediment seafloor assemblages. Generally, they prefer coarser seafloor sediments such as sand and shell grit and are less commonly found on muddy seafloors.

Historical distribution

Death assemblages (old shells buried in/on the seafloor) around the Hauraki Gulf suggest that scallop populations may have been more widespread historically; dead scallop shells can persist in the marine environment for 500 years or more. For example, areas of relic dead scallop shell occur in the western half of Kawau Bay, extending from Matakana Estuary to as far west as at least Rabbit and Takangaroa islands, which appear to match the large suspended sediment plumes that can be seen extending out from the estuary during large stormwater run-off events. In the Firth of Thames, dead relic scallop shell is common at sites along the central axis of the Firth, from 10 m water depth to at least 35 m on the outer central bank.

Present distribution

The Hauraki Gulf, as part of the Coromandel scallop fishery, supports several larger beds that have been commercially fished on and off since the start of the fishery in the 1970s, along with some beds that have only been discovered in more recent decades (Figure 31). Many smaller scallop beds also exist around the Hauraki Gulf, some of which are closed to commercial fishing and reserved for recreational and customary fisheries (e.g., the Kawau Bay area, which contains smaller beds around its islands and on its shoals). The western bays of Great Barrier Island also support scallop beds, but these also have substantially declined over the last decade or so.

The most recent formal science survey of the Hauraki Gulf commercial fisheries scallop beds was completed in 2012 (Williams et al. 2013, Figure 31), followed by a fishers led survey with some science support in 2015 (no report published). Corrected to the same survey extents, a comparison of the two surveys showed a very large scallop population decline, from 110.4 million scallops in 2012 to 14.5 million scallops in 2015 (a 86% decline) (Table 9). For the very large scallop bed discovered in 2011 (stratum Hauraki Gulf/HGulf, areas 6–8) which held 77% of all surveyed scallops in 2012, the declines were even greater, at 97%. Although scallop populations are well known for population variations over time, these are well above what is seen in year-to-year natural variations.

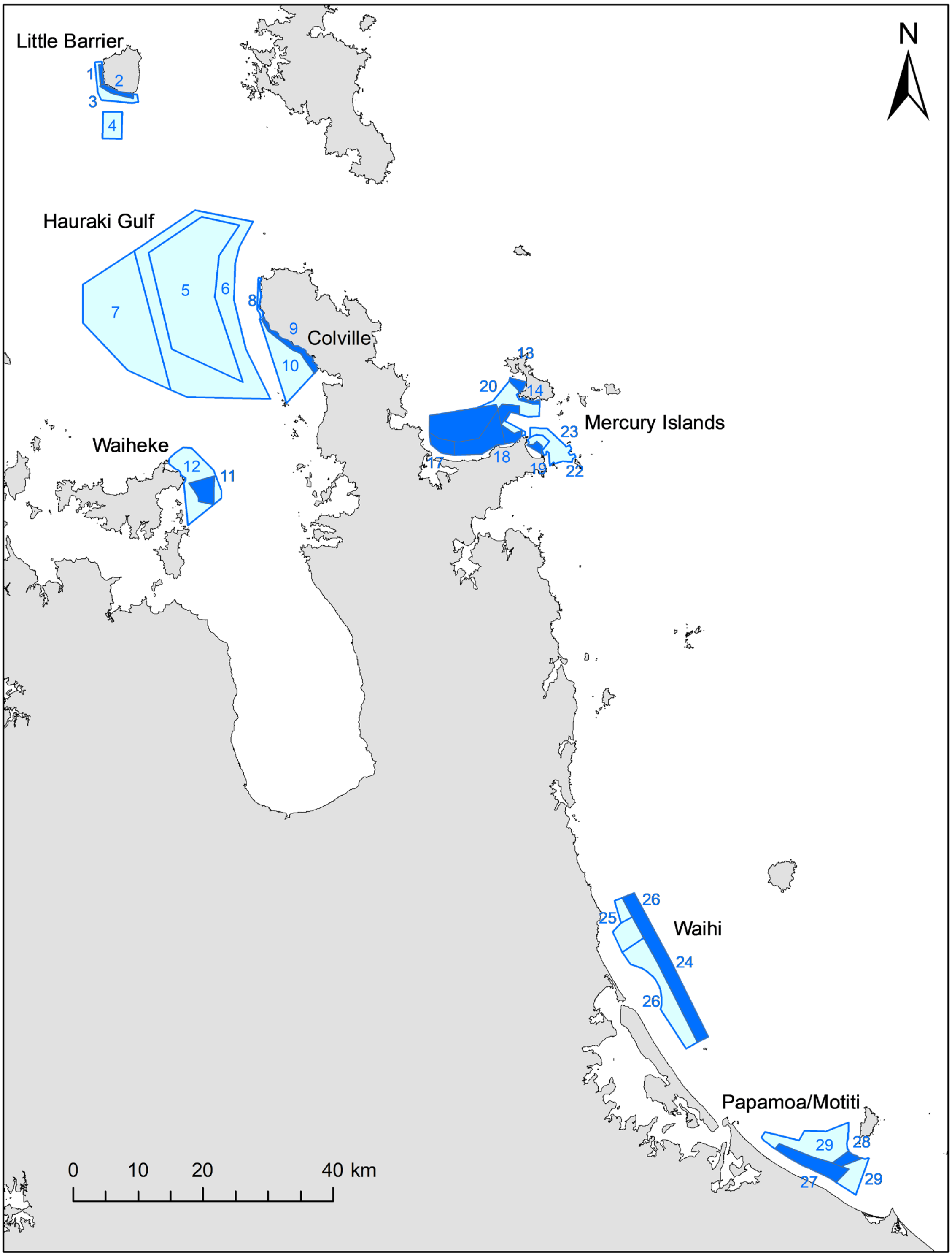


Figure 31: Location of strata for the 2012 Coromandel scallop stock survey. The dark blue polygons are 'Core' strata—areas consistently surveyed and fished in the past. Strata 5, 6, and 7 represent 'new' beds (located in 40–50 metres water depth) not previously known to the fishery before 2011. At the time of the 2012 scallop dredge survey, stratum 5 had been fished for the first time in the 2011 season, and strata 6 and 7 had not yet been fished. Source; Williams et al. (2013).

Table 9: Comparisons of 2012 and 2015 scallop surveys of the Hauraki Gulf, expressed as scallop abundance, green weight (whole scallop in metric tons), and meat weight (roe and adductor muscle only). Source: table 8 of scallop section, Fisheries New Zealand (2021).

		Abundance					Green weight					Meat weight				
Stratum	Area (km^2)	2012 (mlllions)	CV	2015 (mlllions)	CV	% change	2012 (tons)	CV	2015 (tons)	CV	% change	2012 (tons)	CV	2015 (tons)	CV	% change
Barrier	4	6.4	0.23	1.9	0.36	-0.70	466	0.2	136	0.37	-0.71	57	0.34	16	0.39	-0.72
HGulf	205	77.1	0.23	2.6	0.29	-0.97	6505	0.23	191	0.29	-0.97	794	0.36	23	0.32	-0.97
Colville	10	1.8	0.28	0.4	0.45	-0.78	156	0.31	27	0.45	-0.83	19	0.34	3	0.47	-0.84
Mercury	46	15.4	0.16	9.6	0.45	-0.38	1147	0.15	698	0.23	-0.39	137	0.2	83	0.29	-0.39
Total	**265**	**100.4**	**0.18**	**14.5**	**0.25**	**-0.86**	**8235**	**0.19**	**1065**	**0.18**	**-0.87**	**1014**	**0.21**	**128**	**0.20**	**-0.87**

Services and value of habitat

Scallops provide a similar range of ecological goods and services to that of other shellfish such as green-lipped mussels but are much more highly valued as food by humans, with wild-caught scallops retailing at 10 to 20 times the price of farmed mussels (M. Morrison, fish-market observations).

Recovery and restoration potential

Scallop enhancement in the Hauraki Gulf was attempted in the late 1980s but failed (Bartrom 1990). About 6 million scallop spat were released in the Firth of Thames in March 1988; their survival by September 1988 was estimated at 3.6%. In 1989, 13.7 million spat were released in Mercury Bay, Whitianga; their survival rate by June 1989 was effectively zero (Bartrom (1990). The reasons for these failures were not clear, but contributing factors appeared to be poor handling of spat during seeding, bad weather (including Cyclone Bola), poor selection of release sites, and loss of badly stressed spat to bottom predators. The lack of detailed monitoring of spat mortality and stress throughout each of the different steps (in spat bags, during release operations, immediately post-release, and subsequently on the seafloor) was a problem. Morrison & Cryer (2003), in a report on the possibility of scallop enhancement for Northland that referenced Bartrom (1989), emphasised that "*future work should be directed more at assessing the scallop survival rates during each of the steps and avoid simply releasing large numbers of spat to the seafloor and 'hoping for the best'*".

Of note, scallop spat counts per bag greater than 500 individuals were recorded at three sites within the Hauraki Gulf area: South Ponui, Ruth Passage, and North Gulf. A South Gulf site had a peak settlement of 497 spat per bag, and Wilsons Bay had a peak of 238 scallops per bag. Fouling by non-target organisms was suggested to be a potentially major problem in the Hauraki Gulf, with all collecting sites experiencing heavy fouling apart from those in the south and north gulf (both in deeper, more open water). The fouling species included the nesting mussel *Modíolarca impacta*, the green-lipped mussel *Perna canaliculus*, polychaete worms, and various algae. Eastern gulf sites held higher mussel numbers, whereas the western side sites had more algal growth.

Subsequent spat catching work was carried out further north in Greater Omaha and Kawau bays, from 1992 to 1994 (Morrison 1999). In 1992, high scallop spat catches were taken across all five sites assessed, with catches ranging from 100 to 3000 spat per bag. In 1993, there was an almost complete recruitment failure, with few spat caught. This coincided with a large-scale algal bloom event and heavy scallop mortalities throughout large parts of the Hauraki Gulf. In 1994, higher spat falls were again recorded, though lower than in 1992. Green-lipped mussel spat was also caught at times, ranging from a few tens to the high hundreds per bag. Following this, limited scallop spat catching trials were carried out by East Northland scallop fishers in the mid-1990s, giving returns of 100s to 1000s per bag.

Morrison (1999) also undertook small-scale spat release trials in Greater Omaha Bay. Because of the scarcity of spat at that time (as single droppers were used rather than bulk longlines), spat were removed from the spat bags, placed in plastic bags, and released by divers to the seafloor. Spat seemed stressed by this process and, on release to the seafloor, assumed a 'gaping' behaviour and did not exhibit their normal 'escape' responses. The release sites were over fine to coarse sand, in 7–16 m depth. Upon release, large numbers of invertebrate predators were immediately seen to erupt from the sediment surface and move towards the seeded spat. These predators included whelks, hermit crabs, and two species of starfish. In a stressed state, the spat was unable to avoid the predators, and mortality was high. Some predators were seen to consume more than one spat, and the scent plume seemed to attract predators from downstream of the release site. These observations show that predation of scallops immediately following seeding can be substantial, and that careful treatment of spat during seeding and selection of appropriate seafloor release areas are probably central to maximising survival (Morrison 1999).

Further research work in Omaha and Kawau bays was undertaken by Nesbit (1998). Spat catching was carried out at two of the locations monitored by Morrison (1999), using the same collection methods. Spat catches were very low, averaging 6–10 per bag. Release trials were undertaken with these spat onto two sites, one of sand, the other of shell gravel. Almost all the spat had disappeared from both sites after 2 months. Aquarium trials using invertebrate predators occurring at these sites showed that hermit crabs and starfish could successfully attack and consume most sizes of scallop spat used in the field experiments, resulting in up to 100% mortality over 2 weeks. However, scallops of about 30–40 mm (depending on the predator) seemed much less vulnerable to predation.

Morrison & Cryer (2003) provide a desktop review on the potential for artificial enhancement of scallops in Northland, which equally could be applied to the Hauraki Gulf. Enhancement is aimed more at establishing large high-density scallop beds for commercial harvest rather than restoration, though the same techniques can be used to re-establish beds that have been severely overfished or lost to some other short-term event. However, for the purposes of restoration and the ongoing health of scallop beds and the values that they provide, a more useful management direction may be in establishing and protecting 'brood-stock' reserves for scallops.

Knowledge gaps

Scallops, as with many other bivalves, display density-dependent spawning success; being effectively a sedentary species that broadcast spawn eggs and sperm up into the water column, where fertilisation occurs. Dense scallop aggregations (beds) generate better overall reproductive success because the encounter rates of sperm and eggs, and therefore successful egg fertilisation, are much higher (an 'Allele effect'). Scallops occurring as occasional solitary individuals may in fact be reproductively irrelevant at the population scale. Some beds may act as sources (produce many larvae that are then exported by water transport to other beds and areas), whereas others may act as sinks (receiving larvae from other beds); some may act as both sources and sinks. Such networks may even switch over time depending on weather patterns and their effects on water currents.

Scallop harvesting, both commercial and recreational, preferentially targets the densest beds, moving to less productive areas as the beds are diminished. Some beds are protected from commercial fishing by obstacles such as adjacent subtidal reefs and the presence of marine cables. Commercial fishing effort for a given bed ceases once scallop densities fall below those required to stay in profit; in contrast, recreationally fished beds often take longer for fishing effort to fall away once densities are low, as multiple different fishers may each fish the bed to learn it no longer holds scallops densities worth targeting. As technology (e.g., GPS to re-locate beds) and human populations increase, the probability of some dense beds continuing to survive and to act disproportionately as brood stock sources that provide new recruits to other more fished beds greatly declines. Ultimately, no reliable ongoing supply of new recruits means that all the scallop beds in an area will 'crash' and cease to exist as beds (along with their ecological functions, and fisheries harvests).

This life history 'bottleneck' of needing dense populations (beds) for ongoing successful reproduction is not considered in current management regimes for scallops in New Zealand. Existing known beds are heavily fished each year, and, if new beds are discovered, they are even more extensively targeted for fishing, as shown by the discovery of the mid Hauraki Gulf beds in 2011; these beds were then heavily commercially fished and by 2015 had dropped to only 3% of the population size and biomass measured in 2012, with the wider Hauraki Gulf commercial beds also ‘crashing’ (Table 9). It is quite possible that this large mid Hauraki Gulf scallop bed, previously unfished, was acting as a larval source bed for the other large beds around the gulf. Setting aside small areas of dense scallops as brood-stock reserves, after suitable research to identify the source-sink dynamics that may be operating in the gulf, would be a significant advance in working to rebuild scallop populations, both for ecological and human harvest values.

Another key advancement would be to remove the use of recreational scallop dredges from the Haruaki Gulf, as recommended in the Spatial Plan. Scallop larvae settling from the plankton require foliose surfaces to settle on, provided by other species such as red algae and hydroids, which in turn are often growing attached to dead adult scallop shells which provide hard stable attachment surfaces. Scallop dredging destroys these foliose surface organisms, as well as turning over/moving/removing and then throwing back dead scallop shells. It also re-suspends fine sediments, which are detrimental to both filter-feeding invertebrates and plants, both directly and indirectly (e.g., smothering hard surfaces required for larvae/spores to settle on and grow). Towed with a rope behind vessels, recreational dredge effects include flattening and dislodging species groups such as sponges, seaweeds, and hydroids; some of which are left loose and unattached on the sea floor (eventually to die), and others are retrieved in the dredge as bycatch and then thrown back into the sea (also to die). Species such as horse mussels are less likely to be caught as bycatch but, with relatively thin and weak shells, can be cracked by the impact of the towed dredge, as well as being punctured or chipped by the forward-facing steel tines/teeth on the dredge. Once the integrity of the shell is impaired, they are very prone to predation, as well as being unable to protect their soft bodies from sediments and will die.

The passage of the dredge also disturbs the seafloor surface, through ‘raking’ with the tines including the turning over of sedimentary species such as rhodoliths, physical contact of the dredge body, and dislodgment/suspension by water suction vortices. This can include the resuspension of fine sediments. Heavily fished scallop beds often have low levels of epifaunal species, and where the bottom type can ‘preserve’ the dredge footprint, raking can be seen on the seafloor from dredge tines (e.g., on rhodolith beds on Albert Shoal, Kawau Bay, pers. obs.). Recreational scallop dredges are often used in the gulf in areas where commercial scallop dredging is not allowed; these areas hold more biodiverse seafloor assemblages (e.g., around the island banks and shoals of Kawau Bay, and the Noises Islands) than the surrounding seafloor. A ban on the use of dredges, combined with the creation of brood stock reserves, would fundamentally improve the scallop populations of the gulf, for both human harvest (diving and snorkelling) and their ecological functions; as well as improve the epifaunal biodiversity values of such areas, including foraging habitats for valued finfish fisheries species. This assumes that indirect fishing impacts are the key pressure supressing scallop numbers, and that habitats and scallops can naturally restore themselves once that pressure has been removed (i.e., that these have not been pushed into some alternative state that they cannot recover from). Research in these areas would be useful to determine the value of a ban on the use of scallop dredges.

Commercial scallop dredging is a destructive fishing method, with a pass of a commercial dredge having a substantially greater impact than that of a recreational dredge. Some research work is underway in New Zealand to look at more benign methods using new technologies, e.g., underwater collecting robots that use Artificial Intelligence (AI) approaches to detect legal-sized scallops and retrieve them as targets.

4.2.6 Horse mussels

The large pinnid mussel *Atrina novaezealandiae*, known as the horse mussel, occurs from extreme low tide areas out to at least 45–70 m water depth on the continental shelf (Vooren 1975, Hopkins 2002), with dead shells having been reported from 200 m water depth (Hay 1990). Horse mussels grow to more than 40 cm in length and anchor the lower two-thirds of their shells in soft sediments using subsurface byssal threads. Mussel densities range from occasional scattered individuals through to very densely packed ‘beds’ that extend over hundreds to thousands of metres. They are relatively long lived (more than 10 years) (Butler & Brewster 1979, Hay 1990, Hopkins 2002), and recruitment appears to be highly variable between years, meaning that beds may appear and disappear over decadal scales (e.g., see Hayward et al. 1997). Usually individuals within a bed are all a similar size, suggesting discrete mass recruitment events. Observations of a horse mussel recruitment event (20–30 mm individuals) in Greater Omaha Bay (about 17–22 m, sand seafloor) in 1993 found very discrete and dense clumps (more than 30–40 cm diameter, 2–3 m spacing), extending for at least one kilometre along the depth contour. Mussels grew rapidly over the next six months (to about 100 mm) and, combined with individuals ‘pushing out’ laterally as well as mortality, produced patches of about 80 cm diameter containing lower density, larger animals (Morrison 1999, M. Morrison, pers. obs.).

Historical distribution

Horse mussels were once widespread in the Hauraki Gulf, but are now considered by knowledgeable marine scientists to have greatly declined in abundance over the last decade or so, for unknown reasons. This is based on observations in readily dive-able depths (< 30 metres) and largely in areas closed to commercial fishing using trawls or dredges. In the Mahurangi Harbour, the decline and loss of horse mussel beds since the early 1990s has been attributed to land-based sedimentation (Ellis et al. 2002). Very high-density recreational launch and yacht numbers covering areas of the mid and lower harbour over the summer holiday months may also have impacted heavily through anchor and associated chain damage; diving under some of these dense boat aggregations later revealed a seafloor literally littered with drifts of horse mussel shell shards (Morrison et al. 2014a) (see also Backhurst & Cole 2000). This kind of shell breakage has not been seen in areas that are not used as intensive anchorages.

Historically, horse mussel beds have been deliberately targeted for removal from fished areas of the gulf, to ‘condition’ the seafloor for subsequent trawling (see historical section). This has also occurred in some past scallop dredge fishery areas of the gulf, with tows on scallop beds progressively pushed into adjacent horse mussel beds, on the assumption that knocking them down opened new ground for expansion of the scallop beds. Hay (1990) observed intensive commercial scallop dredging across part of a research survey transect in Guards Bay, Marlborough Sounds (15 December 1987). The fished area was described as a flat featureless area “*completely criss-crossed with the marks of the scallop dredges*”, with no live horse mussels, but abundant broken shell; whereas the un-dredged area (on a bank avoided by the dredgers) had horse mussel densities of 3–5 per m^2, with a “*lush growth of epiphytic foliose red seaweeds and* Ecklonia*, and abundant fish life*”. Fishers were reported to generally avoid areas of high-density horse mussels due to their interference with fishing operations, but also to commonly “*flatten areas of horse mussels to render the bottom terrain more suitable for dredging and trawling in future years*” (Hay 1990). Hay suggested that significant horse mussel habitat on the outer Marlborough Sounds, inside the sounds (e.g., Ketu Bay), and inside Croiselles Harbour had probably been destroyed by commercial trawling and dredging.

Present distribution

There are no systematic maps of horse mussel distributions in the gulf, although in recent years this species has been recorded as bycatch during scallop dredging and finfish trawl surveys (e.g., see Figure 32a, noting this only covers small sampled areas of the gulf). Horse mussels tend to occur in coarser sediments and where current speeds are higher (Figure 32d) but occur across most bottom types and environments, with the exception of very soft muds. Beds can be remotely mapped using sidescan and multibeam sonar technologies, especially over softer sediments that provide stronger contrast to the mussels themselves; however, horse mussel beds on coarser harder sediments such as shell gravels are less clearly defined and require more analytical work to look for spectrum return differences (both in bathymetry fine-scale patterning and in back-scatter values) to allow them to be mapped.

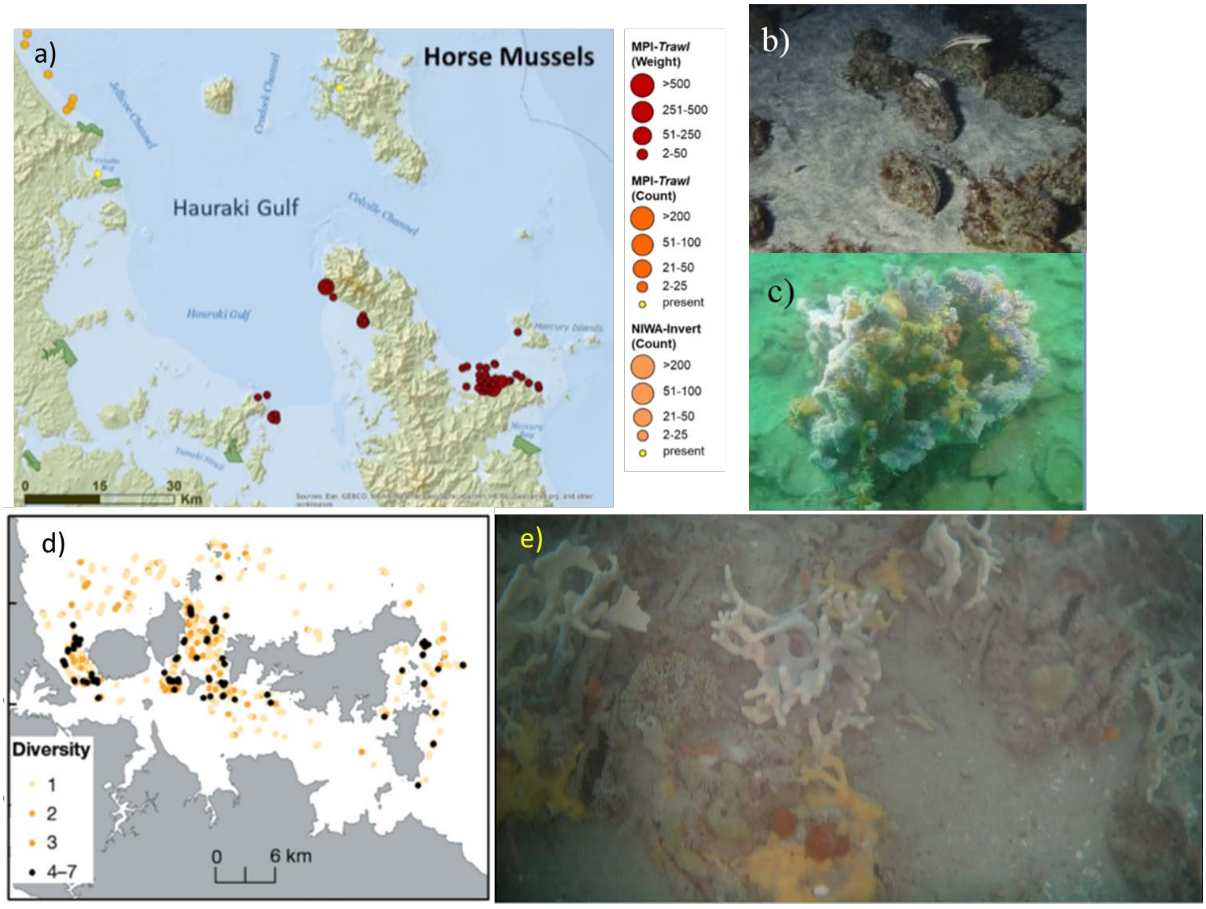


Figure 32: a) Horse mussel occurrences in the Hauraki Gulf. Total abundance (light orange bubble plots); species presence (yellow circles represent all available datasets combined (OBIS-NZ, Te Papa Tongarewa, and NIWA-Inverts); red circles represent research trawl survey catches (green weight; data source: MPI trawl database); dark orange circles represent scallop research dredge survey catches (data source: MPI scallop database). Abundances and weights are raw/unstandardised values used to indicate where bivalve beds (rather than isolated individuals) may occur; b) juvenile blue cod (about 8–10 cm) resting on horse mussels in Goat Island Bay, Leigh, Northland (about 30 m depth); c) horse mussel covered by sponges and hermit crabs, Tamaki Strait, Auckland (about 10 m depth); d) benthic epifaunal diversity (horse mussels, sponges, other larger epifauna) in 10-m long towed video seafloor segments; e) diverse sponge, ascidian, and bryozoan assemblage growing on horse mussel bed, northeastern Waikehe Island, 2020). Source: a) adapted from figure 3-48 of Anderson et al. (2019); b) Grant-Mackie (1987); c) James Williams, NIWA; d) adapted from appendix 1 of Compton et al. (2012); e) video footage collected in MBIE Juvenile fish habitat bottlenecks programme (CO1X1618), 2020.

Services and value of habitat

As well as providing the same range of good and services as other bivalves do (see green-lipped mussels section), the larger size of horse mussels and their extension above the seafloor make them an important emergent substrate habitat for many other epifaunal species, especially sponges, bryozoans, ascidians, and anemones (Figure 32c, d), as well as providing nursery habitat for juvenile fish (Figure 32b) and small mobile animals (Taylor & Morrison 2008). In areas with high water clarities, a range of seaweeds may also grow attached to horse mussels (Hay 1990). Across the shallow areas of the gulf, horse mussels in association with attached sponges provide useful nursery habitats for young-of-the-year snapper, trevally, leatherjackets, goatfish, and spotties, as well as adult triplefins and bastard red cod. Larger fish are also attracted to them for foraging, and snapper longline catches over horse mussel beds are often higher than for surrounding bare sediment areas (M. Morrison, NIWA, pers. obs.). The fish associations of horse mussels in deeper waters of the gulf are unknown, but probably include species such as tarakihi (adults only, juveniles do not occur in the Gulf) (Morrison et al. 2014b).

Recovery and restoration potential

Horse mussels are a good candidate for restoration in the gulf, although without knowing why they have declined recently, caution should be exercised. Ideally, the reason/s for their recent decline should be explored before attempting restoration, to ensure that the mechanisms involved are in the past and will not negatively impact on restoration efforts. Physical transplanting of adults seems to be of little value, because no farmed sources exist and moving animals from once place to another would not increase the number of animals in the gulf. An exception to this might be to establish dense brood stock reserves (as discussed in detail above for scallops), because horse mussels are a sedentary species, and individuals packed closer together are likely to provide much higher reproductive outputs per individual, relative to sparsely scattered individuals. The same source-sink dynamics discussed in the scallop section also apply (e.g., Elasber et al. 2013). However, such transplants into denser beds would be very labour intensive, and expensive, especially if SCUBA divers are required, even at small scales.

An alternative is the use of horse mussel spat catchers to collect settling larvae, and then grow them on to a size more likely to survive transplantation as small juveniles to the seafloor. The standard collectors used for species such as scallops and green-lipped mussels are not suitable for catching horse mussel larvae; however, new alternative horse mussel spat collection approaches offer potential (M. Morrison, NIWA, pers. obs.).

Knowledge gaps

- Map the present and past distribution of horse mussel beds in the Hauraki Gulf
- Develop spat catching devices that capture horse mussel larvae and allow them to be grown on to a suitable juvenile size and then released

4.2.7 Infaunal bivalves: pipi, cockles, dog cockles, *Tawera spissa*, and others

A range of infaunal (living in sediment) bivalve species occur in New Zealand's coastal zone at sufficiently high densities to dominate the seafloor, both as dense beds and as dead shell surface deposits. Examples well known to New Zealanders include dense cockle (*Chione stutchburyi*) and pipi (*Paphies australis*) beds in estuaries. In deeper coastal areas, other bed-forming bivalve species appear such as *Tawera spissa* (morning glory shell). This species is found around New Zealand, ranging in abundance from low density components of general invertebrate sandy/shell grit habitat assemblages through to high density mono-specific beds which can extend over relatively large areas and completely dominate the seafloor as live animals and dead shell (Figure 33).

Figure 33: Dog cockle (*Tucetona laticostata*) habitat from the Noises Islands, inner Hauraki Gulf. a) surface of a dog cockle bed with dead shell; b) close-up of dead shell lying on sediment surface (also note presence of rhodoliths); c) mixture of dead dog cockles and live scallop and horse mussel. Source: S. Dewas, AUT University.

Historical distribution

Although generally poorly documented, widespread declines of pipi and cockles have occurred around much of the Hauraki Gulf, with sedimentation being the most likely dominant cause (Grant & Hay 2003, Hartill et al. 2004). At many locations in the upper area of estuaries, layers of dense cockle shell can be found 50 cm or more of mud, suggesting that many areas once favourable for cockle populations have been lost.

Fisheries New Zealand/MPI (and to a lesser extent the Hauraki Gulf Forum community bivalve monitoring programme) has run an annual monitoring estuary beach series of cockles and pipis in the Hauraki Gulf and more widely, since 1990. These surveys cover specific beds and small areas that are, or used to be, harvested by humans; they do not extend across full estuary and coastal bay extents. That series continues today, though sampling sites have varied over time, and not all beaches have been sampled in all years. Neubauer et al. (2015) analysed this series using a Bayesian state-space model that simultaneously analysed monitoring data from 21 incomplete survey time series (i.e., 21 different beaches, not all in the Hauraki Gulf), between 1999–2000 and 2014–15.

Several Coromandel sites (Whangapoua, Tairua, and Whangamata harbours) showed strong but smooth declines in large-size cockle densities, whereas several Hauraki Gulf sites (Eastern Beach, Cockle Bay, and Kawakawa Bay West) showed distinct increases in cockle densities. Beaches that were closed to harvesting were associated with a clear positive change in their trend estimates. This included posterior mean estimates of overall population growth rates of 17% at Eastern Beach and 24% at Whangateau Harbour, in contrast to negative estimated growth rates of 6% to 9% declines per year, respectively, prior to being closed to harvesting. Neubauer et al. (2015) concluded that beach closures are very likely to enable population recovery.

Present distribution

For example, a Greater Omaha Bay, Hauraki Gulf bed in 18–28 m water depth covered 1.5 km^2, with an average density of 907 individuals per m^2 (maximum 3476), and about 1.4 billion population size (Taylor & Morrison 2008). A larger infaunal bivalve in shallow coastal areas is the dog cockle *Tucetona laticostata* which forms extensive beds with shell drifts at some locations where current speeds are high and the bottom sediments coarser (Figure 33); there is also a smaller bodied, less ‘massive-shelled’ species, *Glycymeris modesta*. In more sheltered, or deeper open coast environments, the heavy and thick nature of shells may produce more complex shell drifts than smaller bivalve species, providing higher structural complexity.

Pipi and cockle beds are found across most of the Hauraki Gulf’s estuaries, with cockles tending to be found on more sheltered intertidal flats and channel edges, whereas pipis are more abundant on low wave energy beaches and can occur in particularly dense large adult beds on the bottom of estuary channels, e.g., Whangateau Harbour (Hooker 1995).

Services and value of habitat

At these densities, they act as key species, providing functions including bentho-pelagic coupling, nutrient transfer, phytoplankton abundance regulation, carbon sequestration, and food provision. There is no evidence that they directly provide shelter for fish, aside from some small cryptic forms (e.g., clingfish). Their dead shells tend to form dense, closely packed drifts, with little three-dimensional elevation, or associated small crevices and nooks, in areas of sufficiently low energy to allow dead shell to accumulate.

Dead shell accumulations (note that this includes epifaunal bivalves such as scallops, and brachiopods) may provide structural complexity for other species on otherwise relatively featureless seafloor. For example, Auster et al. (1991) found significant associations between individual shells of ocean quahog (*Arctica islandica*) and young-of-the-year (0+ juveniles) of ocean pout (*Macrozoarces americanus*) at a 55 m water depth, low relief, outer continental shelf site, North America. Individuals were found under and alongside hinged and single values, which were viewed as shelter. In Greater Omaha Bay, research dredge bycatch associated with dead hinged scallop shells included small octopus and clingfish with egg masses (Mark Morrison, NIWA, pers. obs.). In heavily fished epifaunal bivalve populations, dead shell densities may be significantly reduced through physical removal of live animals as catch and incidental mortality, fewer shellfish growing to larger sizes, mechanical dead shell fragmentation, dispersal of bed shell while being returned as bycatch, and the loss of 'cementing' functions provided by associated biota (e.g., sponges and nesting mussels; see Cranfield et al. 1999 for Foveaux Strait examples).

Dewas & O'Shea (2011) quantified dog cockle shell beds ("*large post-mortem deposits*") around Otara Island (Noises Islands, inner Hauraki Gulf), as well as shell grit and rock gravel. Invertebrate diversities and densities were consistently higher in the dead shell beds over time, compared with bare sediments. Three hundred and fifty-one species (or Operational Taxonomic Units, OTUs) were recorded, of which 30% were found only in dead shell habitat compared with 10.5% being in the shell and rock gravel habitats. Similarly, Hewitt et al. (2005) sampled dead shell drifts in the Tonga Island Marine Reserve (Tasman Bay), found significantly higher diversity than in adjacent bare substrates, and concluded that shell debris increased and maintained biodiversity. They emphasised the need to reduce disturbance regimes and to actively manage seafloor habitats in areas previously largely ignored.

Recovery and restoration potential

Cockle transplantation trials have been carried out in New Zealand. Marsden & Adkins (2010) provided a review of the status of cockle bed restoration in New Zealand and found that most experimental transplantations of cockles have been small-scale studies to investigate survival and growth. Stewart & Creese (2002) investigated transplanting cockle populations in the Whangateau Harbour and reported 30% survival of 10–18 mm cockles after a year, and 60–90% recovery for adults (25–32 mm shell length). More broadly, experimental transplants to assess the survival and health of cockles transplanted from clean sites to those exposed to contaminants or increased sedimentation have found poor survival rates (De Luca-Abbott et al. 2000, De Luca-Abbott 2001, Stewart 2005, Norkko et al. 2006).

Cummings et al. (2007) deployed transplant experiments in Whangarei Harbour (transplant site was predominantly fine sands), using adult cockles (25–32 mm shell). Survival after 35 weeks of winter-transplanted cockles was high, but with increased mortality in the autumn, especially in high-density plots. Approximately 30% of the transplants remained after 1 year. Mortality was associated with weather conditions (wind, higher daily mean temperature, larger temperature range). To avoid density-dependent growth issues, Cummings et al. (2007) recommended that transplant densities of 20 to 70 cockles per 900 cm^2 plot be used.

Knowledge gaps

- More experimental work on how infaunal shellfish beds can be restored through transplants, and how this might be scaled up to larger areas.
- An assessment of death assemblages versus current day distributions to better quantify what has been lost.
- Modelling of where suitable growing areas occur in the present (as per the seagrass modelling work being undertaken by Matheson et al., NIWA), combined with the death assemblage data to identify areas where environmental conditions might be best improved to allow for restoration to proceed.

4.2.8 Sponges

Sponges are a diverse species group and occur over both rocky reef and soft sediment habitat types. Although common in the Hauraki Gulf, no systematic surveys of their distribution and abundance have been undertaken, and they remain poorly described and mapped. Where they grow in higher abundance and extent to form the dominant cover, they are called sponge gardens. In shallow northeastern New Zealand, these often occur on flat reef basements covered by a thin layer of coarse sediment, as well as on more topographically complex reef; in some environmental contexts they can also form dense gardens attached to horse mussel beds. Many species are present; the dominant habitat formers possess morphologies which provide three-dimensional structure off the seafloor, e.g., bowls, finger, tube, and mound forms.

Historical distribution

Sponges are very likely to have been contributors to the 'live ground' described by commercial fishers. In areas of soft sediment that have harder structural elements for attachment (e.g., horse mussel and dog cockle shells), they can become locally abundant and provide significant three-dimensional habitat structure (e.g., Figure 32).

Present distribution

A well-known Hauraki Gulf example is the 'Sponge Garden' off Goat Island, within the Cape Rodney to Cape Okakari Marine Reserve (Battershill 1987). This site is in 18–27 m depths, has had 33 species recorded as present, and is defined by a high density of discrete branching and massive sponges, as well as encrusting sponges, e.g., *Polymastia granulosa*, *Aaptos*, *Raspailia topsenti*, *Axinella* n sp, *Cinachyra* n sp., and algal species (Battershill 1987).

Deeper water sponge assemblages in the Hauraki Gulf are very poorly investigated, and it is likely many discoveries remain to be made. Shears & Usmar (2003) assessed patch reef fish assemblages of a Cable Protection Zone (in theory a de facto marine reserve) in about 33–50 m water depth, west of Great Barrier Island ('North-west Reef'). Diving on the shallowest reef area, they reported a diverse encrusting invertebrate assemblage including the sponges *Ecionemia alata*, *Stellata crater*, *Dendrilla rosea*, *Raspailia* sp., and *Aaptos aaptos*, which provided most of the biogenic structure component. Soft corals (*Alcyonium aurantiacum*) and hydroids (e.g., *Solanderia ericopsis*) also contributed. Off the eastern side of Rakitu Island, Great Barrier Island, Hauraki Gulf, limited camera surveys of the deeper reef systems (55–120 m) found them to be sponge dominated, including some larger habitat formers (Morrison et al. 2001, Sivaguru & Grace 2002). Substantial siltation was evident at some sites. Deepwater reef habitats that are likely to support sponge assemblages are not well mapped for the Hauraki Gulf, but Waikato Regional Council has used old faring sheets and other information sources to map out likely reef areas within their region (Figure 34), with larger areas of deep reef around the Mercury Islands and Alderman Islands, off east Coromandel. Multibeam sonar mapping of the deep reefs of the Mokohinau Islands has also been completed, but not processed (Figure 19).

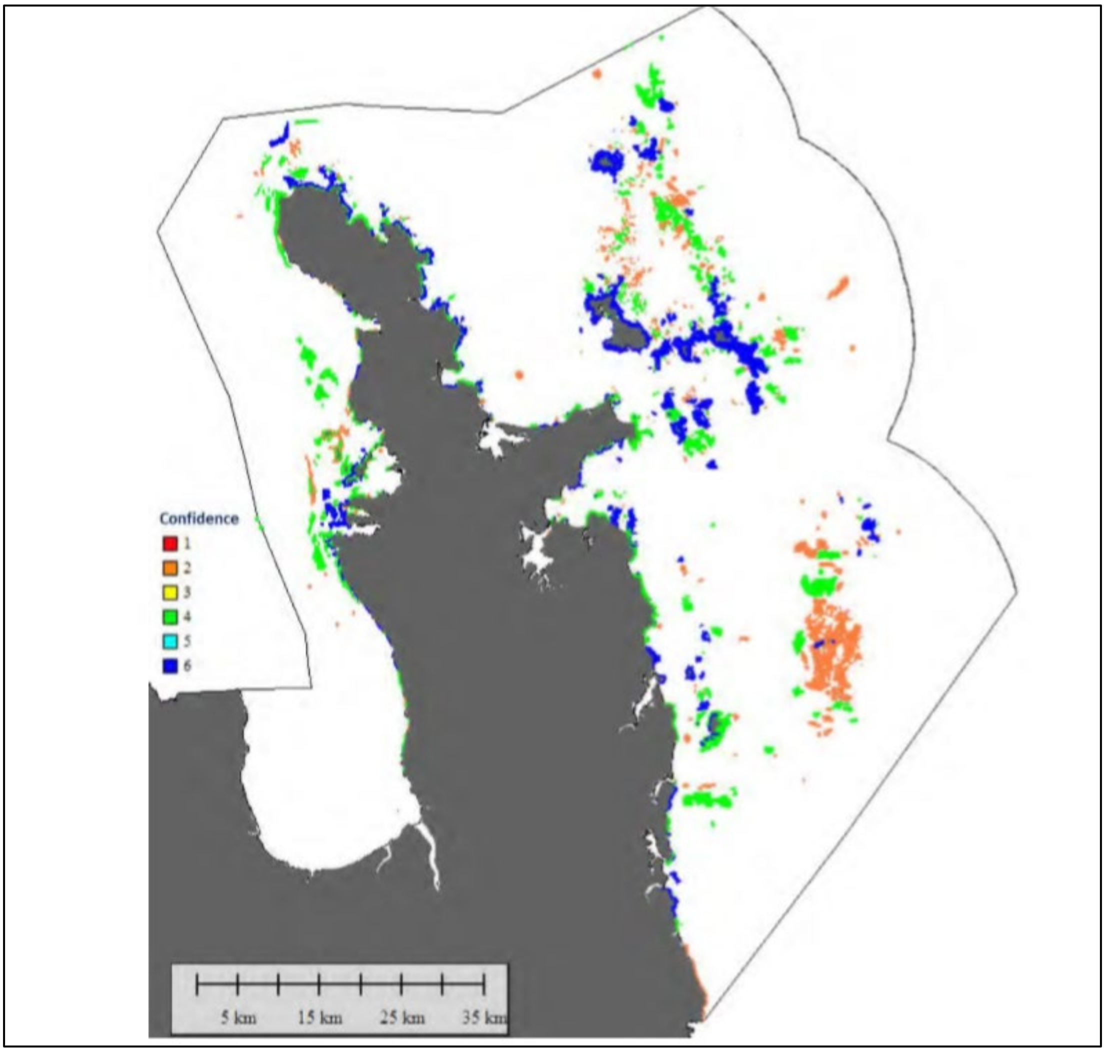


Figure 34: Rocky reef extent and confidence levels for the Coromandel Peninsula of the Hauraki Gulf. Source: figure 23 of Gardiner & Jones (2020).

Services and value of habitat

Bell (2008) reviewed the ecological role of sponges and concluded that they fulfil many important functional roles across temperate, tropical, and polar ecosystems; but that there was insufficient information to show the relative importance of these different functional roles. He stated that "*determining or ranking the functional roles is difficult, but it is important, since some roles that sponges fulfil will almost certainly be more important than others, and this may vary spatially or temporally*".

Bell (2008) grouped their functions under a range of headings, with an assessment of the international literature, as follows:

- Impacts on substratum:
    - Bio-erosion
    - Reef creation, and substrate stabilisation, consolidation, and regeneration
    - Importance of sponges on substrate
- Bentho-pelagic coupling:
    - Carbon cycling and energy flow
    - Silicon cycling
    - Oxygen depletion

    - Nitrogen cycling
    - Importance of sponge bentho-pelagic coupling
- Sponge associations with other organisms:
    - Facilitating primary production
    - Secondary production
    - Provision of microhabitat and sponge associations
    - Enhanced predation protection, survival success, range expansions, and camouflage though association with sponges
    - Sponge as a settlement substrate
    - Disrupting near-boundary and reef level flow regimes
    - Sponges as agents of biological disturbance
    - Sponges as releasers of chemicals
    - Sponges as tools
    - Importance of sponge interactions and associations

- Sponge gardens also have a high aesthetic value for tourism, especially to the diving and freediving communities.

Recovery and restoration potential

There are currently no data on what sponge habitats may have been lost, so it is not possible to determine recovery and restoration potentials. It is likely that sponges contributed to the ‘live ground’ habitat that commercial fishers believe has been lost from the gulf.

Knowledge gaps

- Maps of where different sponge species and assemblages occur in the gulf, both in the present day and historically. A focus might be on the contribution of sponges to the ‘live hard ground’ that commercial fishers believe has largely been lost from the gulf, and all the values that such habitat once provided.
- Quantitative descriptions of what the different sponge assemblages gardens are composed of (species composition and densities); these fundamentally change with factors such as water depth, exposure, and water clarity (including being displaced/out-competed with kelp forests for the latter with high water clarities).
- Much-improved knowledge is needed of what species are associated with different sponge assemblages, and what ecosystem goods and services they provide.

4.2.9 *Galeolaria hystrix* (calcareous mound-forming tubeworms)

This species can occur as both solitary individuals and collectively as mounds more than a metre high and several metres in diameter. Its range extends from the Hauraki Gulf down to Stewart Island (Hare 1992, Smith et al. 2005, Davidson et al. 2010), as well as New South Wales and South Australia (Day & Hutchings 1979). Until very recently in New Zealand, mounds have only been reported from the Marlborough Sounds (Davidson et al. 2010) and in Big Glory Bay, Stewart Island (Smith et al. 2005). In 2019, several mound fields were discovered in the Hauraki Gulf after appearing as mysterious small bathymetric bumps on multibeam sonar data and being ground-truthed by towed video camera (M. Morrison et al., NIWA, unpubl. data) (Figure 35).

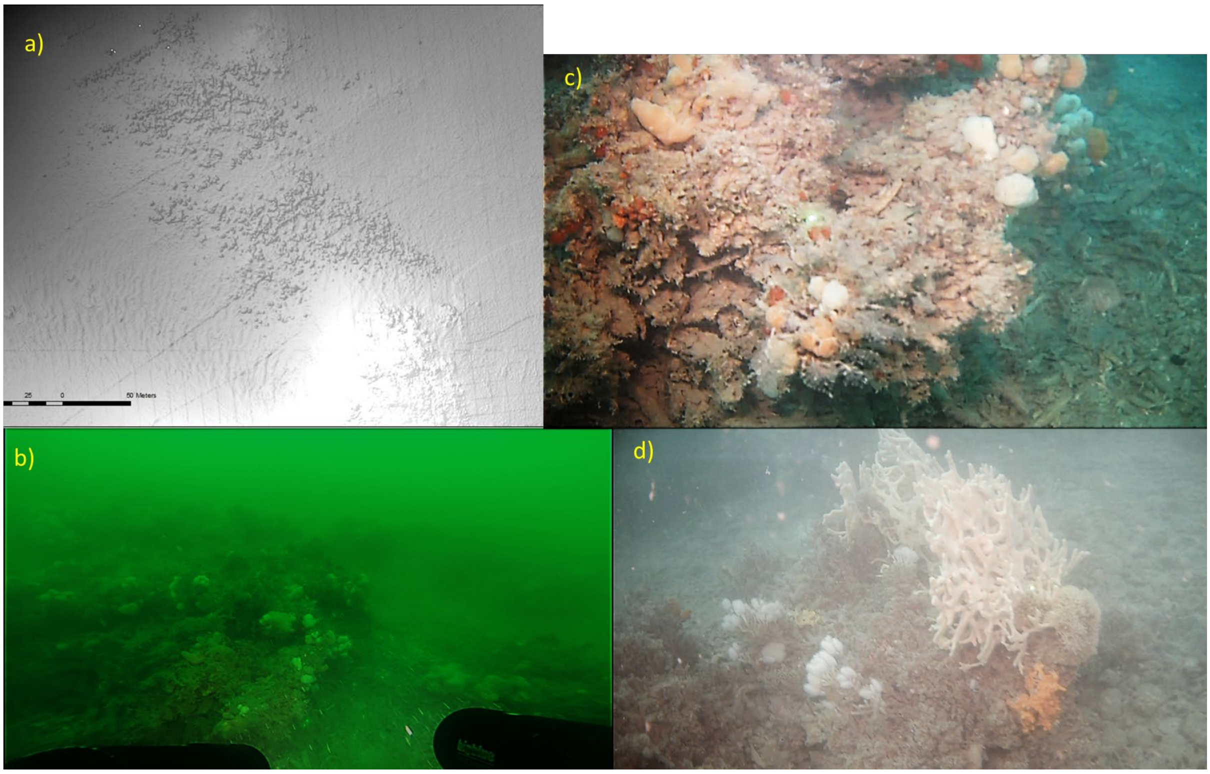


Figure35: *G. hystrix* mound fields in the Hauraki Gulf. a) multibeam sonar bathymetry of a field just north of Pakatoa Island, b) a video landscape view of the same field, c and d) closer views of the attached epifauna associated with the mounds. Source: MBIE Juvenile fish habitat bottlenecks programme (C01X1618) imagery, 2019–2020.

Historical distribution

This species probably was once much more widespread in the gulf in mound field form, although it has not been recorded/reported by divers. Widespread recreational diving commenced in the 1970s and is possible that mound areas were lost well before this, through fishing and sedimentation impacts.

Present distribution

Multiple small fields occur around most of Pakatoa Island on the west side of the Firth of Thames, as well as to either side of Motumorirau Island (north of Coromandel Harbour, on the west Coromandel Peninsula), totalling around 1 km^2 in extent. LINZ multibeam sonar data from 2021 off Coromandel Harbour, including around the Cow and Calf Islands, and Rangipukea Island, has revealed additional new (albeit small) likely fields. More tubeworm fields are likely to exist around the islands that form the Motukawao Islands /Happy Jack Islands, just to the north, and around Deadman's Point, just to the south. A healthy and epifaunally diverse *Galeolaria* field was observed by divers off Moturua Island in 2021 (pers. comm. Shane Kelly, Coast & Catchment Ltd; Shaun Lee, independent blogger), one of the larger islands of the Motukawao Islands.

Towed cameras over some of the mounds have revealed a rich epifaunal assemblage (e.g., sponges, ascidians, bryozoans) associated with the Pakatoa Island fields, whereas those on around Motumorirau Island have less associated epifauna and appear less biodiverse (no video analysis yet undertaken). Two beam trawl tows in areas adjacent to these fields have inadvertently caught *Galeolaria* mounds (one in 2017 off south Rotorua Island) and one in 2020 off Hautapu Rocks, West Coromandel. Both mounds held numbers of bastard red cod, rockfish (*Acanthoclinus* sp.), and black feathery brittle stars. Limited baited underwater video drops (2020) have also shown snapper associations with these mound fields.

Services and value of habitat

These are currently unknown, though it is apparent that they support a range of epifaunal and cryptic species (elevate biodiversity) and provide habitat for a range of fish species.

Recovery and restoration potential

Without evidence of once wider distributions, restoration is not warranted. A systematic assessment of the Hauraki Gulf's soft sediment death assemblages would help answer this question. Restoration efforts could readily involve transplantation of entire mounds from donor areas because these hold together well if treated carefully and could be reattached to the seafloor at transplant recipient sites using drilled holes through the mounds, with a support iron reinforcing bar extending down into the seafloor, secured to the mound itself with appropriate epoxies/glues. Because larval settlement is thought to be gregarious, and new generations of tubeworms build on top of the old ones, using transplanted mounds to establish new populations seems a logical approach. Work by Smith et al. (2005) on the Stewart Island fields has dated the base of a large mound and returned a maximum age of around 50 years, giving some idea of the timeframes involved for restored field development, and regeneration of old fields used as transplant donor sites (noting that smaller mounds will develop over much shorter time intervals).

Knowledge gaps

- Maps of where these tubeworm reef fields occur. The Pakatoa and Motumorirau island fields have been fully surveyed by multi-beam sonar, and the data are in the process of being turned into habitat maps as part of the MBIE Bottlenecks programme (e.g. Morrison et al. in review a –c). New multibeam mapping by LINZ in 2021 has revealed further small fields around the Cow and Calf Islands, and Rangipukea Island. Further fields are highly likely to exist around the Happy Jack Islands to the north, and off the Deadman's Point area, to the south. Full multibeam sonar habitat mapping of these areas, processed for *Galeolaria* tubeworm reefs and used to extend the reef maps being currently developed, would be valuable. Such an overall map coverage would be quite likely to encompass the full extent of *Galeolaria* biogenic reef fields for the gulf.
- The likely to be highly biodiverse species assemblages that are associated with *Galeolaria* reefs in the gulf should be surveyed and spatially inventoried.
- Investigation of death assemblages to assess whether wide-spread spatial loss of these reefs have occurred in the gulf. Potential likely areas of loss include around the Noises Islands, and much of Tamaki Strait and Ponui Channel.

4.2.10 Macroalgae (kelps and other seaweeds) as forests and meadows

In New Zealand, kelp forests are the dominant biogenic habitat on shallow rocky reefs, from the low intertidal to 25 m water depths or more, depending on water clarity levels. Macro-algal beds (primarily red and green algae) can form extensive meadows over soft sediment areas.

Historical distribution

No detailed past distribution maps are available.

Present distribution

Dominant kelp forest forming coastal species in the gulf include *Carpophyllum* spp., *Sarg*assum spp., and *Ecklonia radiata*, with *Lessonia* spp. appearing in more exposed areas offshore. These can occur as both mixed forest and mono-specific stands, though often there is zonation with depth. Kelp forest occupies much narrower depth bands in the inner gulf, related to higher levels of suspended sediments, higher turbidity, and reduced available light regimes. In the mid-gulf region, shallow reefs are well known for the occurrence of kina barrens that are considered to have replaced kelp forests following large reductions in the predators that once limited their numbers (larger snapper, and crayfish) (Babcock et al. 1999).

Many smaller brown macroalgae, as well as a wide range of red and green macroalgae also occur in the gulf, both associated with rocky reefs and soft sediment habitats. Soft sediment algal meadows have not been recorded from the Hauraki Gulf proper, but on the more offshore islands of Great Barrier Island and the eastern side of Great Mercury Island, such meadows do occur, including *Caulerpa* spp. (M. Morrison, NIWA, pers. obs.). Such habitats may have once existed in Tamaki Strait.

Services and value of habitat

- Kelp forests and algal meadows are photosynthetic and can be important primary producers.
- Modify water flow and sediment regimes. Algal meadows on soft sediments can stabilise sediments by trapping fine sediments, and buried rhizomes (roots) can consolidate soft sediment (e.g., green algae *Caulerpa flexilis* stolon growth).
- Provide living structure to a wide range of flora and fauna and provide habitat, food, and refuge for fish and invertebrate species. This includes valued fisheries species.
- Fragmenting/drift macroalgae may be buried in soft sediments, or exported to deep waters, acting as a carbon sink, and drift algae including beach-cast material provides habitat and food for small invertebrates and birds.

Recovery and restoration potential

Two specific current research projects on Hauraki Gulf kelp forests, and giant kelp in the South Island, are covered in Section 5.1.

The issue of urchin barrens replacing brown kelp forests on rocky reefs is a clear example of where human intervention to mitigate a problem created by human activities is needed.

More broadly, more knowledge of what has been lost from the gulf is needed before restoration activities can be considered.

Knowledge gaps

- While Although there are many knowledge gaps for brown algae, a central one is to what extent brown algal forests contribute to fisheries species production. Fisheries species utilise these habitats, but no attempt has yet been made to link this habitat usage to fisheries productivity outcomes. Work is needed to assess how selected brown algae species (e.g., *Ecklonia* and *Carpophyllum* spp.) may support selected fisheries species production, through functions such as providing nursery habitat and foraging.

4.2.11 Other species groups: corals, sea pens, bryozoans, oysters

There are several other habitat-forming species groups that also occur in the Hauraki Gulf, but which are either restricted in distribution, or have only been found in low numbers of individuals (do not occur in high enough densities to be denoted as biogenic habitat). Some of these may have once made more significant contributions to the gulf biogenic habitats as components of fishers 'live ground'. These are briefly discussed below for completeness, noting that as for *G. hystrix* mound fields discovered in 2019, such habitat areas may yet be discovered in the gulf.

Rhodoliths. These are free-living calcareous red algae, with two species commonly found in New Zealand (*Sporolithon durum, Lithothamnion crispatum*) (Harvey et al. 2005, Farr et al. 2009). Rhodoliths can occur as dense beds, scattered small drifts among shelly seafloors (e.g., in the shallow areas between Coromandel Peninsula and Great Mercury Island), and as occasional individuals in coarse sediments associated with reefs. They are long-lived and are sensitive to sedimentation and physical disturbance (Nelson 2009). They often support diverse assemblages of small invertebrates, as well as small attached red macroalgae species. Known distribution in the gulf includes beds on either side of Motuihe Channel (just off Motutapu and Motuihe islands), around the Noises Islands, on Albert Shoal and around Beehive Island and Passage Reef in southern Kawau Bay, off the north side of Whangaparāoa Peninsula, and a large bed off the southern side of Little Barrier Island.

Sea pens. Seldom seen in the Gulf, these are likely to have been present in deeper water before the advent of modern-day commercial trawling; populations may exist offshore in deeper waters. One solitary individual was caught in a grab in the mid gulf in 2002 (Morrison et al. 2003). There are no records of sea pens from the gulf lodged in the OBIS-NZ, NIWA-Inverts, and VME databases (e.g., see figure 3–65 of Anderson et al. 2019). There are currently 31 species of sea pens known from New Zealand waters.

Corals. Black corals (species not known) have been captured/observed by video on some of the deeper reefs of the gulf. Deep rocky reefs are poorly researched, so unknown populations are likely in the gulf, around the Mokohinau, Arid, Cuvier, Mercury, and Alderman islands. Small solitary cup corals (*Flabellum* spp.) can be found on both rocky reefs and in some soft sediment, dead shell habitats, but their small size and low densities do not qualify them as biogenic habitat formers.

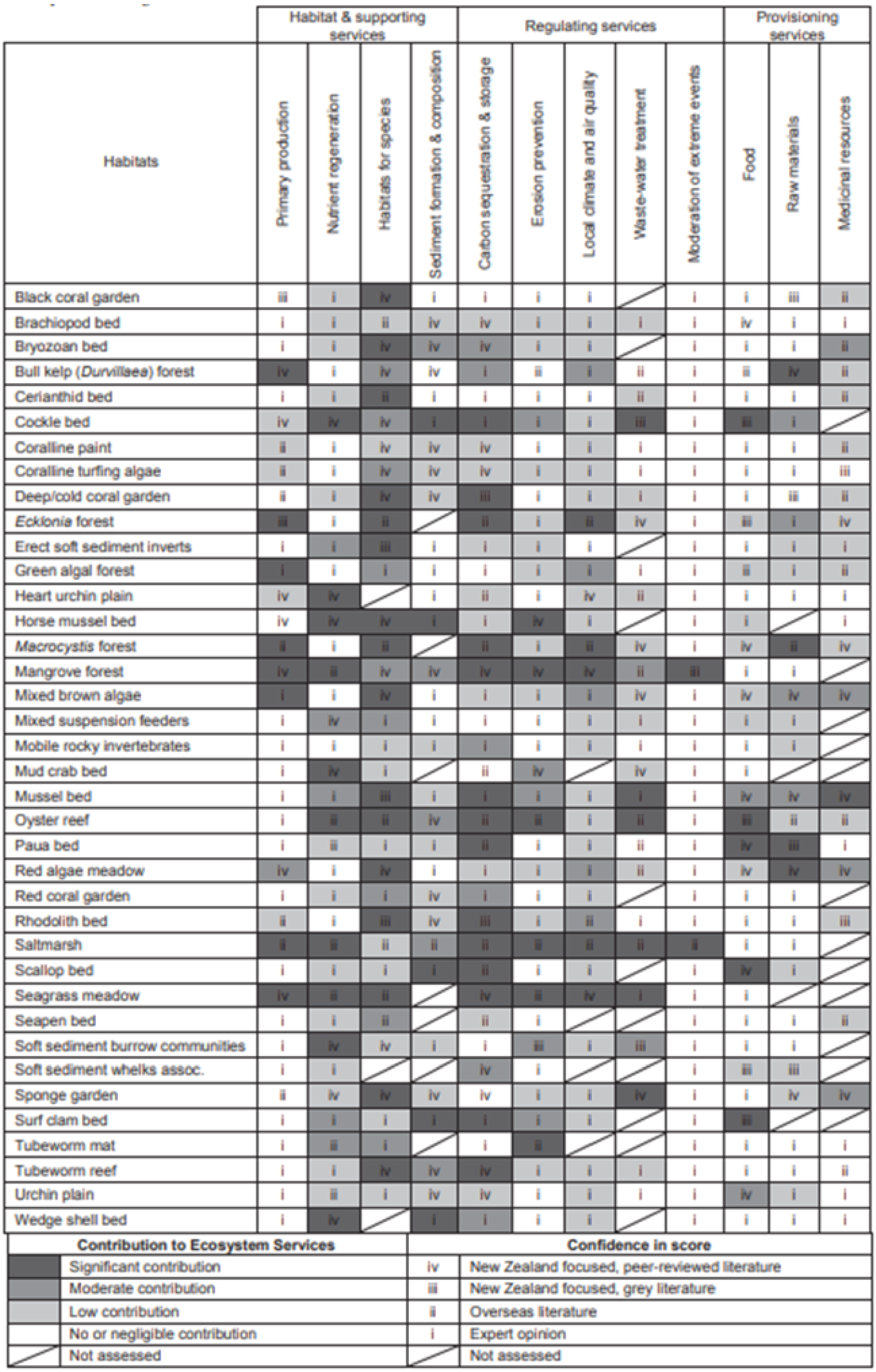
Bryozoans. Many species of this diverse group occur in the gulf, including 'large frame builders' (Wood et al. 2012), but only as individuals contributing to wider invertebrate assemblages. However, large species such as *Celleporina agglutinans*, which can form large mounds and stacks, are present; in other regions of New Zealand, they can create extensive true biogenic reefs that greatly enhance biodiversity and seafloor secondary productivity, as well as providing nursery habitats for fish. Such habitats may yet be discovered in deeper water (> 30 metres), such as around Great Barrier Island, Horn Rock, and other locations where shelter from the open ocean is available. Bryozoans are very likely to have also been a significant contributor to the live ground mentioned by fishers.

Dredge oysters. Greenway (1969) clearly identified an area of dredge oysters in the Firth of Thames, showing them as a discrete polygon in his green-lipped mussel distribution map. This species is still present in the gulf today, though in low densities and at sizes smaller than those seen in Foveaux Strait.

4.2.12 Summary tables

Geange et al. (2019) provided a summary of the relative importance of different habitats for potentially creating ecosystem services. Many of the species/groups in that summary are included in this report. This summary is reproduced here (Table 10) to provide further background on these species groups. Table 11 provides a summary of past and present distributions of possible species/species groups (as discussed above) for restoration.

Table 10: Copy of figure 2 of Geange et al. (2019). A New Zealand explicit view of the relative importance of habitats to Ecosystem Service Potential (ESP). Cell shading indicates the relative contribution to ES potential, with roman numerals specifying the supporting evidence. Scoring assumes that habitats are in a good state of health. The matrix can be read horizontally to observe the mix of ES that a habitat contributes to, or vertically to identify which habitats contribute to a specific ES. Cells with diagonal lines indicate that they could not be assessed due to lack of available literature and expert knowledge. Asterisks have been added to indicate habitats covered in this report for the gulf.



		Habitat & supporting services				Regulating services					Provisioning services		
	Habitats	Primary production	Nutrient regeneration	Habitats for species	Sediment formation & composition	Carbon sequestation & storage	Erosion prevention	Local climate and air quality	Waste-water treatment	Moderation of extreme events	Food	Raw materials	Medicinal resources
*	Black coral garden	iii	i	iv	i	i	i	i	╱	i	i	iii	ii
	Brachiopod bed	i	i	ii	iv	iv	i	i	i	i	iv	i	i
*	Bryozoan bed	i	i	iv	iv	iv	i	i	╱	i	i	i	ii
	Bull kelp (*Durvillaea*) forest	iv	i	iv	iv	i	ii	i	ii	i	ii	iv	ii
	Cerianthid bed	i	i	ii	i	i	i	i	ii	i	i	i	ii
*	Cockle bed	iv	iv	iv	i	i	i	i	iii	i	iii	i	╱
	Coralline paint	ii	i	iv	iv	iv	i	i	i	i	i	i	ii
	Coralline turfing algae	ii	i	iv	iv	iv	i	i	i	i	i	i	iii
	Deep/cold coral garden	ii	i	iv	iv	iii	i	i	i	i	i	iii	ii
*	*Ecklonia* forest	iii	i	ii	╱	ii	i	ii	iv	i	iii	i	iv
*	Erect soft sediment inverts	i	i	iii	i	i	i	i	╱	i	i	i	i
*	Green algal forest	i	i	i	i	i	i	i	i	i	ii	i	ii
	Heart urchin plain	iv	iv	╱	i	ii	i	iv	ii	i	i	i	i
*	Horse mussel bed	iv	iv	iv	i	i	iv	i	╱	i	i	╱	i
	Macrocystis forest	ii	i	ii	╱	ii	i	ii	iv	i	iv	ii	iv
*	Mangrove forest	iv	ii	iv	iv	iv	iv	iv	ii	iii	i	i	╱
*	Mixed brown algae	i	i	iv	i	i	i	i	iv	i	iv	iv	iv
*	Mixed suspension feeders	i	iv	i	i	i	i	i	i	i	i	i	╱
	Mobile rocky invertebrates	i	i	i	i	i	i	i	i	i	i	i	╱
	Mud crab bed	i	iv	i	╱	ii	iv	╱	iv	i	i	╱	╱
*	Mussel bed	i	i	iii	i	i	i	i	i	i	iv	iv	iv
*	Oyster reef	i	ii	ii	iv	ii	ii	i	ii	i	iii	ii	ii
	Paua bed	i	ii	i	i	ii	i	i	ii	i	iv	iii	i
*	Red algae meadow	iv	i	iv	i	i	i	i	ii	i	iv	iv	iv
	Red coral garden	i	i	i	iv	i	i	i	╱	i	i	i	╱
*	Rhodolith bed	ii	i	iii	iv	iii	i	ii	i	i	i	i	iii
*	Saltmarsh	ii	ii	ii	ii	ii	ii	ii	ii	ii	i	i	╱
*	Scallop bed	i	i	i	i	ii	i	i	╱	i	iv	i	╱
*	Seagrass meadow	iv	ii	ii	╱	iv	ii	iv	i	i	i	╱	╱
*	Seapen bed	i	i	ii	╱	ii	i	╱	╱	i	i	i	ii
	Soft sediment burrow communities	i	iv	iv	i	i	iii	i	iii	i	i	i	╱
	Soft sediment whelks assoc.	i	i	╱	╱	iv	i	╱	╱	i	iii	iii	╱
*	Sponge garden	ii	iv	iv	iv	iv	i	i	iv	i	i	iv	iv
	Surf clam bed	i	i	i	i	i	i	i	╱	i	iii	╱	╱
	Tubeworm mat	i	ii	i	╱	i	ii	╱	╱	i	i	i	i
*	Tubeworm reef	i	i	iv	iv	iv	i	i	i	i	i	i	ii
	Urchin plain	i	ii	i	iv	iv	i	i	i	i	iv	i	i
	Wedge shell bed	i	iv	╱	i	i	i	i	╱	i	i	i	i

Contribution to Ecosystem Services		Confidence in score	
■	Significant contribution	iv	New Zealand focused, peer-reviewed literature
▩	Moderate contribution	iii	New Zealand focused, grey literature
░	Low contribution	ii	Overseas literature
□	No or negligible contribution	i	Expert opinion
╱	Not assessed	╱	Not assessed

Table 11: Summary of past and present distributions of possible species/species groups for restoration. (Continued on next two pages)

Species	Historical distribution	Present distribution
Salt marsh	Probably once widespread inside sheltered estuaries and tidal waterways.	Loss in the Waikato region has been estimated at 93% (C. Lundquist NIWA, pers. comm.). For the Auckland region the loss is likely to have been even higher.
Mangroves	Historically much more restricted in local distributions and extents.	Large increases in spatial extents. For the Auckland region, mangrove area increased from 2313 ha in 1940 to 10 483 ha in 2014 (on average 3.2% yr^{-1}), with higher rates in small estuaries (<1000 ha) than in large estuaries (>1000 ha) (Suyadi et al. 2019).
Seagrass	Noted as being widespread in the Waitematā Harbour and Tamaki Strait area by Powell (1936), who expressed concern that it was disappearing. The intertidal versus subtidal distinction was not made. Historical distribution and abundance maps do not exist.	Seagrass meadows have until recently been uncommon in the gulf, but intertidal beds have become established and have expanded over the last decade. Subtidal seagrass is still rare and functionally extinct in terms of roles such as providing important juvenile fish nurseries. Subtidal beds exist at Great Mercury Island and Slipper Island, off East Coromandel (Schwarz et al. 2006).
Green-lipped mussels	Mussel beds once covered around 500 km^2 of the inner gulf and Firth of Thames (broadly mapped by Greenway 1969).	The historically extensive beds were fished to complete loss from the 1920s to the 1960s and have not returned since that time. Small relic patches may exist but have not been found in targeted acoustic and towed camera searches (Morrison et al. 2002, 2003), nor in multibeam sonar surveys (Morrison et al, in review a–c) or from numerous beam trawl stations (Morrison et al. 2019, Morrison, NIWA, unpubl. data). A small natural bed occurred until recently in Okiwa Harbour, west Great Barrier Island (McLeod 2009).
Scallops	Death assemblages suggest that scallop beds were once much more widespread in the shallow coastal areas of the gulf. Historical distribution and abundance maps do not exist.	Scallop populations are currently at low biomass levels and are not currently open to commercial fishing. Some beds historically fished have not held fishable populations for decades, e.g., the Hooks Bay bed area off the northeastern end of Waiheke Island.
Horse mussels	Horse mussels were historically removed from some commercially fished areas due to the damage they did to trawl nets composed of cotton and other materials. Historical distribution and abundance maps do not exist.	Horse mussel beds are thought to have declined across the gulf over the last decade or so, based on observations from multiple coastal ecologists. The reasons are unknown. Recent multibeam surveys by LINZ (via subcontractors) and NIWA have identified present day horse mussel beds in some areas. These include around the islands south of Kawau Island, off the islands east of Waiheke (Pakatoa–Ponui–Tarakihi), and around Motumorirau Island. The east of Waiheke Island and Motumorirau Island locations horse mussel beds support diverse associated epifaunal assemblages, especially of sponges. A low-density bed also occurs in the northern part of Fitzroy Harbour, Great Barrier Island (Morrison et al. in review, a, b, c).

Species	Historical distribution	Present distribution
Infaunal bivalves	Death assemblages of cockle shells, often buried under a foot or more of mud in estuaries, suggest that some species were once much more abundant and widespread. Historical distribution and abundance maps do not exist.	Monitoring of selected beaches that support/have supported recreational gathering shows population to not recover to higher densities unless harvesting is halted. These survey locations were selected to cover recreational harvesting since the late 1980s; they were not selected to be representative of the historical distribution of these species.
Sponges	Historical distribution and abundance maps do not exist.	Limited descriptions of very small areas exist, e.g., the 'Sponge Garden' inside the Leigh Marine Reserve, and on deep reefs off the east side of Great Barrier Island.
Galeolaria tubeworm biogenic reefs	Historical distribution and abundance maps do not exist.	This habitat type (actual biogenic reefs) was discovered in late 2019 around Pakatoa and Motumorirau islands. Previously, such reefs were only known from the South Island (Marlborough Sounds and Stewart Island). LINZ multibeam sonar mapping in early 2021 off Coromandel Harbour, including around the Cow and Calf Islands, and Rangipukea Island, has revealed additional new (albeit small) likely fields. More tubeworm fields are likely to exist around the islands that form the Motukawao Islands /Happy Jack Islands, just to the north, and around Deadman's Point, just to the south. A healthy and epifaunally diverse *Galeolaria* field was observed by divers off Moturua Island in 2021 (pers. comm. Shane Kelly, Coast & Catchment Ltd; Shaun Lee, independent blogger), one of the Motukawao Islands.
Kelp forests and macroalgal meadows	Historical distribution and abundance maps do not exist.	No systematic maps of these habitats exist, but there is a range of shallow water reef sites (<15 m) that have been surveyed for a range of research and monitoring projects. The Leigh Marine Reserve kelp forests have been mapped.
Rhodoliths	Historical distribution and abundance maps do not exist.	Current day beds are known from Kawau Bay (Iris Shoal, Pembles Island, Elizabeth Reef), either side of the Motuihe Channel, and as small drifts between Great Mercury Island and the Opito Peninsula. A large extensive bed runs along the southern side of Little Barrier Island.
Sea pens	Historical distribution and abundance maps do not exist.	No information.
Corals	Historical distribution and abundance maps do not exist.	Black corals have been observed by video on deep reefs (> 60 metres) in the outer Gulf, around Little and Great Barrier islands, and the Mokohinau Islands.

Species	Historical distribution	Present distribution
Bryozoans	Unknown, but may have been a component of the ‘live ground’ mentioned by retired commercial fishers. Historical distribution and abundance maps do not exist.	No bryozoan biogenic reefs are known from the gulf, but reef building species such as *Celleporina agglutins* are present in the gulf. Undiscovered biogenic reefs may exist beyond normal diving depths (> 30 m) in areas not heavily fished (e.g., off the east coast of Great Barrier Island).
Dredge oysters	A large patch in Firth of Thames was reported by Greenway (1969).	Unknown. A population exists on the wreck of the *Rewa* adjacent to Moturekareka Island, south of Kawau Island.

5. GENERAL KNOWLEDGE GAPS AND CHALLENGES

Despite being arguably one of the most researched and managed coastal marine areas in New Zealand, there are still may gaps in our knowledge of the Hauraki Gulf ecosystem. Several identified during this review are particularly relevant to potential restoration efforts for the gulf.

A key point to make repeatedly is that for anything being considered for restoration, it is necessary to understand well the reason/s for past declines before any restoration is attempted. If the mechanism/s of decline is still operating in the present day, then any restoration attempts are likely to prove futile. An example would be trying to restore filter feeders or macroalgae in waters with high levels of suspended sediments. The issue of high suspended sediment loads would need to be resolved prior to any active restoration attempts.

5.1 What are the habitats and habitat landscapes of Hauraki Gulf, past and present?

Most of the seafloor has not been formally mapped using remote sensing techniques, and there are no systematic habitat landscape maps nor inventories. This is true for both the present day, and historically before major human impacts occurred. Without knowing what was out there in the past, and how much of that is left today, it is problematic to set effective restoration targets, such as which habitats should be restored, where, and how much? Our mapping knowledge is slowly increasing in a piecemeal fashion, but the fact that significant 'new' biogenic habitats can be still be discovered in even shallow dive-able depths (e.g., *G. hystrix* tubeworm mounds in two separate areas, in 12–20 m water depth, in the gulf proper in 2019) suggests there are more to uncover. Systematic targeted mapping of areas likely to hold significant biogenic habitats would address this gap.

Similarly, our knowledge of which habitats have been lost, and where, is very poor. For example, Powell (1936) commented on the large-scale loss of seagrass in the Waitematā Harbour and adjacent coast but provided no maps or areal estimates. Many habitat areas were probably lost without record in the early days of Auckland and other smaller coastal settlements, through large-scale sediment runoff from land clearance and development, and the development of large-scale commercial fishing. This information gap could be addressed in part through the strategic spatial investigation of invertebrate 'death assemblages' left in the seafloor sediments, as well as interviewing and learning from older generations of fishers about what they once observed in the gulf during their operations there. Jones et al. (2016) interviewed retired fishers (mainly trawling) around New Zealand to help identify areas of biogenic habitat, including in the gulf. However, the national scale of that work meant that no area was exhaustively covered, so there is still more to learn about the gulf (and other areas) from trawl fishers, past and present. In addition, other fishery method types (e.g., scallop dredging, longlining, and dropper fishing) were not covered in that work and remain as largely untapped knowledge sources (noting that as human generations pass, such knowledge is irrevocably lost if not recorded).

The use of predictive habitat models using the knowledge gained might also help fill in the large spatial gaps that will still exist (albeit smaller and more constrained) after such efforts. For example, biogenic horse mussel reefs formed by *Modiolus modiolus* in European waters are important habitats under threat, and predictive modelling has been used to predict their occurrence and spatial connectivity (e.g., Elasber et al. 2013, Gormley et al. 2013). Such new knowledge would help to better direct overall restoration strategies for the gulf, including setting clearer targets about what to restore, where, and how much.

In terms of present-day habitat distribution and landscapes, there are several research programmes and projects that are working (directly or indirectly) to fill in some of these knowledge gaps. These include (but are not limited to):

- NIWA's MBIE CO1X0608 programme 'Juvenile fish habitat bottlenecks', which includes work on the life history of juvenile 0+ snapper (fish <1 year old, < 100 mm in length) in the Hauraki Gulf and East Northland. That work has included extensive fish-habitat sampling across the gulf for water depths of 0 to 30 metres, which identified a range of soft-sediment biogenic habitat locations (e.g., horse mussels, sponges, algal meadows, dog cockles, and *Galeolaria* (tubeworm) calcareous biogenic reefs).

- Habitat focused multibeam sonar seafloor mapping. The Foundation North GIFT fund for the Hauraki Gulf has resourced (with support from Auckland Council, Waikato Regional Council, and NIWA) the seafloor mapping of key juvenile snapper and/or biodiversity areas. These are Port Fitzroy (Great Barrier Island), the high biodiversity area east of Waikehe Island on the edge of the Firth of Thames, and the Waimate Bay area on the west Coromandel coast. Ground-truthing was completed using towed video (funded by the Bottlenecks programme). A key discovery from this has been the existence of fields of *Galeolaria* tubeworm biogenic reefs on both sides of the firth.
- Navigational focused multibeam sonar seafloor mapping. LINZ funds bathymetric surveys to assist with safe navigation; though these are not focused on ecological habitats per se, the information can be used to do so. A recent LINZ project has mapped much of the inner gulf using multibeam sonar, including the Kawau Bay region, and from Whangaparāoa Peninsula through East Coast Bays to Tamaki Strait. Although back-scatter was collected by default, LINZ does not include its output in navigational projects; however, in this case Auckland Council funded the multibeam provider to fully process the backscatter, to make it available for use in habitat studies. A current LINZ 2020/21 project is mapping large blocks on either side of the Coromandel Peninsula, which on the western side extends from south of Manaia Harbour, to just north of the Happy Jack Islands. Based on the discovery of the *Galeolaria* biogenic reefs around either side of Motumorirau Island just south of the Happy Jack Islands, it seems highly likely that new *Galeolaria* fields will be found on the flanks of islands and the submarine banks to be mapped. Work in NIWA's Bottlenecks programme is advancing approaches to extract maps of these reefs from the Pakatoa and Motumorirau islands multibeam data (where they are clearly visible as raised structure); this approach will be applied to new multibeam sonar data nominally available in 2021.
- DOC is funding a 2021 project looking at the distribution and ecological assemblages of deep rocky and biogenic reefs (50–300 m water depth) around New Zealand. As a desk-top study only, it is using a range of data sources to identify where such reefs may be present, including in the Hauraki Gulf.
- Auckland Council has an ongoing Marine Habitats programme that has included past ecological survey work on Kawau Bay and Tamaki Strait. Although that programme does not involve remote sensing mapping (e.g., multibeam sonar) it does create knowledge on species distributions and assemblages. Waikato Regional Council has similar ongoing work programmes in its area, including the identification of Marine Key Ecological Areas (e.g., Lundquist et al. 2020).
- A current desktop Fisheries New Zealand project being undertaken by NIWA will model the distribution of biogenic habitats in the gulf, and where they might be restored in the future, to help increase juvenile fish production (see Bottlenecks programme above). A focus is on the new MPAs or MPA extensions being advanced in the Hauraki Gulf Sea-change Marine Spatial Plan, and their value in protecting juvenile fish habitats (Note: those MPAs have been advanced to protect marine biodiversity; their planning has not specifically considered their potential role in fisheries management of the gulf).
- Work by Auckland University and local iwi, funded by the Foundation North GIFT fund, is mapping some of the urchin barrens on the shallow rocky reefs of the gulf (Nick Shears, Auckland University, pers. comm.).

5.2 Ecological connectivity, including the likely role of brood-stock reserves

Most biogenic habitat-forming species (and more broadly, most species) distribute themselves spatially through larval transport, with a few exceptions, such as the largely vegetative clonal strategy used by New Zealand's *Z. muelleri* seagrass. Although various current circulation models exist for some areas of the gulf (Hadfield et al. 2018), little is known about larval transport spatial and temporal dynamics, and how different areas are connected to each other around the gulf. As discussed in the scallop section, this may involve source and sink networks, where some localised populations essentially support other populations with their larval exports. Improved knowledge of the connectivity of certain species may inform restoration efforts of species in areas that are most likely to support wider population recoveries in the gulf, as well as creating self-sustaining populations more resilient to new threats and stressors. As well as actively restoring some areas, such knowledge would also allow for important areas that still hold populations to be set aside as important ecological areas, as a passive restoration management approach.

Along with this, the quantification of how density levels affect the spawning success of sessile invertebrates that form dense beds (e.g., scallops and horse mussels) would be very useful. As discussed in the scallop section, high density beds are likely to produce most reproductive outputs (competent larvae), with more widespread but low-density individuals potentially being reproductively irrelevant. Currently, no management actions are taken to protect some of the high-density beds from being intensively fished (scallops) or incidentally damaged (e.g., horse mussels). If clear density-dependent reproductive success is demonstrated, and with large-scale modelling can be shown to be clearly important in maintaining healthy shellfish populations, then it would be logical to set aside brood-stock reserves, embedded within the source-sink dynamics likely to exist in the gulf.

5.3 Research on the ecological requirements of habitats/species targeted in restoration efforts

For many species that might be considered for restoration, there are large fundamental knowledge gaps on their basic life history and habitat requirements. For instance, the early life history of New Zealand's horse mussel species is effectively unknown. This includes how stressors such as sedimentation may interfere with, or eliminate, the ability of planktonic larvae to settle to the seafloor successfully and generate new mussel beds. In assessing whether restoration is viable for a given species in a given area, it is necessary to know the ecological requirements of that species, and if the potential restoration area can meet those requirements (or be improved over time to move towards meeting them). A much greater knowledge of the basic ecology of many species is needed. This not only includes for restoration work, but also for wider human management of New Zealand's marine ecosystems, such as for protecting biodiversity and ecosystem-based fisheries management.

5.4 Successful restoration methods for Hauraki Gulf (and New Zealand more widely)

The literature on marine species restoration in New Zealand is very limited and effectively restricted to trials with intertidal seagrass (Matheson et al. 2015, unpubl. data), cockles (Cummings et al 2007), arguably scallops but from a commercial enhancement perspective, and recent green-lipped mussel work (Wilcox & Jeffs 2018, 2019). Apart from scallop enhancement (which is not restoration *per se*), which worked for a time in the Nelson region and then failed for unknown reasons, marine restoration work in New Zealand has been small-scale and experimental research only. There are no substantive associated financial model frameworks which with to explore cost-benefit assessments, or to use to make funding cases to potential sponsors, either commercial or environmental. Basic research on successful transplantation approaches, and ways to improve larval settlement and recruitment to seafloor habitats, remain a fundamental need.

Current innovative approaches

There are several current research projects underway in New Zealand on habitat restoration.

Giant kelp (Macrocystis pyrifera) genetics (Chris Hepburn, Matthew Desmond et al., Otago University)

Kelp-forest habitats are in decline along New Zealand's coastlines, echoing global patterns of biodiversity loss. A key symptom of kelp-forest decline is a collapse of valuable coastal fisheries that are reliant on the services they provide. Predicted increases in sea surface temperature, a major driver of decline, threatens remaining kelp-forests and restoration using climate change resilient strains is required to preserve and stimulate fisheries value (pāua, lobster, finfish) in a changing ocean. This project aims to develop a low-cost, self-sustaining bottom-up approach to inshore fisheries restoration by rebuilding *M. pyrifera* forests (Figure 36) that provide the foundation of many coastal fisheries. This project will first examine the genotypic diversity of *M. pyrifera* from across its New Zealand range and use physiological experiments to select climate change resilient strains for reseeding. These strains will be cultured en masse and reseeding techniques will be trialled to determine the most successful approach. Large scale reseeding will then be undertaken, and the success of reseeding and recovery of associated fish stocks will be examined to validate methodology and estimate the value of restoration. This will be the first attempt of habitat restoration in New Zealand using a forward-looking approach where genetic selection is incorporated to promote climate change resilience. This project aims to establish a best-practice approach to kelp forest restoration in New Zealand/Aotearoa, by selecting for, cultivating, and reseeding the most resilient strains of *M. pyrifera* to combat the inevitable effects of climate change. Although giant kelp does not occur in the gulf (nor northern New Zealand), if this approach proves successful, it might be expanded to other species where climate stressors prove to be a significant issue.

Figure 36: Butterfish (*Odax pullus*) browsing in a giant kelp forest, southern New Zealand. Source: Matthew Desmond, University of Otago.

Urchin barrens in the Hauraki Gulf (Nick Shears et al., University of Auckland)

A project looking at kelp restoration (restoring kelp forests in urchin barrens) in the gulf is focused on Little Barrier Island (Hauturu), Leigh, and the Noises Islands, working with iwi. Initial results show that kina barrens are extensive on exposed coasts and the islands of the gulf, typically covering about half of the shallow reef (<15 m). Previously, the researchers have shown that small-scale removal of kina from barrens promotes recovery of seaweed habitats including kelp forests. This project is looking at the practicality and potential of larger-scale kina removal as a tool for kelp forest restoration. The researchers note that it is important to recognise that this is only part of the process and that it manages the symptom rather than the problem (Nick Shears, University of Auckland, pers. comm.).

5.5 Other considerations for restoration

Large-scale marine habitat restoration in New Zealand is just starting to be explored as a possible path. Recent trials of bulk drops of farmed green-lipped mussels to the seafloor as a first initial attempt to restore historical beds has revealed some of the regulatory considerations involved. Those drops were done without Auckland Council resource consents, but, for ongoing work, resource consent has now been granted. This is important, because regulatory agencies have a key role to play in determining what society wants for the marine environment, especially where habitat creation rather than habitat restoration is proposed (e.g., in creating green-lipped mussel beds in Mahurangi Harbour), or in potential scallop enhancement of seafloor areas to increase fisheries yields.

For the green-lipped mussel restoration trials, biosecurity issues were also encountered, with MPI raising concerns about the transport of unwanted NIS organisms around the gulf with the adult mussels taken from aquaculture farms for transplanting and requiring suitable cleaning treatment of these mussels before allowing their movement. Such treatments (as currently exist) are very expensive per unit of mussel and considered beyond the means of the ‘Revive our Gulf’ group (and for NGOs more widely). The counter-argument was made that the NIS organisms discussed were widespread and well-established in the gulf already, and present on both the donor farms and the transplant sites (e.g., Mediterranean fan worm). This is true but does not account for the possibility that a significant new NIS organism could be introduced in the gulf at any time, and then be advantaged in its spread by mussel restoration transfers. Ideally, the distance mussels are transported from donor to recipient sites should be minimised.

Protection of transplanted plants and animals once transplanted is also not possible without invoking regulatory protection. One approach is to seek a suitable resource consent (which must be renewed every five years) and use it for up to the maximum period of 25 years. Then, if restoration has been successful and the habitat has been restored, it is possible to ask regional councils to consider bringing these areas into their unitary plans as ‘Special Ecological Areas’ that will give them long-term protection, assuming they are self-sustaining. More broadly, changes in spatial area use can also be dealt with under unitary plans by regional councils. An example in the Auckland region was the removal of boat moorings from Ōkahu Bay, as a site especially important for Ngāti Whātua o Ōrākei. Further afield, the recent court case on the closing of areas around Motiti Island, off Tauranga, to protect key areas from fishing impacts, is also changing the regulatory landscape (as enacted), with regional councils now possibly able to control fishing activities within the 12 n. mile limit, where they cause significant damage to the natural values of the seafloor.

The Fisheries Act, though not currently used in this way, does allow provision for the restoration of habitats and species that have been reduced in their natural functioning by fishing impacts. There are also other fisheries regulations that have been used to close ‘Areas of particular significance to fisheries management’, such as part of Spirits Bay in Northland (from scallop dredging affecting rich epifaunal biodiversity), the Wairoa Hard in Hawkes Bay (higher epifaunal biodiversity and potential fish nurseries), and Separation Point, near Nelson (large bryozoan fields considered to act as important fish nurseries). These approaches could be used to support restoration efforts for species such as green-lipped mussels and horse mussels where fishing impacts have been key or significant, as well as set aside, from fishing, habitat areas that still exist and have important fisheries support functions, which are not currently protected by fisheries regulations (e.g., deeper water horse mussel beds that exist beyond the Hauraki Gulf ‘no trawl’ line).

6. GUIDANCE FOR PRIORITISING RESTORATION EFFORTS BASED ON AVAILABLE INFORMATION

Although there is a rapidly expanding literature on marine restoration, a great deal of it is focused on generic principles to follow, at the concept level rather than being practical and specific; these are rather repetitive. Fitzsimons et al. (2020) summarises guidance well with their 'Key guidelines for practitioners when undertaking shellfish reef restoration projects' (see Table 4).

As argued by Geist & Hawkins (2016), our ability to restore marine systems is greatest in enclosed systems where there is greater scope to intervene in water quality, ecosystem processes, and structural attributes. As we move into more open systems, which have the potential to recover more rapidly (due to inputs from other adjacent regions such as larvae and water masses), our scope for restoration becomes more limited and largely is "*limited to aiding replacement of biogenic structuring species or assemblages*" (Figure 37). Added to that is the caveat that restoration difficulty will increase with water depth, with most marine restoration globally being focused on shallow water system components (e.g., salt marsh, mangroves, seagrass, oysters, and tropical corals). With these caveats in mind, the following potential themes for restoration are suggested.

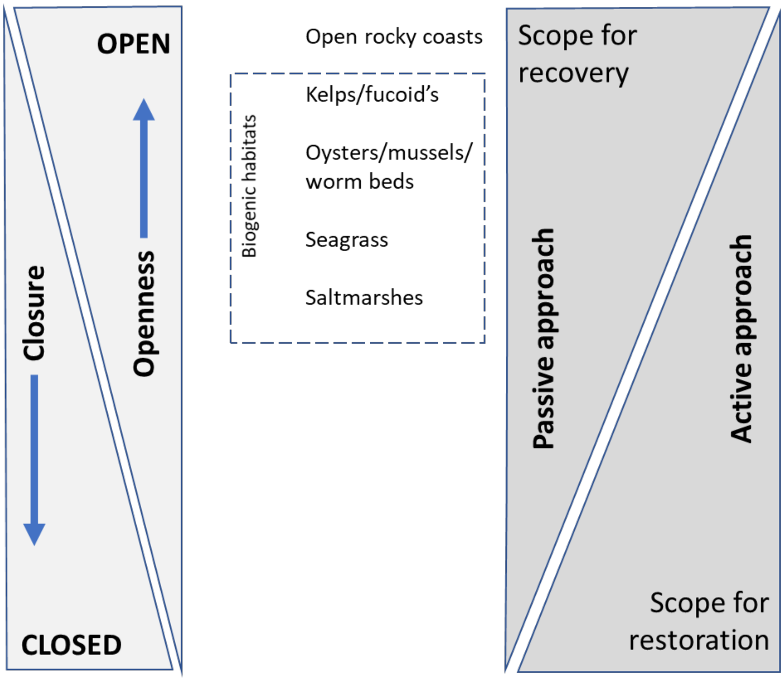


Figure 37: Openness of different aquatic ecosystems and scope for restoration and/or ability to recover naturally. Generally, more open systems can recover most rapidly; but in such systems scope for restoration is limited to aiding replacement of biogenic structuring species or assemblages. More enclosed systems present greater scope for intervention in water quality, ecosystem processes, as well as structural attributes. Modified from figure 2 of Guest & Hawkins (2016).

The greater Hauraki Gulf occupies a relatively large area, with a diversity of estuarine, coastal, and semi-oceanic environments. Although there have been fundamental changes in its species, habitats, and environmental conditions since the arrival of humans, it remains a productive ecosystem, with most of its components still present, albeit some greatly reduced in extent and abundance.

What to prioritise for restoration efforts is a question for society to debate. At the most fundamental level, for the inner and possibly mid-gulf, the best advance would be to improve water quality and associated clarity, as well as sedimentation issues on the seafloor, by reducing land-based inputs of sediments and other pollutants. Suspended and deposited sediment is a key issue, and, without ongoing improvements in these factors, detailed attempts at restoring the ecological function of the system may be futile.

The issue of scale is key in any restoration efforts for the gulf, which covers a very large area, whereas active restoration is limited to relatively small areas. The seagrass, cockle, and green-lipped mussel restoration experiments that have been done in New Zealand have been on the scale of 1s to 10s square metres. Scaling up active restoration to 10s to 100s square kilometres (many orders of magnitude increases) is a huge undertaking and is unrealistic when the cost, logistics, and the number of plant/animal individuals is calculated for such large scales. Rather, a more realistic objective is to restore small areas within the targeted overall restoration extents, and then work to allow natural processes to slowly expand these patches out into their wider historical range, while ensuring the stressors that drove their loss are no longer present (and no new stressors have appeared since). It has taken several hundred years of human activity to cause the environmental decline of the gulf; its (partial) restoration is likely to take a multi-decadal timespan to achieve. The expectations of funding agencies, and the wider public, need to be tempered to be aware of the time scales that will be required.

This report is the result of a small desktop review and does not include any new data analyses, nor new mapping synthesis work to help provide spatial analyses of where restoration might be best directed in the gulf. Such work is well beyond the scope here. However, it is noted that a new much larger Fisheries New Zealand project has been won by MIWA to look at where biogenic habitats might be best targeted for restoration in the gulf. This review informs into that new much larger desktop, spatial analyses project.

Suggested principles/questions on which stakeholders might analyse the appropriateness of active restoration proposal include:

- Did the habitat occur at the location historically?
- How much information is already known about the species? The location?
- Are current environmental conditions suitable for the species involved? This includes the removal of the stressors that caused the original species/habitat loss, where known.
- Do the locations conflict with other planned or existing human uses of the area?
- Is the restoration likely to significantly improve the ecological function and health of an area? What are the specific goals, e.g., biodiversity, fisheries, primary productivity, water quality/clarity?
- Will it provide wider benefits to other areas through processes such as source-sink dynamics?
- How are local area users likely to be affected by the restoration? Does this involve active benefits, e.g., cultural harvest?
- What types of restoration best suit the situation? Is there an existing larval/propagule supply of the target species to the restoration area?
- What are the likely relative costs of restoration (e.g., passive, versus active)?

Noting that explicit spatial analyses of where active restoration might be done are well beyond the scope of this project, several suggestions for which species and areas might be targeted for restoration are given below.

6.1 Passive restoration (here, the removal of stressors)

Subtidal seagrass

Observations suggest that intertidal seagrass has ‘returned’ to the Hauraki Gulf, especially around the Auckland region, over the last decade or so. The mechanisms for this are unknown but may be part of larger long-term abundance cycles that may exist for this species in at least northern New Zealand. However, its extension below the low tide zone, to create subtidal fringes and meadows, has not followed. These subtidal elements are of much higher value in terms of acting as high-density fish nurseries, and directly supporting higher secondary productivity (invertebrates) and biodiversity. It appears that beds around the Auckland region do show some initial extension into the subtidal zone, but this fails to establish and expand. The most likely explanation for this is insufficient light penetration through the water column to allow seagrass to be successful in the subtidal zone. Current work by Fleur Matheson (NIWA) and collaborators in Tamaki Strait on seagrass and light regimes is likely to offer key advances in our understanding of this issue. Regardless, any improvements that can be made through better land-based management of sediment run-off are likely to assist the regeneration of subtidal (and intertidal) seagrass in the Hauraki Gulf.

The 'live ground' described by fishers, and scallop and horse mussel beds

Fishing, both commercial and recreational, in tandem with land-based sedimentation effects, has greatly reduced the extent and abundance of these species and habitat-formers. Although the exact species nature of 'live ground' is not currently known, it is likely to be composed of mixed biogenic habitat assemblages of sponges, bryozoans, ascidians, hydroids, and other groups. Interviews with older and retired fishers, using good visual species guides and nautical charts, could be undertaken to better understand what this mixed biogenic type is, where it historically occurred, how it varied spatially and temporally, and what finfish fisheries species were associated with it. Targeted mapping could then look for remaining spatial remnants that could provide the starting populations for its potential regeneration. The most likely approach to facilitate that would be to close such areas to fishing gear impacts, which also fits some of the longer-term objectives of the Hauraki Gulf Marine Spatial Plan. The goal then would be to see if the natural regeneration of heavily trawled seafloor for key habitat areas occurred without a need for active restoration; although if/how fisheries would then still be able to catch the quota previously taken in these areas with trawling is an unknown. Given the likely water depth, large spatial scales, and species mixes involved, direct restoration seems unlikely for the foreseeable future. The declining growth rates of individual snapper in the gulf proper (Walsh et al 2017), as the snapper stock rebuilds, strongly supports the possibility that the carrying capacity of the gulf has diminished and is no longer as productive as it was. Insufficient food (reduced carrying capacity of the ecosystem) seems the most likely issue, though how that might divide between benthic and pelagic food resources is unclear. It seems likely that restoration of live ground, along with other restoration activities, would help mitigate this carrying capacity issue.

Scallop beds would benefit from the removal of recreational dredging in the Gulf from areas already closed to commercial dredging (e.g., Kawau Bay, and the Noises Islands), to allow for natural spat settlement surfaces (e.g., foliose red algae, hydroids, and small tubeworms) to recover and increase scallop recruitment and early juvenile survival (Talman et al. 2003). This would also allow for general biogenic habitat covers and associated biodiversity to improve. Similarly, the identification of horse mussel beds, and the closure of such horse mussel areas to bottom-contacting fishing methods, would also improve the ecological health of those areas of the Gulf.

Other biogenic habitat assemblages

According to fisher accounts, North-West Reef and the Colville Channel area between the end of Coromandel Peninsula and Channel Island were once diverse biogenic habitat areas, with large associated fish populations. For North-West Reef at least, smaller fish were also reported to be abundant, suggesting a nursery habitat role. Both areas were heavily impacted by towed wires, bobbins, and old nets to clear away the biogenic habitats (kelp forest and habitat-forming invertebrates), followed by heavy fishing pressure to harvest the finfish left without cover. Today, the actual rock stacks of North-West Reef are protected from fishing and other disturbances by the Cableway Protection Zone, but the lower relief ground extending west that probably held most of the biogenic habitat is still open to fishing and shows little sign of recovery (as observed from two short towed video transects in February 2020, M. Morrison pers. obs.). The Colville Channel area is also open to fishing at present, with part of it proposed to become an Marine Protected Area as part of the ongoing Hauraki Gulf Sea Change process.

As both are potentially quite unique biogenic habitat areas for the Gulf (and probably more widely of northeastern New Zealand), they warrant consideration for restoration. To learn more about what was historically there, fishers that participated in conditioning these grounds (who may still be alive today) could be interviewed to gather information on what species were present, and over what spatial scales. Some information may also be held in the seafloor death assemblages at these locations, especially of calcareous species such the large bryozoan biogenic reef-builder *C. agglutins*. The depth and open nature of these areas, along with the likely diversity of biogenic habitat-forming species involved, probably makes active restoration problematic. However, the closures of these areas to fishing, as the pressure that caused the original habitat changes, may allow passive restoration to occur, because all the biogenic species involved are likely to still be present in the Gulf, providing potential larvae for regeneration.

6.2 Active restoration

Subtidal seagrass

Although increasing in some areas, seagrass is still at low abundance relative to what it probably was in historical times in intertidal areas and is not returning to subtidal areas. Work by Matheson et al. (2017) has shown that (intertidal) seagrass transplantation can successfully be used to create new seagrass areas without major detriment to donor populations, and, that in turn, newly established areas can subsequently also be used without negative effect to provide transplants for new areas. As seagrass is largely clonal in its reproduction, if transplants are sourced from the local region, genetic diversity issues are not of great concern. Where present day environmental conditions permit, new areas that historically support seagrass could be targeted for restoration efforts using transplants. For shallow subtidal locations (<1 m during low tides), such work could meaningfully involve local schools and community groups, if clear guidelines and a framework of where restoration is desirable could be provided. An education campaign alongside this on the goods and services seagrass provides, and evidence that this habitat was once naturally widespread, would assist in gaining community support. Some opposition might be expected, with natural seagrass return to beaches such as Snell’s Beach and St Heliers Bay prompting requests from some residents for its removal using weed-killer or bulldozer, due to it being considered unsightly, to make the seafloor more muddy, and unpleasant to stand on when swimming.

Scallop, horse mussel, and green-lipped mussel brood stock reserve areas

Effort could be directed to restore small areas of high-density shellfish beds (assuming that the factors that led to their decline are no longer present), to act as source populations for shell larvae, which would then disperse to other beds and areas as part of a source-sink spatial framework. To achieve this, work will need to be done to quantify how density-dependence spawning success operates in these species, and how larval connectivity works across the relevant areas of the Gulf (but noting this latter task would be a very large one). This does assume that larval supply has become limiting for these species, and that the general environment remains conducive to larval settlement and establishment of new beds – both these assumptions should be experimentally tested first. For both scallops and horse mussels, this could involve the gathering of adults from wider low-density areas (whose individuals are assumed to be reproductively irrelevant) and transplanting them into small high-quality habitat areas to create dense beds. Scallops are easily to deal with, because they are easily gathered by dredging or hand-gathering and released to the seafloor to form their own dense patches (e.g., scallops released in grid patterns will subsequently move and cluster together in groups as a natural behaviour (Morrison 1999). There are also deeper water scallop populations in the Gulf (50–100 m water depth), as reported by fishers, which never reach legal sizes, have very thin shells, and poor watery roes and muscles. These could be collected by dredge and used to restore shallow water dense beds (assuming their deep-water condition is due to poor habitat conditions, rather than some other intrinsic mechanism inherent to these animals). Alternatively, scallop spat could be collected on artificial collectors, grown to small juvenile sizes (10–15 mm), and then reseeded to create high density brood-stock beds. Scallops can grow to 100 mm in eighteen months in the Gulf when living in good-quality habitats. Horse mussels would be more difficult, because being two-thirds embedded in the seafloor, they would probably have to be harvested from donor areas by hand (using divers) and then re-buried in the sediment at the recipient sites, also by hand (divers). Such labour-intensive work would be very spatially limited, and expensive.

Alternatively, new kinds of spat collectors could be developed for both scallop and horse mussels, that could be deployed in situ on the bed to be restored and left to operate without further human intervention. For horse mussels, growth rates are poorly known, but it is probably around 4 to 5 years before individuals reach a reproductive adult size.

The costs of such work are unknown. After restoration, additional costs would include effective compliance and enforcement of the brood-stock reserve areas to prevent poaching of scallops (or other edible shellfish included). Ideally, these reserves would also be closed to boat anchoring and other forms of disturbance to minimise unwanted mortalities of individuals and associated declines in density. Such efforts could be spatially packaged together with other spatial management efforts in the Gulf, including their incorporation into new marine reserves.

Although green-lipped mussels could also be used in a similar approach (if current issues preventing their successful restoration to the seafloor can be solved), an alternative approach might be to fund mussel farms to put aside some of their growing lines from harvest. These individuals could be allowed to grow larger and older, and presumably spawn multiple times, to increase spat supply to the surrounding seafloor (this assumes that a lack of spat fall supply is the problem—that should be experimentally tested first). Such an approach would probably be cheaper and involve less effort and permitting than trying to initially create seafloor brood stock areas. Once seafloor populations had recovered, and were self-sustaining, then farms could cease keeping some lines aside for spawning purposes.

Green-lipped mussel direct seafloor restoration

The work done to date on these species shows that adults transplanted from mussel farms can initially form beds/mosaics on the seafloor, and then survive for a year or more; but that the beds slowly decline with mortality, and that no new spat fall recruitment occurs into these adult beds to replenish their numbers. They do not qualify as successfully restored beds, because they do not have the ability to replenish themselves over time through new recruitment and, without ongoing human intervention, eventually disappear. As well as a lack of new recruits, mortality issues appear to include the use of poor substrates (mud seafloors), their predation by eleven-armed starfish, and disruption/loss from storm events. Finding suitable seafloor areas has also been an issue, with ‘good’ coarse sediment seafloor areas already being used by a range of sensitive species.

Lack of spat fall. Effort could be directed at developing low cost, in situ green-lipped mussel spat collectors that once deployed require no further intervention. Past spat catch working for scallop enhancement trials have shown that green-lipped mussel spat are present in numbers in both the Firth of Thames (Bartrom 1990) and the western Gulf (Morrison 1999). These new spat collectors could be deployed along with transplants of adult mussels to help increase the chances of a given bed becoming self-sustaining. Ideally multiple new bed deployments would be made, with larval connectivity being assessed to help direct where beds would be most likely to retain larvae, and/or receive them from other new beds. The potential use of mussel farms lines to increase spawned spat supply could also be incorporated.

Lack of suitable seafloor habitats not already used by other species. There are two options here. The first is to develop new ways to utilise mud seafloors, which are currently not particularly suitable for mussel beds. Adding new structural elements to these environments could be done, ideally using materials from the waste streams of other industries. As an ‘out there’ example, exotic pine (*Pinus radiata*) forests require branch trimming of younger trees to generate high quality tall trunks with few knots. Suitable, more complex trimmed branches could be collected, held underwater in mussel farms, and seeded with adult mussels to weigh down the branches (which might also develop some degree of waterlogging). If the inherent buoyancy of the wood could be sufficiently countered by mussel weight and water-logging (the latter eventually happens with natural wood delivered to marine systems, depending on species), then these branches could subsequently be seeded to the seafloor with adult mussels attached, as well as their foliose finer ends that are suited to larval mussel settlement. Suitable branch shapes could be selected (i.e., curved/forked/bent, rather than straight), so that when they settled into the mud seafloor, much of the branch would remain elevated above the mud, along with its attached mussels. By the time the branches have eventually decayed away (many years), multiple generations of mussels would have formed mounds of dead shell under the branches, allowing the population to expand out over the seafloor. Ironically, the inputs of large volumes of terrestrial debris following human clearance of the land is one theory that has been advanced as to how extensive green-lipped mussel beds first formed in the Firth of Thames and Hauraki Gulf (Paul 2012).

Waste shell from the mussel aquaculture industry (and/or the scallop fishery) could also be used to create localised mounds of shell on muddy seafloors, perhaps one or more metres high and 30 metres in diameter, on which adult mussels and spat collecting surfaces could then be deployed. Such mounds could support re-establishing mussel populations, and, presumably, their elevation and the increased current speeds formed over them, combined with wave action energy, might help keep them clearer of deposited fine sediments from the surrounding mud seafloor. Currently such waste shell accumulates as very large piles on private farmland, where it serves no purpose, beyond perhaps carbon sequestration.

The second option is to locate and utilise better transplant locations than those that have been used in the Gulf to date. Two possible areas are suggested based simply on more recent knowledge being available for these than for other areas, though others may yet be discovered as new data is gathered. Recent survey work for juvenile snapper nurseries has revealed areas of dead relic shell (green-lipped mussels, scallops, larger infaunal bivalves) along the central axis of the Firth of Thames, with few living epifaunal species (e.g., sponges, ascidians, hydroids). These extend from around 10 metres water depth to at least 30 metres or more; and though the water column is still turbid, the seafloor does not appear to be excessively muddy. This axis also runs past the large Wilson Bay mussel aquaculture area.

The second area is Tamaki Strait. Multibeam sonar mapping with associated ground-truthing has revealed large extents of dead relic surface shell in large drifts and patches, interspersed with muddy seafloor (Figure 38). Although some epifauna occasionally occur (e.g., horse mussels, sponges, wandering anemones) in some places, these are solitary and in very low densities, and absent from many sites. Some of the dead shell is from green-lipped mussels, and parts of Tamaki Strait once held green-lipped mussel beds (Figure 11) but most of the shell is derived from as yet unidentified large infaunal shellfish (possibly one or more surf clam species). Exploration by divers through and around these surface shell areas has not returned live shellfish of any species, aside from the introduced date mussel, which at times can form extensive low mounds at some sites within the strait. If green-lipped mussels can survive and function in the more turbid waters of Tamaki Strait, then there are extensive potential areas for restoration to proceed.

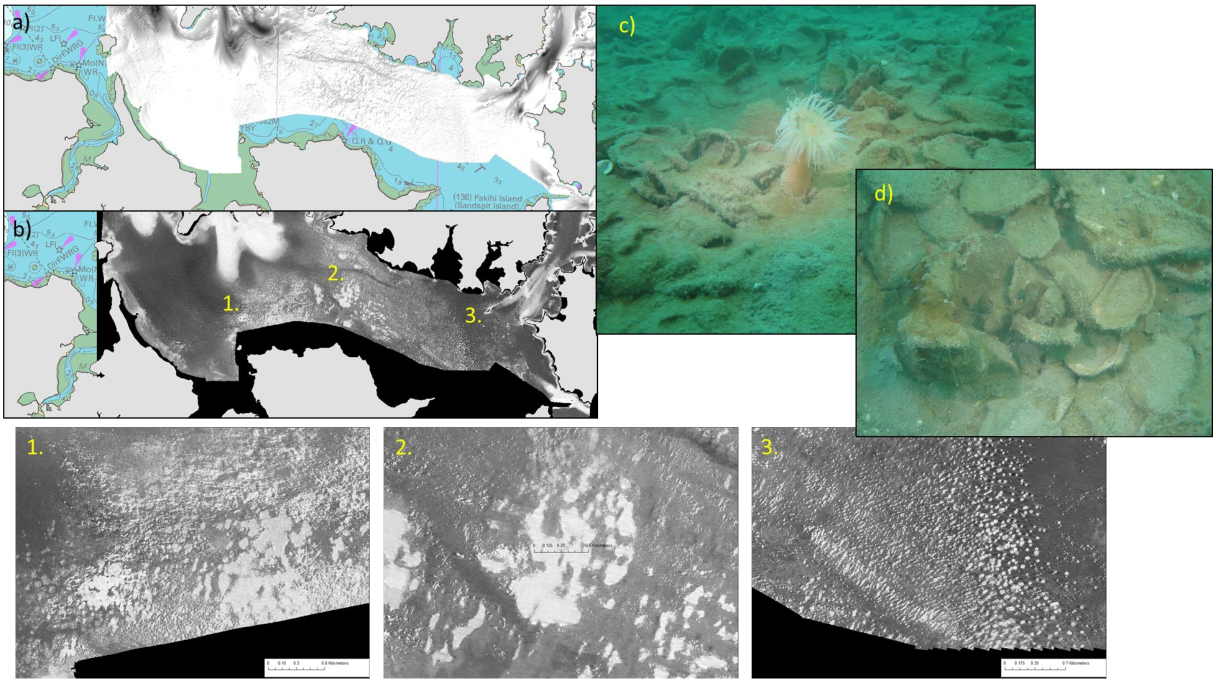


Figure 38: **The relic shellfish system of Tamaki Strait, as observed with multibeam sonar and underwater imagery. a) bathymetry of the area, showing a seafloor that is full of low height variation, as well as two small channels; b) backscatter returns, showing a very mosaicked seafloor of hard (dead shell accumulations) and soft (mud) reflectivity returns, with 1–3 showing close-ups of that variation, at the 100s–1000s metre scale; c-d) diver images of the seafloor, showing dead-shell drifts, with some silt overlay, and the occasional epifauna. Dead mussel shell can be seen in e. Source: a-b, 1–3) LINZ data, with the backscatter processed with Auckland Council support; c–d) images collected in MBIE programme ‘Marine Recreation, CO1X0506.**

6.3 Summary Tables

A brief summary of possible restoration species-area fits to suggested restoration principles or questions is given in Table 12.

Table 12: Possible restoration species-area fits to suggested restoration principles or questions. (Continued over next three pages)

Suggested restoration principles /questions	Subtidal seagrass	Live ground (from fisher accounts)	Other biogenic habitats assemblages	Scallop, horse mussel, and green-lipped mussel brood stock reserve areas	Tamaki Strait habitat landscape
Occurred historically?	Yes, but historical locations poorly/not mapped. Clear evidence of large-scale habitat loss.	Yes, but historical location/s currently unknown. Older fishers hold this knowledge. Clear evidence of large-scale habitat loss.	Yes. Two specific locations are suggested here; North-West Reef and Colville (between Channel Island and Coromandel Peninsula) – other high value biogenic habitat areas are likely to also have existed. Clear evidence of large-scale habitat loss.	Yes. Green-lipped mussel beds were mapped out for the Gulf using fisheries data and knowledge (1920s–1960s). Large beds areas were restricted to the inner Gulf and the Firth of Thames. Commercial scallop beds and some recreational fishing only beds have been surveyed using fisheries surveys. Horse mussel beds occur throughout the Gulf, from 0–100 m water depth, but have never been systematically mapped. Clear evidence of large-scale habitat loss.	Yes. Limited evidence from human account of a once very different system, dominated by possibility macroalgae, subtidal seagrass, and other species. Green-lipped beds in some areas. In-situ shellfish death assemblages, as observed by diver. Multibeam sonar mapping work shows extensive mosaics of dead relic shells on/immediately under the seafloor surface.
Information available	Significant knowledge from New Zealand research, including passive (Whangarei Harbour improved environment) and active restoration trials there and elsewhere.	Fishers need to be interviewed with suitable species imagery to quantify the species mixes this habitat comprises/ed. Limited searches in the Spark Cableway have failed to locate such habitat.	Limited indirect anecdotal accounts from retired fishers; targeted fisher interviews are needed to learn more.	All three species have had some research work; green-lipped mussels have had small-scale experimental restoration trials; scallops have had some enhancement work; horse mussels have had minimal research.	Current day ecological work has shown it to be a mud-dominated system, with some sponge gardens, rhodolith, and kelp forest habitats in the high current flow channels around Motuihe Island, on its northern side (not Tamaki Strait proper). A juvenile snapper nursery area exists on its eastern end, on mud habitats heavily pocked with larger invertebrate burrows (probably crab and shrimp species).

Current environmental conditions	Current research at Waikehe Island/Tamaki Strait is quantifying growing conditions and their effects. Other seagrass research work outside the Gulf may also provide further insights.	Fishers believe that this ground was lost due to fishing. Fishing could be removed from areas to promote passive restoration.	Fishers believe that these habitat areas were eliminated by targeted fishing. Fishing could be removed from these areas to promote passive restoration.	Green-lipped mussel beds were eliminated by fishing. Environmental changes may be preventing their recovery (heavy sedimentation of past habitat areas, loss of spat settlement surfaces. Scallop beds are still present in some areas – commercial and recreational fishing, land-based sedimentation, and loss of source-sink ‘nodes’ may be preventing their recovery. Horse mussels were removed as nuisance habitats for fishing gears in earlier decades, as well as being lost to general impacts of towed seafloor gears (trawls, dredges). Sedimentation is also likely to be having significant negative effects in some areas.	Tamaki Strait today is a muddy, turbid water coastal waterway. Research is ongoing on seagrass dynamics in this area (Matheson et al.)
Other human uses	Some areas are heavily used for swimming etc (e.g., Kohimarima Bay), but many less accessible shallow subtidal areas are available.	Occurs/occurred in deeper areas of the Gulf that are largely used for fishing only (although the Spark Communications Cable may traverse some of the area).	Recreational fishing and diving occur in these areas, although their relative remoteness and exposure to sea conditions reduces access. Restoring these habitats would not necessarily prevent these activities from continuing.	Scallops and green-lipped mussels are harvested fisheries species – though commercial mussel aquaculture has replaced wild mussel harvests. Restoration of beds for brood-stock reserve purposes would require these small areas to be protected from harvest. Improved adjacent land catchment uses to reduce land-based run-off of sediments and other pollutants would restrict some human activities on land (although many of these changes are already progressing).	Tamaki Strait is not a main navigation route for commercial shipping. Commercial fishing is limited to a small winter-only longline fishery (closed in the warmer months to prevent conflict with recreational fishing); and gill-netting for flatfish and grey mullet across its shallow estuaries, bays, and some fringing reefs (mullet only).

Ecological function improvement. Specific goals	Elevated biodiversity, greatly increased juvenile production of fisheries and other species.	Suspected to provide significant high value habitat to a range of fisheries species. May have provided nursery functions for deeper water species. Likely to have provided high biodiversity values.	Reported as high biodiversity areas, with associated high-density finfish populations.	Green-lipped mussel beds will provide biogenic habitat for many other species, increasing biodiversity, feeding grounds for adult fish, and potentially providing juvenile fish nurseries. Better water clarity and reduced nutrient levels due to their filtering activity. Horse mussel beds will provide similar functions as green-lipped mussel beds; and are a known important biogenic habitat type that supports juvenile snapper nurseries.	More productive ecosystem with increased biodiversity and fisheries production,
Wider benefits, e.g., sources-sink dynamics.	Drift seagrass may establish in other local areas.	Links through fish movements to other areas and regions; including healthier, potentially faster growing, higher quality fish.	Larvae/propagule supply to other areas of the Gulf. Potential for diver-based tourism relatively close to Auckland.	Greatly increased abundance of scallops in surrounding areas to support fisheries harvests. Improved green-lipped and horse mussel biogenic habitat creation and maintenance in surrounding areas through larval export.	Larvae/propagule supply to other areas of the Gulf.
Local user benefits	Improved fisheries.	Improved fisheries.	Improved non-commercial fishing and diving opportunities.	Improved fisheries.	Improved fisheries. Improved environmental conditions including water clarity.
Potential restoration approach	Passive – improving environmental conditions (underwater light levels, and less muddy seafloor sediments). Active – Planting of seagrass patches to promote bed formation.	Passive – remove ongoing fishing impacts.	Passive – remove ongoing fishing impacts.	Passive – improving environmental conditions in inshore coastal areas. Remove dredging and trawling from some small areas. Active – transplanting of shellfish into higher density patches in small protected areas. New improved shellfish spat collecting techniques that require minimal human intervention (labour) to operate.	Restoring Tamaki Strait would be a very ambitious undertaking. Passive – improved water clarity and reduced seafloor muddiness through adjacent land catchment management. Active – as environmental conditions improve, transplanting of green-lipped mussel, horse mussel and clam (species identity/s yet to be resolved) patches. And possibility seagrass, and meadow-forming macro-algal species. Deployment of in situ spat collectors.

Existing larval/propagule supply?	Yes, intertidal seagrass beds exist	Unknown	The species involved are likely to still be present in adjacent areas	Yes, for some areas (e.g. scallops in Kawau Bay), but poorly known for most areas	Yes, for seagrass. Unknown for other species
Likely relative costs	Unknown	Unknown Main cost would be removal of commercial fishing, and enforcement as appropriate if not a voluntary approach by the industry.	Unknown Main cost would be removal of commercial fishing, and enforcement as needed.	Unknown	Unknown

Areas potentially restorable as a habitat landscape

Although perhaps the most ambitious of all, consideration might be given to attempting to restore some areas as mosaics of habitats. For example, Tamaki Strait today is a body of water with turbid waters, and a seafloor composed of a mixture of extensive drifts of dead relic shell and muddy seafloors. Its adjacent rocky shores hold narrow bands of kelp forests, and intertidal seagrass has returned to several of its relatively protected beaches. One narrow subtidal seagrass fringe extending out from a larger intertidal bed is known, located at Clarks Bay. Juvenile snapper are present in the strait, with snapper (<100 mm) nurseries occurring at its eastern end. However, historical evidence, though limited, suggests a very different ecosystem once existed. A research trawl survey tow in 1901 in the centre of Tamaki Strait reported '*a net full of grass and weed*', and Greenway (1969) mapped several green-lipped mussel beds. The existence of large deposits of large dead relic shellfish (species not yet known, nor the age of the relic shells) suggest that in the past large abundances of large-bodied filter-feeding shellfish once dominated the system. The 1901 trawl observation suggests that seagrass may once have been common in the system, likely subtidal (the 'grass' referred to), along with macroalgae (the 'weed' referred to), which may have been species of the green seaweed *Caulerpa*, which forms larger seafloor meadows, and/or red macroalgal beds (range of possible species), as is seen in part of eastern Te Rawhiti Strait (Bay of Islands) in the present day. Alternatively, it may have been brown seaweeds, either growing attached to the bivalve shells, or as drift from adjacent rocky reef populations.

If these historical hints have been interpreted correctly, this system was historically profoundly different from the present day. Although aspirational, efforts could be made to learn more about the past ecological system of Tamaki Strait and, combined with work on potential restoration of seagrass, green-lipped mussels, and other species, future efforts could be undertaken to potentially move it back towards a more desirable and productive state.

7. CONCLUSIONS

Undoubtedly, the Hauraki Gulf is fundamentally different in the present day than it was historically. Large declines in the numbers and sizes of many species have occurred, which are best documented for fisheries species where empirical data has been collected. Habitats, especially biogenic habitats, are much less well documented, and there are no historical maps of most habitats of where they occurred in the past (aside from some information on mangrove forests and green-lipped mussels). That information gap continues today and makes it in many ways problematic to set restoration targets and objectives, when it is not known what we are trying to restore towards. Green-lipped mussel and scallop beds are the only real exception, and that knowledge comes from 50+ years of intensive fishing of these beds as a resource (mussels in 1910s–1960s, scallops in 1970s–present). Restoration work on those species is logical and justifiable, assuming that it is undertaken in areas where beds are well documented to have occurred historically. Other species are less certain. Present day seafloor habitat mapping, reconstructions from in-situ death assemblages, species distribution extraction from historical photography, and further detailed interviewing of relevant retired fishers (before their knowledge is lost forever) would be a significant step towards trying to reduce this uncertainty.

8. ACKNOWLEDGEMENTS

The Ministry for Primary Industries (now Fisheries New Zealand) is thanked for initiating and funding this desktop review, with special thanks to Conor Paul for initiating and seeing the project through. Ian Tuck (NIWA) is thanked for providing internal peer review of the draft report. Karen Tunley and other Fisheries New Zealand staff thoroughly reviewed the report and improved its readability and context, along with staff from DOC and Auckland Council, and are thanked for their reviews. Suze Baird of Fisheries New Zealand is thanked for providing detailed formal editorial on the draft report.

9. REFERENCES

Abrahim, G. (2005). Holocene sediments of Tamaki Estuary: characterisation and impact of recent human activity on an urban estuary in Auckland, New Zealand. Unpublished PhD Thesis, University of Auckland, Auckland, New Zealand 361 p.

Alexander, J, (2019). Zebra mussels are transforming the Great Lakes and fueling rampant algae growth. Bridge Magazine, Michigan. https://www.mlive.com/environment/2012/08/zebra_mussels_are_transforming.html

Alfaro, A.C. (2006). Benthic macro-invertebrate community composition within a mangrove/seagrass estuary in northern New Zealand. *Estuarine, Coastal and Shelf Science 66(1–2)*: 97–110.

Anderson, T.J.; Morrison, M.; MacDiarmid, A.; Clark, M.; D'Archino, R.; Nelson, W.; Tracey, D.; Gordon, D.; Read, G.; Kettles, H.; Morrisey, D.; Wood, A. ; Anderson, O.; Smith, A.M.; Page, M.; Paul-Burke, K.; Schnabel, K.; Wadhwa, S. (2019). Review of New Zealand's key biogenic habitats. Report prepared for the Ministry for the Environment, January 2019. NIWA Client Report No: 2018139WN. Available at https://environment.govt.nz/publications/review-of-new-zealands-key-biogenic-habitats/

Armiger, L.C. (1964). An occurrence of *Labyrinthula* in New Zealand. *New Zealand Journal of Botany 2*: 3–9.

Armiger, L.C. (1965). A contribution to the autecology of *Zostera*. Unpublished MSc thesis, University of Auckland, 89 p.

Attrill, M.J.; Strong, J.A.; Rowden, A.A. (2000). Are macroinvertebrate communities influenced by seagrass structural complexity? *Ecography 23*: 114–121.

Auckland Council (2011). North Shore Heritage Thematic Review Report. Compiled by Heritage Consultancy Services for Auckland Council. 1 July 2011. *Auckland Council Document TR 2011/010.*

Auckland Regional Council (2000a). Coastal Planting Guide No. 1. ARC Wonderful Wetlands.pdf (aucklanddesignmanual.co.nz)

Auckland Regional Council (2000b). Wonderful wetlands: wetland restoration and planting guide for the Auckland Region. Wetlands Restoration Guide (aucklandcouncil.govt.nz)

Auster, P.J.; Malatesta, R.J.; LaRosa, S.C.; Cooper, R.A.; Stewart, L.L. (1991). Microhabitat utilization by the megafaunal assemblage at a low relief outer continental shelf site – Middle Atlantic Bight, USA. *Journal of Northwest Atlantic Fishery Science 11(1–2)*: 59–69.

Ayson, L.F. (1901). Report on experimental trawling. Appendix to the Journals of the House of Representatives, 1901 Session I, H-15. https://paperspast.natlib.govt.nz/parliamentary/AJHR1901-I.2.4.2.23

Ayson, L.F. (1908). Report on experimental trawling. Appendix to the Journals of the House of Representatives, 1908 Session I, H-15B. https://paperspast.natlib.govt.nz/parliamentary/AJHR1908-I.2.5.2.22/1

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.

Backhurst, M.K.; Cole, R.G. (2000). Biological impacts of boating at Kawau Bay, north-eastern New Zealand. *Journal of Environmental Management 60*: 239–251.

Baird, K. (2015). Statement of Evidence in reply of Karen Baird on behalf of The Environmental Defense Society Incorporated and Royal Forest and Bird Protection Society of New Zealand Submission numbers 4735 and 4848. Hearing Topic 33 - General Coastal Marine Zone and Activities.

Baker, C.S.; Boren, L.; Childerhouse, S.; Constantine, R.; van Helden, A.; Lundquist, D.; Rayment, W.; Rolfe, J.R. (2019). Conservation status of New Zealand marine mammals, 2019. *New Zealand Threat Classification Series 29*. 18 p.

Baker, C.S.; Clapman, P.J. (2004). Modeling the past and future of whales and whaling. *Trends in Ecology and Evolution 19(7)*: 365–371.

Bartrom, A.A. (1990). The Coromandel scallop enhancement project, 1987 to 1989. A joint programme between the Overseas Fishery Co-operation Foundation of Japan and the Ministry of Agriculture and Fisheries, New Zealand. (Unpublished report by MAF Fisheries North.)

Battershill, C.N. (1987). Factors affecting the structure and dynamics of subtidal communities characterised by sponges. Unpubl. PhD thesis, University of Auckland. 227 p.

Beadman, H.A.; Kaiser, M.J.; Galandidi, M.; Shucksmith, R.; Willows, R.I. (2004). Changes in species richness with stocking density of marine bivalves. *Journal of Applied Ecology 41(3)*: 464–475.

Beauchamp, A.J. (2012). Statement of Evidence prepared for Environment Court hearing by Mangawhai Harbour Restoration Society Incorporated, Northland Regional Council, and Royal Forest and Bird Protection Society Incorporated, Topic ENV-2011-AKL-000110.

Beauchamp, A.J.; Parrish, G.R. (1999). Bird use of the sediment settlement ponds and roost areas at Port Whangarei. *Notornis 46(4)*: 470–483.

Beaugrand, G. (2004). The North Sea regime shift: evidence, causes, mechanisms and consequences. *Progress in Oceanography 60*: 245–62.

Beaumont, J.; Anderson, T.J.; MacDiarmid, A. (2015). Benthic flora and fauna of the Patea Shoals region, South Taranaki Bight. NIWA Client Report No: WLG2012-55. 184 p. NIWA Client report (epa.govt.nz)

Bell, J.; Blayney, A. (2017). Use of mangrove habitat by threatened or at-risk birds – Literature review prepared for Waikato Regional Council. Waikato Regional Council Technical Report 2017/23. 33 p.

Bell, J.J. (2008). The functional roles of marine sponges. *Estuarine, Coastal, and Shelf Science 79*: 341–353.

Bell, R.G.; Hume, T.M.; Hicks, D.M. (2001). *Planning for climate change effects on coastal margins*. Report prepared for the Ministry for the Environment as part of the New Zealand Climate Change Programme, Ministry for the Environment, Wellington. 86 p.

Bellingham, M. (2013). Banded rail. In C. M. Miskelly (Ed.), New Zealand Birds Online. Retrieved from http://www.nzbirdsonline.org.nz/species/banded-rail.

Bergin, D.O. (1994). Performance of transplanted indigenous salt marsh species, Maketu estuary. New Zealand Forest Research Institute Contract Report, Wellington.

Beu, A.G. (2004). Marine mollusca of oxygen isotope stages of the last 2 million years in New Zealand. Part 1: Revised generic positions and recognition of warm-water and cool-water migrants. *Journal of the Royal Society of New Zealand 34*: 111–265.

Botha, A. (2011). Foraging distances and habitat preferences of banded rail in the Ohiwa Harbour. *Bay of Plenty Regional Council Environmental Publication No. 2010/06*. Whakatane: Bay of Plenty Regional Council.

Bradshaw, A.D. (2002). Introduction and philosophy, pp. 3–9. *In*: Bradshaw, A.D.; Perrow, M.R., Davy, A.J. (Eds.), Handbook of Ecological Restoration, Principles of Restoration, Vol. 1, Cambridge University Press, Cambridge.

Broekhuizen, N.; Zeldis, J.; Stephens, S.A.; Oldman, J.W.; Ross, A.H.; Ren, J.; James, M.R. (2002). Factors related to the sustainability of shellfish aquaculture in the Firth of Thames: a preliminary analysis. NIWA Client Report EVW-02243 prepared for and presented as *FOT* Carrying capacity Environment Waikato 02/09, ARC Technical Publication TP 182 available at https://www.waikatoregion.govt.nz/assets/WRC/WRC-2019/TR02-09.pdf.

Brumbaugh, R.D.; Hancock B. (2019). Shellfish reef restoration: An introduction, pp. 2–6. *In*: Fitzsimons, J.; Branigan, S.; Brumbaugh, R.D.; McDonald, T.; zu Ermgassen, P.S.E. (Eds.), *Restoration Guidelines for Shellfish Reefs*. The Nature Conservancy, Arlington VA, USA.

Bulmer, R.H.; Stephenson, F.; Hannah, J.F.E.; Townsend, M.; Hillman, J.R.; Schwendenmann, L.; Lundquist, C.J. (2020). Blue carbon stocks and cross-habitat subsidies. *Frontiers in Marine Science 7*: 380. doi: 10.3389/fmars.2020.00380

Burns, B.R.; Ogden, J. (1985). The demography of the temperate mangrove [Avicennia marina (Forsk.) Vierh.] at its southern limit in New Zealand. *Australian Journal of Ecology 10(2)*: 125–133.

Butler, A.J.; Brewster, F.J. (1979). Size distributions and growth of the fan shell *Pinna bicolor* in South Australia. *Australian Journal of Marine and Freshwater Research 30*: 25–39.

Campbell, D.A. (1946). Down to the sea in slips. Soil Conservation and Rivers Control Council, Bulletin No. 5.

Carroll, E.L.; Jackson, J.A.; Paton, D.; Smith, T.D. (2014). Two intense decades of 19th century whaling precipitated rapid decline of right whales around New Zealand and East Australia. *PLoS ONE 9(4)*: e93789. doi:10.1371/journal.pone.0093789

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

Clapman, P.; Ivashchenko, Y. (2009). A whale of a deception. *Marine Fisheries Review 71*: 44–52.

Clapman, P.; Mikhalev, Y.; Franklin, W.; Paton, D.; Baker, C.S.; Ivashchenko, Y.; Brownell, R.L. (2009). Catches of Humpback Whales, *Megaptera novaeangliae*, by the Soviet Union and Other Nations in the Southern Ocean, 1947–1973. *Marine Fisheries Review 71*: 39–43.

Clemens, S.; Boss, R.; Light, A.; Stockin, K.A. (2011). Attack on blue penguin by a New Zealand fur seal. *New Zealand Journal of Zoology 38*: 333–336, DOI: 10.1080/03014223.2011.595421

Cloern, J.E. (1999). The relative importance of light and nutrient limitation of phytoplankton growth: A simple index of coastal ecosystem sensitivity to nutrient enrichment. *Aquatic Ecology 33 (1)*: 3–16.

Cloern, J.E. (2001). Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series 210*: 223–253.

Coen, L.D.; Grizzle, R.E. (2007). The importance of habitat created by molluscan shellfish to managed species along the Atlantic Coast of the United States. ASMFC Habitat Management Series #8. 108 p.

Cohen, J.E. (1997). Population, economics, environment and culture: an introduction to human carrying capacity. *Journal of Applied Ecology 34*: 1325–1333.

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

Commito, J.A.; Como, S.; Grupe, B.M.; Dow, W.E. (2008). Species diversity in the soft-bottom intertidal zone: Biogenic structure, sediment, and macrofauna across mussel bed spatial scales. *Journal of Experimental Marine Biology and Ecology 366(1–2)*: 70–81.

Compton, T.J.; Morrison, M.A.; Leathwick, J.R.; Carbines, G. (2012). Ontogenetic habitat associations of a demersal fish species (*Pagrus auratus*) identified using boosted regression trees. *Marine Ecology Progress Series 462*: 219–230.

Connolly, R.M.; Hindell, J.S.; Gorman, D. (2005). Seagrass and epiphytic algae support the nutrition of a fisheries species, *Sillago schomburgkii*, in adjacent intertidal habitats. *Marine Ecology Progress Series 286*: 69–79.

Cook, S. de C. (Ed.) (2010). *New Zealand Coastal Marine Invertebrates Volume I*. Canterbury University Press. 640 p.

Cox, G.J. (1977). Utilization of New Zealand mangrove swamps by birds. Unpubl. MSc thesis, University of Auckland.

Craeymeersch, J.A.; Jansen, H.M. (2019). Bivalve assemblages as hotspots for biodiversity, pp. 275–294. *In*: Smaal, A.C.; Ferreira, J.G.; Grant, G.; Petersen, J.K.; Strand, O. (Eds.), *Goods and services of marine bivalves*. Springer.

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: Marine and Freshwater Ecosystems 9(5)*: 461–483.

Cranfield, H.J.; Rowden, A.A.; Smith, D.J.; Gordon, D.P.; Michael, K.P. (2004). Macrofaunal assemblages of benthic habitat of different complexity and the proposition of a model of biogenic reef habitat regeneration in Foveaux Strait, New Zealand. *Journal of Sea Research 52(2)*: 109–125.

Cranfield, P.J. (2019). Magnitude and extent of water clarification services provided by bivalve suspension feeding, pp. 119–141. *In*: Smaal, A.C.; Ferreira, J.G.; Grant, G.; Petersen, J.K.; Strand, O. (Eds.), Goods and services of marine bivalves. Springer.

Crisp, P.; Daniel, L.J.; Tortell, P.; Wagemaker, P.; Nature Conservation Council. (1990). *Mangroves in New Zealand: trees in the tide*. Wellington: GP Books.

Cummings, V.; Hewitt, J.; Halliday, J.; Mackay, G. (2007). Optimizing the success of *Austrovenus stutchburyi* restoration: preliminary investigations in a New Zealand estuary. *Journal of Shellfish Research 26*: 89–100

Cummings, V.J.; Thrush, S.F.; Hewitt, J.E.; Turner, S.J. (1998). The influence of the pinnid bivalve *Atrina zelandica* (Gray) on benthic macroinvertebrate communities in soft sediment habitats. *Journal of Experimental Marine Biology and Ecology 228(2)***:** 227–240.

Davidson, R.J.; Richards, L.A.; Duffy, C.A.J.; Kerr, V.; Freeman, D.; D'Archino, R.; Read, G.B.; Abel, W. (2010). Location and biological attributes of biogenic habitats located on soft substrata in the Marlborough Sounds. Prepared by Davidson Environmental Ltd for Department of Conservation and Marlborough District Council. Unpublished Research, Survey and Monitoring Report no. 675. 51 p.

Davies, N.M.; Walsh, C.; Hartill, B. (1993). Estimating catch at age of snapper from west coast and Hauraki Gulf fisheries, 1992–93. Northern Fisheries Region Internal Report No. 17. 58 p. (Unpublished report held by NIWA, Auckland.)

Day, J.H.; Hutchings, P.A. (1979). An annotated checklist of Australian and New Zealand Polychaeta, Archiannelida, and Myzostomida. *Records of the Australian Museum 32*: 80–161.

de Lange, P.J.; Rolfe, J.R.; Barkla, J.W.; Courtney, S.P.; Champion, P.D.; Perrie, L.R.; Beadel, S.M.; Ford, K.A.; Breitwieser, I.; Schönberger, I.; Hindmarsh-Walls, R.; Heenan, P.B.; Ladley, K. (2017). Conservation status of New Zealand indigenous vascular plants, 2017. *New Zealand Threat Classification Series 22*. 82 p.

De Luca-Abbott, S. (2001). Biomarkers of sublethal stress in the soft-sediment bivalve *Austrovenus stutchburyi* exposed in-situ to contaminated sediment in an urban New Zealand harbour. *Marine Pollution Bulletin 42*: 817–825.

De Luca-Abbott, S.; Creese, R.G.; Lewis, G.D.; Wells, R.M.G. (2000) Adenylate energy charge and total adenylate nucleotide pool as biomarkers of sublethal stress in the cockle, *Austrovenus stutchburyi*. *Australian Journal of Ecotoxicology 6*: 35–44.

Department of Conservation (2011). Seals can take care of themselves. DOC media release on NZ fur seals coming ashore along the Northland coast. See http://www.doc.govt.nz/aboutdoc/news/media-releases/2011/seals-can-take-care-of-themselves/

Department of Conservation (2012). Leave seals be. DOC media release on NZ fur seals coming ashore in the Bay of Islands. See http://www.doc.govt.nz/about-doc/news/mediareleases/2012/leave-seals-be/

Dencer-Brown, A.M.; Alfaro, A.C.; Bourgeois, C.; Sharma, S.; Milne, S. (2020). The secret lives of mangroves: Exploring New Zealand's urban mangroves with integrated biodiversity assessments. *Ocean & Coastal Management 191*: 105185.

Dencer-Brown, A.M.; Alfaro, A.C.; Milne, S. (2019). Muddied waters: perceptions and attitudes towards mangroves and their removal in New Zealand. *Sustainability 11(9)*: 2631.

Dewas, S.E.A.; O'Shea, S. (2011). The influence of *Tucetona laticostata* (Bivalvia: Glycymeridae) shells and rhodolith patches on benthic-invertebrate assemblages in Hauraki Gulf, New Zealand. *New Zealand Journal of Marine and Freshwater Research 46(1)*: 47–56.

Dittmann, S. (1990). Mussel beds - amensalism or amelioration for intertidal fauna? *Helgoländer Meeresuntersuchungen 44(3–4)*: 335–352.

Duarte, C., Jarramillo, E., Contreras, H., Figueroa, L. (2006). Community structure of the macroinfauna in the sediments below an intertidal mussel bed (*Mytilus chilensis* (Hupe)) of southern Chile. *Revista Chilena de Historia Natural 79(3)*: 353–368.

Eckman, J.E. (1987). The role of hydrodynamics in recruitment, growth, and survival of *Argopecten irradians* (L.) and *Anomia simplex* (D'Orbigny) within eelgrass meadows. *Journal of Experimental Marine Biology and Ecology 106*: 165–191.

Elasber, B.; Farinas-Franco, J.M.; Wilson, C.D., Kregting, L., Roberts, D. (2013). Identifying optimal sites for natural recovery and restoration of impacted biogenic habitats in a special area of conservation using hydrodynamic and habitat suitability modelling. *Journal of Sea Research 77*: 11–21.

Elderton (1974). The Resources of New Zealand; The Cyclopedia of New Zealand, vol. 2. On the drainage and conversion of the Hauraki Plains into dairy farms. (See Rufus Edward Tye, Hauraki Plains Story (Paeroa: Thames Valley News Ltd, 1974).

Ellis J.; Cummings V.; Hewitt J.; Thrush S.; Norkko A. (2002). Determining effects of suspended sediment on condition of a suspension feeding bivalve (*Atrina zelandica*): Results of a survey, a laboratory experiment and a field transplant experiment. *Journal of Experimental Marine Biology and Ecology 267 (2)*: 147–174.

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(3)*: 349–366.

Elliott, M.; Quintino, V. (2007). The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. *Marine Pollution Bulletin 54*: 640–645.

Erftemeijer, P.L.A.; Lewis III, R.R. (1999). Planting mangroves on intertidal flats: Habitat restoration or habitat conversion? Paper presented at the ECOTONE-VIII Seminar 'Enhancing coastal ecosystem restoration for the 21st century. Ranong & Phuket, 23–28 May 1999.

Farr, T.; Broom, J.; Hart, D.; Neill, K.; Nelson, W. (2009). Common coralline algae of northern New Zealand: an identification guide. *NIWA Information Series 70*. 249 p.

Fisheries New Zealand (2021). Fisheries Assessment Plenary, May 2021: stock assessments and stock status. Compiled by the Fisheries Science Team, Fisheries New Zealand, Wellington, New Zealand. 1782 p.

Fitzsimons, J.A.; Branigan, S.; Gillies, C.L.; Brumbaugh, R.D.; Cheng, J.; DeAngelis, B.M.; Geselbracht, L.; Hancock, B.; Jeffs, A.; McDonald, T.; McLeod, I.M.; Pogoda, B.; Theuerkauf, S.J.; Thomas, M.; Westby, S.; zu Ermgassen, P.S.E. (2020). Restoring shellfish reefs: Global guidelines for practitioners and scientists. *Conservation Science 2(6)*: e198. https://doi.org/10.1111/csp2.198

Fonseca, M.S.; Koehl, M.A.R. (2006). Flow in seagrass canopies: The influence of patch width. *Estuarine, Coastal and Shelf Science 67*: 1–9.

Fonseca, M.S.; Zieman, J.C.; Thayer, G.W.; Fisher, J.S. (1983). The role of current velocity in structuring eelgrass (*Zostera marina* L.) meadows. *Estuarine, Coastal and Shelf Science 17:* 367–380.

Francis, M.; Morrison, M.; Leathwick, J.; Walsh, C.; Middleton, M. (2005). Predictive models of small fish presence and abundance in northern New Zealand harbours. *Estuarine, Coastal, and Shelf Science 64*: 419–435

Francis, M.P.; Morrison, M.A.; Leathwick, J.A.; Walsh, C.M. (2011). Predicting patterns of fish richness, occurrence, and abundance in New Zealand estuaries. *Marine and Freshwater Research 62*: 1327– 1341.

Francis, R.I.C.C; McKenzie, J.R. (2015). Assessment of the SNA 1 stocks in 2013. *New Zealand Fisheries Assessment Report 2015/76*. 82 p.

Gacia, E.; Duarte, C.M. (2001). Sediment retention by a Mediterranean *Posidonia oceanica* meadow: the balance between deposition and resuspension. *Estuarine, Coastal and Shelf Science 52:* 505–514.

Gales, N.J. (2017). New Zealand sea lion *Phocarctos hookeri*, pp. 763–765. *In*: Perrin, W.F.; Würsig, B.; Thewissen, J.G.M. (Eds), *Encyclopedia of marine mammals*. London, Academic Press.

Gall, M.; Swales, A.; Davies-Colley. R.; Bremner, D. (2020). Predicting visual clarity and light penetration from water quality measures in New Zealand estuaries. *Estuarine, Coastal and Shelf Science 219*: 429–443.

Gardiner, S.; Jones, H. (2020). Habitat mapping for the Waikato region coastal marine area: bathymetry and substrate type. *Waikato Regional Council Technical Report 2017/34*.

Gaskin, C.P.; Rayner, M.J. (2013). Seabirds of the Hauraki Gulf: natural history, research, and conservation. Auckland, New Zealand: Hauraki Gulf Forum presentation, 2013. DOI: 10.13140/RG.2.1.3745.4888

Geange, S.; Townsend., M.; Clark, D.; Ellis, J.I.; Lohrer, A.M. (2019). Communicating the value of marine conservation using an ecosystem service matrix approach. *Ecosystem Services 35*: 150–163.

Geist, J.; Hawkins, S.J. (2016). Habitat recovery and restoration in aquatic ecosystems: current progress and future challenges. *Aquatic Conservation: Marine and Freshwater Ecosystems 26(5)*: 942–962.

Gerovasileiou, V.; Smith, C.J.; Sevastou, K.; Papadopoulou, N.; Dailianis, T.; Bekkby, T.; Fiorentino, D.; McOwen, C.J., Amaro, T.; Tunka Bengil, E.G.; Bilan, M.; Boström, C.; Carreiro-Silva, MJ.; Cebrian, E.; Cerrano, C.; Danovaro, R.; Fraschetti, S.; Gagnon, K.; Gambi, C.; Grehan, A.; Hereu, B.; Kipson, S.; Kotta, J.; Linares, C.; Morato, T.; Ojaveer, H.; Orav-Kotta, H.; Sarà, A.; Scrimgeour, R. (2019). Habitat mapping in the European Seas - is it fit for purpose in the marine restoration agenda? *Marine Policy 106*: 103521

Gilby, B.L; Olds, A.D.; Connolly, R.M.; Henderson, C.J.; Schlacher, T.A. (2018). Spatial restoration ecology: placing restoration in a landscape context. *BioScience 68(12)*: 1007–1019.

Gillespie, J.L.; Nelson, C.S.; Nodder, S.D. (1998). Post-glacial sea-level control and sequence stratigraphy of carbonate-terrigenous sediments, Wanganui shelf, New Zealand. *Sedimentary Geology 122*: 245–266

Glade, T. (2003). Landslide occurrence as a response to land use change: A review of evidence from New Zealand. *Catena 51 (3–4)*: 297–314.

Gomez, B.; Carter, L.; Trustrum, N. (2007). A 2,400 year record of natural events and anthropogenic impacts in inter-correlated terrestrial and marine sediment cores: Waipaoa Sedimentary System, New Zealand. *Geological Society of America Bulletin 119*: 1415–1432.

Gormley, K.S.G.; Porter, J.S.; Bell, M.C.; Hull, A.D.; Sanderson, W.G. (2013). Predictive Habitat Modelling as a Tool to Assess the Change in Distribution and Extent of an OSPAR Priority Habitat under an Increased Ocean Temperature Scenario: Consequences for Marine Protected Area Networks and Management. *PLoS ONE 8(7)*: e68263. doi:10.1371/journal.pone.0068263

Grant, C.M.; Hay, B.E. (2003). A review of issues related to depletion of populations of selected infaunal bivalve species in the Hauraki Gulf Marine Park. A report prepared for the Hauraki Gulf Forum, Auckland. AquaBio Consultants Ltd.

Grech, A.; Chartrand-Miller, K.; Erftemeijer, P.; Fonseca, M.; McKenzie, L.; Rasheed, M.; Coles, R. (2012). A comparison of threats, vulnerabilities and management approaches in global seagrass bioregions. *Environmental Research Letters 7(2):* 1–8. 024006.

Greenway, J.P.C. (1969). Survey of mussels (Mollusca: Lamellibranchia) in the Firth of Thames, 1961–67. *New Zealand Journal of Marine & Freshwater Research 3*: 304–317.

Gutiérrez, J.L.; Jones, C.G.; Gareth, J.C.; Strayer, D.; Iribarne, O. (2003). Mollusks as ecosystem engineers: The role of shell production in aquatic habitats. *Oikos 101*: 79–90.

Haacks, M.; Thannheiser, D.T. (2003). The salt-marsh vegetation of New Zealand. *Phytocoenologia 33(2–3)*: 267–288.

Hadfield, M.; O'Callaghan, J.; Pritchard, M.; Stevens, C. (2018). Regions of freshwater influence (RoFI) and the implications for sediment deposition in the Hauraki Gulf. (Unpublished NIWA Client Report No: WLG2012-29 prepared for the Department of Conservation.)

Hancock, B.; zu Ermgassen, P. (2019). Enhanced production of finfish and large crustaceans by bivalve reefs, pp. 295–312. *In*: Smaal, A.C.; Ferreira, J.G.; Grant, G.; Petersen, J.K.; Strand, O. (Eds.), *Goods and services of marine bivalves*. Springer.

Handley, S.J.; Willis, T.J.; Cole, R.G.; Bradley, A.; Cairney, D.J.; Brown, S.N.; Carter, M.E. (2014). The importance of benchmarking habitat structure and composition for understanding the extent of fishing impacts in soft sediment ecosystems. *Journal of Sea Research 86*: 58–68.

Hare, I.J. (1992). Paterson Inlet marine benthic assemblages: Report on Coastal Investigations. *Southland Conservancy Technical Series No. 5*. Department of Conservation. Invercargill. 88 p.

Harrow, G.; Hawke, D.J.; Holdaway, R.N. (2006). Surface soil chemistry at an alpine procellariid breeding colony in New Zealand, and comparison with a lowland site. *New Zealand Journal of Zoology 33(2)*: 165–174. http://dx.doi.org/10.1080/03014223.2006.9518441

Hartill, B.W.; Cryer, M.; Morrison, M.A. (2004). Estimates of biomass, sustainable yield and harvest: neither necessary nor sufficient for the management of amateur intertidal fisheries. *Fisheries Research 71*: 209–222.

Harty, C. (2009) Mangrove planning and management in New Zealand and South East Australia – a reflection on approaches. *Ocean Coast Management 52(5)*: 278–286.

Harvey, A.; Woelkerling, W.; Farr, T.; Neill, K.; Nelson, W. (2005). Coralline algae of central New Zealand: an identification guide to common ‘crustose’ species. *NIWA Information Series No. 57*. 145 p.

Hatvany, M. (2008). Environmental failure, success and sustainable development: The Hauraki Plains wetlands through four generations of New Zealanders. *Environment and History 14*: 469–95.

Hawke, D.J.; Holdaway, R.N. (2003). Mainland petrel breeding as a driver of terrestrial ecosystem processes. *ConScience (Department of Conservation) 47*: 6–7.

Hawke, D.J.; Holdaway, R.N. (2005). Avian assimilation and dispersal of carbon and nitrogen brought ashore by breeding Westland petrels (Procellaria westlandica): a stable isotope study. *Journal of Zoology 266(4)*: 419–426. doi: 10.1017/S0952836905007065.

Hawke, D.J.; Holdaway, R.N. (2009). Nutrient sources for forest birds captured within an undisturbed petrel colony, and management implications. *Emu 109*: 163–169. http://dx.doi.org/10.1071/MU08035.

Hay, C.H. (1990). The ecological importance of the horse mussel *Atrina novaezelandica* with special reference to the Marlborough Sounds. Department of Conservation report prepared for the Nelson/Marlborough Regional Office. 7 p.

Hayward, B.W.; Stephenson, A.B.; Morley, M.; Riley, J.L.; Grenfell, H.R. (1997). Faunal changes in Waitemata Harbour sediments, 1930s-1990s. *Journal of the Royal Society of New Zealand 27*: 1–20.

Hayward, B.W.; Stephenson, A.B.; Morley, M.S.; Blom, W.M.; Grenfell, H.R.; Brook, F.J.; Riley, J.L.; Thompson, F.; Hayward, J.J. (2001). Marine biota of Parengarenga Harbour, Northland, New Zealand. *Records of the Auckland Institute and Museum 37*: 45–80.

Hewitt, J.E.; Cummings, V.J. (2013). Context-dependent success of restoration of a key species, biodiversity and community composition. *Marine Ecology Progress Series 479*: 63–73.

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.

Hickford, M.J.H.; Cagnon, M.; Schiel, D.R. (2010). Predation, vegetation and habitat-specific survival of terrestrial eggs of a diadromous fish, *Galaxias maculatus* (Jenyns, 1842). *Journal of Experimental Marine Biology and Ecology 385(1–2)*: 66–72.

Hindell, J.S.; Jenkins, G.P.; Keough, M.J. (2000). Evaluating the impact of predation by fish on the assemblage structure of fishes associated with seagrass (*Heterozostera tasmanica*) (Martens ex Ascherson) den Hartog, and unvegetated sand habitats. *Journal of Experimental Marine Biology and Ecology 255*: 153–174.

Hindell, J.S.; Jenkins, G.P.; Keough, M.J. (2001). Spatial and temporal variability in the effects of fish predation on macrofauna in relation to habitat complexity and cage effects. *Marine Ecology Progress Series 224*: 231–250.

Hooker, S.H. (1995). Life history and demography of the pipi, *Paphies australis* (Bivalvia: Mesodesmatidae) in northeastern New Zealand. Unpubl. PhD thesis, University of Auckland.

Holdaway, R.N. (1999). Introduced predators and avifaunal extinction in New Zealand, pp. 189–238. *In*: MacPhee, R.D.E. (ed.), *Extinctions in Near Time*. New York, Kluwer Academic.

Holdaway, R.N.; Hawke, D.J.; Hyatt, O.M.; Wood, G.C. (2007). Stable isotopic (δ15N, δ13C) analysis of wood in trees growing in past and present colonies of burrow-nesting seabirds in New Zealand. I. δ15N in two species of conifer (Podocarpaceae) from a mainland colony of Westland petrels (*Procellaria westlandica*), Punakaiki, South Island. *Journal of the Royal Society of New Zealand 37(2)*: 75–84.

Hopkins, G.A. (2002). Aspects of the biology of the horse mussel *Atrina zelandica* Gray in Doubtful Sound and off the Otago Coast, New Zealand. Unpubl. MSc thesis, University of Otago. 141 p.

Horstman, E.M.; Lundquist, C.J.; Bryan, K.R.; Bulmer, R.H.; Mullarney, J.C.; Stokes, D.J. (2018). The dynamics of expanding mangroves in New Zealand, pp. 25–31. *In*: Makowski C.; Finkl, C. (eds.). Threats to mangrove forests. *Coastal Research Library vol 25*. Springer, Cham. https://doi.org/10.1007/978-3-319-73016-5_2.

Hume, T.M.; McGlone, M.S. (1986). Sedimentation patterns and catchment use change recorded in the sediments of a shallow tidal creek, Lucas Creek, upper Waitemata Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research 20*: 677–687.

Hutchings, J.A.; Reynolds, J.D. (2004). Marine fish population collapses: consequences for recovery and extinction risk. *Bioscience 54*: 297–309.

Inglis, G.J. (2003). The seagrasses of New Zealand, pp. 148–157. *In*: Green, E.P.; Short, F.T. (Eds.) *World atlas of seagrasses*. University of California Press, Berkeley, California.

Isdale, A.M. (1964). The history of gold mining on "The River Thames". Ohinemuri Regional History Journal 1, June 1964.

Jackson, J.; Carrol, E.; Smith, T.D.; Zerbini, A.N.; Patenaude, N.; Baker, C.S. (2016). An integrated approach to historical population assessment of the great whales: case of the New Zealand southern right whale. *Royal Society Open Science 3*: 150669. http://dx.doi.org/10.1098/rsos.150669

Jones, E.G.; Morrison, M.A.; Davey, N.; Hartill, B.W.; Sutton, C. (2016). Biogenic habitats on New Zealand's continental shelf. Part I: Local Ecological Knowledge. *New Zealand Aquatic Environment and Biodiversity Report No. 174*. 95 p.

Jones, H. (2008). Coastal sedimentation: What we know and the information gaps, Environment Waikato Internal Series TR2008/12. 55 p.

Kaiser, M.J.; Attrill, M.J.; Jennings, S.; Thomas, D.N.; Barnes, D.K.A.; Brierley, A.S.; Hiddink, J.S.; Kaartokallio, H.; Polunin, N.V.C.; Raffaelli, D.G. (2005). *Marine Ecology: Processes, Symptoms, and Impacts*. Oxford University Press, Oxford.

Kelly, S.; Sim-Smith, C.; Faire, S.; Pierre, J.; Hikuroa, D.C.H. (2014). State of our Gulf 2014, Hauraki Gulf - Tīkapa Moana / Te Moananui-ā-Toi, State of the Environment Report 2014 (4). Auckland Council: Hauraki Gulf Forum. 195 p.

Keough, M.J.; Jenkins, G.P. (1995). Seagrass meadows and their inhabitants, pp. 221–239. *In*: Underwood, A.J.; Chapman, M.G. (Eds). *Coastal marine ecology of temperate Australia*. UNSW Press, Sydney, Australia.

Kettles, H.; Bell, R. (2016): Estuarine ecosystems, pp. 24–30. *In*: Robertson, H.; Bowie, S.; White, R.; Death, R.; Collins, D. (Eds). Freshwater conservation under a changing climate. Proceedings of a workshop hosted by the Department of Conservation, 10-11 December 2013, Wellington. Department of Conservation, Christchurch. 86 p.

Kohlmeier, D.; Pilditch, C.A.; Bornman, J.F.; Bischof, K. (2014a). Site specific differences in morphometry and photophysiology in intertidal *Zostera muelleri* meadows. *Aquatic Botany 116*: 104–109.

Kohlmeier, D.; Pilditch, C.A.; Bornman, J.F.; Bischof, K. (2014b). Adjustment of photoprotection to tidal conditions in intertidal seagrasses. *Journal of the Marine Biological Association of the United Kingdom 97(3)*: 571–579.

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*: 477–497.

Lausch, A.; Blaschke, T.; Haase, D.; Herzog, F.; Syrbe, R.U.; Tischendorf, L.; Walz, U. (2015). Understanding and quantifying landscape structure – a review on relevant process characteristics, data models and landscape metrics. *Ecological Modelling 295*: 31–41.

Layton, C.; Coleman, M.; Marzinelli, E.M.; Steinberg, P.D.; Swearer, S.E.; Vergés, A.; Wernberg, T.; Johnson, C.R. (2020). Kelp forest restoration in Australia. *Frontiers in Marine Science 7*.

Lindsey, E.L.; Altieri, A.H.; Witman, J.D. (2006). Influence of biogenic habitat on the recruitment and distribution of a subtidal xanthid crab. *Marine Ecology Progress Series 306*: 223–231.

Ling, J.K. (2002). Impact of colonial sealing on seal stocks around Australia, New Zealand and sub Antarctic islands between 150 and 170 degrees East. *Australian Mammalogy 24*: 117–126.

Livingston, R.J. (2006). Restoration of aquatic ecosystems, Taylor and Francis, Boca Raton, Florida. 423 p.

Lovelock, C.E.; Feller, I.C.; Ellis, J.; Schwarz, A.M.; Hancock, N.; Nichols, P.; & Sorrell, B. (2007). Mangrove growth in New Zealand estuaries: The role of nutrient enrichment at sites with contrasting rates of sedimentation. *Oecologia 153*: 633–641.

Lovelock, C.E., Sorrell, B.K.; Hancock, N.; Hua, Q.; Swales, A. (2010). Mangrove forest and soil development on a rapidly accreting shore in New Zealand. *Ecosystems 13 (3)*: 437–451.

Lowe, M. (2013). Factors affecting the habitat usage of estuarine juvenile fish in northern New Zealand. Unpubl. PhD thesis, University of Auckland. 282 p.

Lundquist, C.J.; Jones, T.C.; Parkes, S.M.; Bulmer, R.H. (2018). Changes in benthic community structure and sediment characteristics after natural recolonisation of the seagrass *Zostera muelleri*. *Scientific Reports 8*. 13250 (2018). https://doi.org/10.1038/s41598-018-31398-2

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. (Draft New Zealand Aquatic Environment and Biodiversity Report held by Ministry for Primary Industries.) 615 p.

MacDiarmid, A.; Smith, I.; Paul, L.; Francis, M.; McKenzie, A.; Parsons, D.; Hartill, B.; Stirling, B; Cleaver, P. (2012). 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 held by Ministry for Primary Industries.) 38 p.

MacDiarmid, A.B.; Abraham, E.; Baker, C.S.; Carroll, E.; Chagué-Goff, C.; Cleaver, P.; Francis, M.P.; Goff, J.; Horn, P.; Jackson, J.A.; Lalas, C.; Lorrey, A.; Marriot, P.; Maxwell, K.; McFadgen, B.; McKenzie, A.; Neil, H.; Parsons, D.; Patenaude, N.; Paton, D.; Paul, L.J.; Pitcher, T.; Pinkerton, M.H.; Smith, I.; Smith, T.D.; Stirling B. (2016). 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.

Mace, J. (1981). Separation Point closed. Catch 8 (July) (6): 15–16.

MacLeod, M.; Cooper, J.A.G. (2005) Carrying capacity in coastal areas. *In*: Schwartz M.L. (eds) *Encyclopedia of Coastal Science. Encyclopedia of Earth Science Series*. Springer, Dordrecht. https://doi.org/10.1007/1-4020-3880-1_62.

Mair, G. (1902). Notes on fish found in the Piako River. *Transactions and Proceedings of the Royal Society of New Zealand 35*: 319–320.

Marden, M. (2011). Sedimentation history of Waipaoa catchment. Landcare Research, Contract Report No. LC 790. 46 p.

MarLIN Glossary (2005). The Marine Life Information Network for Britain and Ireland: online glossary of scientific terms. Available from: <http://www.marlin.ac.uk/Glossaries/Gen_Glossary.htm> (2005).

Marsden, I.D.; Adkins, S.C. (2010). Current status of cockle bed restoration in New Zealand. *Aquaculture International 18*: 83–97.

Marsden, I.D.; Heremaia, C. (1998). Saltmarsh restoration and management, pp. 136–147. *In*: Wallace, C.; Weeber, B.; Buchanan, S. (Eds.) *SeaViews Marine ecosystem management: obligations and opportunities Proceedings of the conference held in Wellington, 11–14th of February 1998*. Environment and Conservation Organisations of New Zealand. 278 p.

Matheson, F.; Dos Santos, V.D.; Pildritch, C. (2008). Environmental stressors of New Zealand seagrass: an introduction and progress report. (Unpublished NIWA Client Report HAM208–53.) 37 p.

Matheson, F.; Wadhwa, S. (2012). Seagrass in Porirua Harbour: preliminary assessment of restoration potential. NIWA report for Greater Wellington Regional Council. 35 p. (Unpublished report, available from Greater Wellington Regional Council.)

Matheson, F.; Wadhwa, S.; Taumoepeau, A.; Smith, J. (2010). Seagrass in the eastern Bay of Islands: past and present abundance, threats, and management options. (Unpublished NIWA Client Report HAM2010-43). 24 p. BoI Seagrass Report Final (nrc.govt.nz).

Matheson, F.E.; Lundquist, C.J.; Gemmill, C.E.C.; Pilditch, C.A. (2011). New Zealand seagrass more threatened than IUCN review indicates. Letter to Editor. *Biological Conservation 144*: 2749–2750.

Matheson, F.E.; Reed, J.; Dos Santos, V.M.; Mackay, G.; Cummings, V.J. (2017). Seagrass rehabilitation: successful transplants and evaluation of methods at different spatial scales. *New Zealand Journal of Marine and Freshwater Research 51(1)*: 96–109.

McDowall, R.M. (1965). The composition of the New Zealand whitebait catch, 1964. *New Zealand Journal of Science 8(3)*: 285–300.

McDowall R.M.; Charteris, S.C. (2006). The possible adaptive advantages of terrestrial egg deposition in some fluvial diadromous galaxiid fishes (Teleostei: Galaxiidae). *Fish and Fisheries 7(3)*: 153–164.

McGarigal, K.; Tagil, S.; Cushman, S.A. (2009). Surface metrics: an alternative to patch metrics for the quantification of landscape structure. *Landscape Ecology 24*: 433–50.

McGlone, M.S. (2009). Postglacial history of New Zealand wetlands and implications for their conservation. *New Zealand Journal of Ecology 33(1)*: 1–23.

McKenzie, A.; MacDiarmid, A. (2012). Biological parameters and biomass estimates for some commercial fish stocks in the Hauraki Gulf and along the Otago-Catlins shelf for the period 1930–2006. (Unpublished draft New Zealand Aquatic Environment and Biodiversity Report held by Fisheries New Zealand.) 100 p.

McLeod, I.M. (2009). Green-lipped mussels, *Perna canaliculus*, in soft sediment systems in northeastern New Zealand. Unpubl. MSc thesis, University of Auckland. 107 p.

McLeod, I.M.; Parsons, D.M.; Morrison, M.A.; Le Port, A.; Taylor, R.B. (2012). Factors affecting the recovery of soft-sediment mussel reefs in the Firth of Thames, New Zealand. *Marine and Freshwater Research 63*: 78–83.

McLeod, I.M.; Parsons, D.M.; Morrison, M.A.; Van Dijken, S.G.; Taylor, R.B. (2014) Mussel reefs on soft sediments: a severely reduced but important habitat for macroinvertebrates and fishes in New Zealand. *New Zealand Journal of Marine and Freshwater Research 48(1)*: 48–59.

McLusky, D.S.; Elliott, M. (2004). *The Estuarine Ecosystem: Ecology, Threats and Management* (third ed.), Oxford University Press, Oxford. 222 p.

Metzger, J.P.; Brancalion, P.H.S. (2016). Landscape ecology and restoration processes, pp. 90–120. *In*: Palmer M.A.; Zedler J.B.; Falk D.A. (Eds), *Foundations of Restoration Ecology*. Island Press, Washington, DC. https://doi.org/10.5822/978-1-61091-698-1_4

Mills, V.S.; Berkenbusch, K. (2009). Seagrass (*Zostera muelleri*) patch size and spatial location influence infaunal macroinvertebrate assemblages. *Estuarine, Coastal and Shelf Science 81(1)*: 123–129.

Moksnes, P.O. (2002). The relative importance of habitat-specific settlement, predation and juvenile dispersal for distribution and abundance of young juvenile shore crabs *Carcinus maenas* L. *Journal of Experimental Marine Biology and Ecology 271(1)*: 41–73.

Moksnes, P.O.; Pihl, L.; Montfrans, J. van. (1998). Predation on post larvae and juveniles of the shore crab *Carcinus maenas*: Importance of shelter, size and cannibalism. *Marine Ecology Progress Series 166*: 211–225.

Möllmann, C.; Diekmann, R.; Muller-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.

Monismith, S.G.; Koseff, J.R.; Thompson, J.K.; O'Riordan, K.A.; Nepf, H.M. (1990) A study of model bivalve siphonal currents. *Limnology & Oceanography 35*: 680–696.

Morley, M.S.; Hayward, B.W.; White, A. (2001). Changes to the intertidal biota 1950's–2000 at Howick Beach, Auckland. (Unpublished report, Auckland Museum).

Morrisey, D.; Beard, C.; Morrison, M.; Craggs, R.; Lowe, M. (2007). The New Zealand mangrove: Review of the current state of knowledge. *Auckland Regional Council Technical Publication No. TP325*. (Prepared for Auckland Regional Council by National Institute of Water & Atmospheric Research.)

Morrisey, D.; Swales, A.; Dittmann, S.; Morrison, M.A.; Lovelock, C.E.; Beard, C.M.; Gibson, R.N. (2010). The ecology and management of temperate mangroves. *Oceanography and Marine Biology: An Annual Review 4*8: 43–160.

Morrisey, D.J.; Skilleter, G.A.; Ellis, J.I.; Burns, B.R.; Kemp, C.E.; Burt, K. (2003). Differences in benthic fauna and sediment among mangrove (*Avicennia marina* var. *australasica*) stands of different ages in New Zealand. *Estuarine Coastal and Shelf Science 56(3-4)*: 581–592.

Morrisey, D.J.; Swales, A.; Dittmann, S.; Morrison, M.A.; Lovelock, C.E.; Beard, C.M. (2010). The ecology and management of temperate mangroves, pp. 43–160. *In*: Gibson, R.N.; Atkinson, R.J.A.; Gordon, J.D.M. (Eds). *Oceanography and Marine Biology - An Annual Review 48*.

Morrison, M.; Drury, J.; Shankar, U. (2001). An acoustic survey of the seafloor habitats of Tiritiri Matangi Island and of the northeastern side of Great Barrier Island. (Unpublished consultancy report prepared by NIWA for the Department of Conservation.)

Morrison, M.; Drury, J.; Shankar, U.; Hill, A. (2002). A broad scale seafloor habitat assessment of the Firth of Thames using acoustic mapping, with associated video and grab sample ground-truthing. (Unpublished NIWA Client Report AKL2002–014.) 71 p.

Morrison, M.; Drury, J.; Shankar, U.; Middleton, C.; Smith, M. (2003). A broad scale, soft sediment habitat assessment of the Hauraki Gulf. (Unpublished NIWA Client Report AKL2003–64.) 62 p.

Morrison, M.; Tuck, I.D.; Taylor, R.B.; Miller, A. (2016). An assessment of the Hauraki Gulf Cable Protection Area, relative to the adjacent seafloor. *Auckland Council Technical Report TR2016/004*. 54 p.

Morrison, M.A. (1999). Population dynamics of the scallop *Pecten novaezelandiae* in the Hauraki Gulf. Unpubl. PhD thesis, University of Auckland.

Morrison, M.A. (2003). A review of the natural marine features and ecology of Whangarei Harbour. (Unpublished NIWA Client Report AKL2003-122, prepared for the Department of Conservation.) 60 p.

Morrison, M.A.; Cryer, M. (2003). Potential for artificial enhancement of scallops in Northland. NIWA Client Report: AKL2003- 038. Prepared for Enterprise Northland Aquaculture Development Group March 2003.

Morrison, M.A.; 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.A.; Jones, E.; Parsons, D.P.; Grant, C. (2014b). Habitats and areas of particular significance for coastal finfish fisheries management in New Zealand: A review of concepts and current knowledge, and suggestions for future research. *New Zealand Aquatic Environment and Biodiversity Report No. 125*. 202 p.

Morrison, M.A.; Lowe, M.L.; Grant, C.G.; Smith, P.J.; Carbines, G.; Reed, J.; Bury, S.J.; Brown, J. (2014c). Seagrass meadows as biodiversity and productivity hotspots. *New Zealand Aquatic Environment and Biodiversity Report No. 137*. 147 p.

Morrison, M.A.; Lowe, M.L.; Parsons, D.M.; Usmar, N.R.; 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.A.; McKenzie, J.; Bian, R. (2019). Pre-recruit (0+) snapper (*Chrysophrys auratus*) beam trawl and beach seine surveys of East Northland and the Hauraki Gulf (SNA 1). *New Zealand Fisheries Assessment Report 2019/72*. 50 p.

Morrison, M.A.; Pallentin, A.; MacKay, K.; Kane, T.; Woolley, J-M.; MacKay E.; Tuck, I. (in review, a). Beneath the Waves: Port Fitzroy. Draft NIWA chart, Miscellaneous Series. Published by the National Institute of Water and Atmospheric Research Ltd.

Morrison, M.A.; Pallentin, A.; MacKay, K.; Kane, T.; Woolley, J-M.; MacKay E.; Tuck, I. (in review, b). Tamaki Strait and Ponui Channel. Draft NIWA chart, Miscellaneous Series. Published by the National Institute of Water and Atmospheric Research Ltd.

Morrison, M.A.; Pallentin, A.; MacKay, K.; Kane, T.; Woolley, J-M.; MacKay E.; Tuck, I. (in review, c). Beneath the Waves: Koputauaki Bay to Waimate Island. Draft NIWA chart, Miscellaneous Series. Published by the National Institute of Water and Atmospheric Research Ltd.

Nelson, W.A. (2009). Calcified macroalgae - critical to coastal ecosystems and vulnerable to change: A review. *Marine and Freshwater Research 60*: 787–801.

Nesbit, G.J (1998). Reseeding and hatchery potential of *Pecten novaezelandiae* and effects of recreational harvesting. Unpublished MSc thesis, University of Auckland.

Neubauer, P.; Abraham, E.R.; Berkenbusch, K. (2015). Predictability of cockle (*Austrovenus stutchburyi*) population trends in New Zealand's northern North Island. PeerJ PrePrints. https://dx.doi.org/10.7287/peerj.preprints.1422v1.

NIWA (2010). Restoring shellfish beds to harbours and estuaries A guide for community groups. https://niwa.co.nz/sites/niwa.co.nz/files/restoring_shellfish_beds_final.pdf

Nogales, M.; Martín, A.; Tershy, B.R.; Donlan, C.J.; Veitch, D.; Puerta, N.; Wood, B.; Alonso, J. (2004). A review of feral cat eradication on islands. *Conservation Biology 18*: 310–319

Norkko, J.; Hewitt, J.E.; Thrush, S.F. (2006) Effects of increased sedimentation on the physiology of two estuarine soft-sediment bivalves, *Austrovenus stutchburyi* and *Paphies australis*. *Journal of Experimental Marine Biology and Ecology 333*: 12–26.

Nyström, M.; Norström, A.V.; Blenckner, T.; de la Torre-Castro, M.; Eklöf, J.S.; Folke, C.; Österblom, H.; Steneck, R.S.; Thyresson, M.; Troell, M. (2012). Confronting feedbacks of degraded marine ecosystems. *Ecosystems 15(5)*: 695–710.

O'Donnell, E. (2011). Tairua Harbour and catchment management plan (Waikato Regional Council Technical Report No. 2011/40). Waikato Regional Council.

O'Riordan, C.A.; Monismith, S.G.; Koseff, J.R. (1993). A study of concentration boundary-layer formation over a bed of model bivalves. *Limnology & Oceanography 38*: 1712–1729.

Orth, R.J.; Carruthers T.J.B.; Dennison, W.B.; Duarte, C.M.; Fourqurean, J.W.; Heck Jr., K.L.; Hughes, A.R.; Kendrick, G.A.; Kenworthy, W.J.; Olyarnik, S.; Short, F.T.; Waycott, M.; Williams, S.L. (2006). A global crisis for seagrass ecosystems. *BioScience 56 (12)*: 987–996.

Palmer, M.A.; Zedler, J.B.; Falk, D.A. (Eds) (2016). Chapter 1, Ecological Theory and Restoration Ecology, pp. 3–26. *In:* Palmer M.A.; Zedler J.B.; Falk D.A. (eds). *Foundations of Restoration Ecology*. Island Press, Washington, DC. https://doi.org/10.5822/978-1-61091-698-1_1

Park, S.G. (1999). Changes in abundance of seagrass (*Zostera* spp.) in Tauranga Harbour from 1959–96. Environment Bay of Plenty Environmental Report 99/30. 19 p.

Park, S.G. (2000). Bay of Plenty Maritime Wetlands Database. Environmental Report 2000/21. Environment Bay of Plenty.

Parsons, D.; Morrison, M.; MacDiarmid, A.B.; Stirling, B.; Cleaver, P.; Smith, I.W.G.; Butcher, M. (2009). Risks of shifting baselines highlighted by anecdotal accounts of New Zealand's snapper (*Pagrus auratus*) fishery. *New Zealand Journal of Marine & Freshwater Research 43*: 965–983.

Paul, L.J. (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.J. (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*. 177 p

Petersen, J.K.; Hansen, J.W.; Laursen, M.B.; Clausen, P.; Carstensen, J.; Conley, D.J. (2008). Regime shift in a coastal marine ecosystem. *Ecological Applications 18*: 497–510.

Peterson, G. (2000). Political ecology and ecological resilience: an integration of human and ecological dynamics. *Ecological Economics 35*: 323–336.

Petraitis, P.S.; Methratta E.; Rhile, E.; Vidargas, N.; Dudgeon, S.R. (2009). Experimental confirmation of multiple community states in a marine ecosystem. *Oecologia 161*: 139–148.

Pinkerton, M.H.; MacDiarmid, A.; Beaumont, J.; Bradford-Grieve, J.; Francis, M.P.; Jones, J.; Lalas, C.; Lundquist, C.J.; McKenzie, A.; Nodder, S.D. Paul, L.; Stenton-Dozey, J.; Thompson, D.; Zeldis, J. (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*. 349 p.

Poole, A.L.; Adams, N.M. (1994). *Trees and shrubs of New Zealand.* Field Guide Series. Manaaki Whenua Press, Lincoln, New Zealand

Powell, A.W.B. (1936). Animal communities of the Waitemata and Manukau Harbours. *Transactions of the Royal Society of New Zealand 66*: 354–401.

Ragnarsson, S.A.; Raffaelli, D. (1999). Effects of the mussel *Mytilus edulis* L. on the invertebrate fauna of sediments. *Journal of Experimental Marine Biology and Ecology 241(1)*: 31–43.

Reid, B. (1969). Mussel survey Hauraki Gulf and Firth of Thames 1968. *New Zealand Marine Department Fisheries Technical Report No. 34.* 24 p.

Richard, Y.; Abraham, E.; Berkenbusch, K. (2020). Assessment of the risk of commercial fisheries to New Zealand seabirds, 2006–07 to 2016–17. *New Zealand Aquatic Environment and Biodiversity Report No. 237*. 57 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.

Roe, E.; Van Eeten, M. (2001). Threshold-based resource management: a framework for comprehensive ecosystem management. *Ecological Applications 27*: 195–214.

Roe, E.; van Eeten, M. (2002). Reconciling ecosystem rehabilitation and service reliability mandate in large technical systems: findings and implications of three major US ecosystem management initiatives for managing human-dominated aquatic-terrestrial ecosystems. *Ecosystems 5*: 509–528.

Rose, K.A.; Cowan Jr, J.H.; Winemiller, K.O.; Myers, R.A.; Hilborn, R. (2001). Compensatory density dependence in fish populations: importance, controversy, understanding and prognosis. *Fish and Fisheries 2*: 293–327.

Rullens, V.; Lohrer, A.M.; Townsend, N.; Pilditch, C.A. (2019). Ecological mechanisms underpinning ecosystem service bundles in marine environments – a case study for shellfish. *Frontiers in Marine Science 6*: 409.

Saxton, F.L. (1980a). The coral beds of Tasman and Golden Bay. Ministry of Agriculture and Fisheries. (Unpublished report available from Fisheries New Zealand, Nelson.) 13 p.

Saxton, F.L. (1980b). Coral loss could deplete fish stocks. *Catch '80 7(8)*: 12–13.

Scheffer, M.; Carpenter, S.; Foley, J.A.; Folke, C.; Walker B. (2001). Catastrophic shifts in ecosystems. *Nature 413*: 591–596

Schulte, D.M.; Burke, R.P.; Lipcius, R.N. (2009). Unprecedented restoration of a native oyster metapopulation. *Science 325*: 1124–1128.

Schwarz, A.; Morrison, M.; Hawes, I.; Halliday, J. (2006). Physical and biological characteristics of a rare marine habitat: sub-tidal seagrass beds of offshore islands. Department of Conservation, Wellington. *Science for Conservation No. 269*. 39 p.

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.; Usmar, N. (2003). The role of the Hauraki Gulf Cable Protection Zone in protecting exploited fish species: de facto marine reserve? *DOC Research & Development Series 253*. 27 p.

Simenstad, C.; Reed, D.; Ford, M. (2006). When is restoration not?: incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. *Ecological Engineering 26(1)*: 27–29.

Sim-Smith, C. (2015). Proposed restoration of soft-sediment mussel beds using waste mussel shell. Ecological assessment of effects. Report prepared by Coast & Catchment Ltd. 13 p.

Sim-Smith, C.; Kelly, S. (2015). Subtidal habitat survey around Rotoroa Island. Client report for the Mussel Reef Restoration Trust Report Number: 2015-06. Prepared by Coast & Catchment Ltd. 33 p.

Sim-Smith, C.; Kelly, S. (2016). Monitoring of a deployed subtidal mussel bed at Rotoroa Island. Client report for the Mussel Reef Restoration Trust Report Number: 2016-005. Report prepared by Coast & Catchment Ltd. 21 p.

Sivaguru, K.; Grace, R. (2002). Habitat and species diversity of deep reefs and sediments at Great Barrier Island. (Unpublished DOC report, held at DOC Auckland)

Smaal, A.C.; Ferreira, J.G.; Grant, G.; Petersen, J.K.; Strand, O. (Eds.). *Goods and services of marine bivalves*. SpringerOpen. https://link.springer.com/content/pdf/10.1007%2F978-3-319-96776-9.pdf

Smith, A.M.; McGourty, C.R.; Kregting, L.; Elliot, A. (2005). Subtidal *Galeolaria hystrix* (Polychaeta: Serpulidae) reefs in Paterson Inlet, Stewart Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research 39*: 1297–1304.

Smith, A.M.; Wood, A.C.L.; Liddy, M.F.A.; Shears, A.E.; Fraser, C.I. (2010). Human impacts in an urban port: The carbonate budget, Otago Harbour, New Zealand. *Estuarine, Coastal and Shelf Science 90(2): 73–79.*

Smith, I. (2013). Pre-European Māori exploitation of marine resources in two New Zealand case study areas: species range and temporal change. *Journal of the Royal Society of New Zealand 43(1)*: 1–37.

Smith, I.W.G. (2005). Retreat and resilience: fur seals and human settlement in New Zealand, pp. 6–18. *In*: Monks, G. (Ed.), *The exploitation and cultural importance of sea mammals*. Cambridge, Oxbow Books.

Smith, I.W.G. (2011). Estimating the magnitude of pre-European Māori marine harvest in two New Zealand study areas. *New Zealand Aquatic Environment and Biodiversity Report No. 82*. 70 p.

Smith, T.D.; Reeves, R.R.; Josephson, E.A.; Lund, J.N. (2012). Spatial and seasonal distribution of American whaling and whales in the age of sail. *PLoS ONE 7(4)*: e34905. doi:10.1371/journal.pone.0034905

Society for Ecological Restoration (2004). The SER international primer on ecological restoration. Tucson: Society for Ecological Restoration International. www.ser.org.

State of our Gulf (2020). Hauraki Gulf / Tīkapa Moana / Te Moananui-ā-Toi State of the Environment Report 2020.

Steneck, R.S.; Graham, M.H.; Bourque, B.J.; Corbett, D.; Erlandson, J.M.; Estes, J.A.; Tegner, M.J. (2002). Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental Conservation 29*: 436–459.

Stewart, C. (2018). The role of East Northland seagrass nurseries in growth and condition of juvenile snapper, *Chrysophrys auratus*, an important northern New Zealand coastal fishery. Unpubl. BSc (Hons) Thesis, University of Auckland.

Stewart, M.J. (2005) Ecological effects associated with urban development on populations of the New Zealand cockle (*Austrovenus stutchburyi*). PhD thesis, University of Auckland.

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.

Suyadi; Gao, J.; Lundquist, C.J.; Schwendenmann, L. (2019). Land-based and climatic stressors of mangrove cover change in the Auckland Region, New Zealand. *Aquatic Conservation: Marine and Freshwater Ecosystems 29(9)*: 1466–1483.

Swales, A.; Bell, R.; Gorman, R.; Oldman, J.; Altenberger, A.; Hart, C.; Ovenden, R. (2009). Potential future changes in mangrove habitat in Auckland's east-coast estuaries. Hamilton, New Zealand: National Institute of Water & Atmospheric Research, Research Report

Swales, A.; Bentley, S.; Lovelock, C.; Bell, R. (2007). Sediment processes and mangrove-habitat expansion on a rapidly-prograding muddy coast, New Zealand, pp. 1441–1454. *In*: Coastal Sediments '07: Proceedings of the Sixth International Conference on Coastal Engineering and Science of Coastal Sediment Processes, New Orleans. American Society of Civil Engineers.

Swales A.; Bentley, S.J.; Lovelock, C.E. (2015). Mangrove-forest evolution in a sediment-rich estuarine system: opportunists or agents of geomorphic change? *Earth Surface Processes and Landforms 40(12)*: 1672–1687.

Swales A.; Denys, P.; Pickett, V.I.; Lovelock, C.E. (2016). Evaluating deep subsidence in a rapidly accreting mangrove forest using GPS monitoring of surface-elevation benchmarks and sedimentary records. *Marine Geology 380*: 205–218.

Swales, A.; Hume, T.M.; Oldman, J.W.; Green, M.O. (1997). Holocene sedimentation and recent human impacts in a drowned valley estuary, pp. 895–900. *In*: Lumsden, J. Proceedings of the 13th Australian Coastal and Ocean Engineering Conference. Centre for Advanced Engineering, University of Canterbury, Christchurch, New Zealand.

Swales, A.; Williamson, R.B.; Van Dam, L.F.; Stroud, M.; McGlone, M.S. (2002). Reconstruction of urban stormwater contamination of an estuary using catchment history and sediment profile dating. *Estuaries 25(1)*: 43–56.

Tait, L.W.; Lohrer, A.M.; Townsend, M.; Atalah, J.; Floerl, O.; Inglis, G.J. (2020). Invasive ecosystem engineers threaten benthic nitrogen cycling by altering native infaunal and biofouling communities. *Scientific Reports 10*: Article 1581.

Talman, S.G.; Norkko, A.; Thrush, S.F.; Hewitt, J.E. (2003). Habitat structure and the survival of juvenile scallops *Pecten novaezelandiae*: Comparing predation in habitats with varying complexity. *Marine Ecology Progress Series 269*: 197–207.

Tan, Y.M.; Dalby, O.; Kendrick, G.A.; Statton, J.; Sinclair, E.A.; Fraser, M.W.; Macreadie, P.I.; Gillies, C.L.; Coleman, R.A.; Waycott, M.; van Dijk, K.; Vergés, A.; Ross, J.D.; Campbell, M.L.; Matheson, F.E.; Jackson, E.L.; Irving, A.D.; Govers, L.L.; Connolly, R.M.; McLeod, I.M.; Rasheed, M.A.; Kirkman, H.; Flindt, M.R.; Lange, T.; Miller, A.D.; Sherman, C.D.H. (2020). Seagrass restoration is possible: insights and lessons from Australia and New Zealand. *Frontiers in Marine Science 7*. https://doi.org/10.3389/fmars.2020.00617

Tennyson, A.; Martinson, P. (2006). *Extinct birds of New Zealand*. Te Papa Press, Wellington, New Zealand. 180 p.

Tett, P.; Gowen, R.; Mills, D.; Fernandes, T.; Gilpin, L.; Huxham, M.; Kennington, K.; Read, P.; Service, M.; Wilkinson, M.; Malcolm, S. (2007). Defining and detecting undesirable disturbance in the context of eutrophication. *Marine Pollution Bulletin 53*: 282–297.

Taylor, M.J. (1996). How native fish spawn on land. *Water & Atmosphere 4 (3)*: 23–25.

Taylor, R.B.; Morrison, A. (2008). Soft-sediment habitats and fauna of Omaha Bay, northeastern New Zealand. *Journal of the Royal Society of New Zealand 38(3)*: 187–214.

Thom, R.M. (2000). Adaptive management of coastal ecosystem restoration projects. *Ecological Engineering 15(3–4)*: 365–372.

Thomsen, D.; Marsden, I.D., Sparrow, A.D. (2005). A field experiment to assess the transplant success of salt marsh plants into tidal wetlands. *Wetlands Ecology and Management 13*: 489–497.

Thompson, D.R.; Lundquist, C.J.; Nelson, W.A.; Nodder, S.; Francis, M.P.; Hupman, K.; Watson, S.; Woelz, S. (2020). Sites and habitats of significance within the Waikato region's coastal marine area. (Unpublished NIWA Report prepared for Waikato Regional Council.) 133 p.

Thomsen, M.S.; Adam, P.; Silliman, B.R. (2009). Chapter 18: Anthropogenic threats to Australasian coastal salt marshes, pp. 361–390. *In*: Silliman, B.R.; Grosholz, E.D.; Bertness, M.D. (Eds.). *Human impacts on salt marshes: a global perspective*. 432 p.

Towns, D.R; Simberloff, D.; Atkinson, I.A.E. (1997). Restoration of New Zealand islands: redressing the effects of introduced species. *Pacific Conservation Biology 3*: 99–124.

Townsend, M.; Lohrer, A.M. (2019). Empirical valuation of an ecosystem services map developed from ecological principles and biophysical parameters. *Frontiers in Marine Science 6(21)*. 11 p.

Townsend, M.; Thrush, S.F.; Lohrer, A.M.; Hewitt, J.E.; Lundquist, C.J.; Carbines, M.; Felsing, M. (2014). Overcoming the challenges of data scarcity in mapping marine ecosystem service potential. *Ecosystem Services 8:* 44–55.

Truttman, L. (2013). Timespanner: "Poor old Britomart ... They chuck in the sea": the demise of Point Britomart 1872–1885. https://timespanner.blogspot.com/2013/07/poor-old-britomart-they-chuck-in-sea.html

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.

Turner, M.G. (1989). Landscape ecology: the effect of pattern on process. *Annual Review of Ecology and Systematics 20*: 171–197.

Turner, S.; Schwarz, A. (2004). Information for the management and conservation of seagrass in New Zealand. (Unpublished NIWA report prepared for Department of Conservation.) 48 p.

Turner, S.; Schwarz, A. (2006). Management and conservation of seagrass in New Zealand: An Introduction. *Science for Conservation 264*. Department of Conservation, Wellington, New Zealand. 90 p.

Turner, S.J.; Cummings, V.J.; Hewitt, J.E. (1998). Guidelines for community shellfish restoration and enhancement projects. *NIWA Information Series No. 4*. 34 p.

van der Heide, T.; van Nes, E.H.; Geerling, G.W.; Smolders, A.J.P.; Bouma, T.J.; van Katwijk, M.M. (2007). Positive feedbacks in seagrass ecosystems: implications for success in conservation and restoration. *Ecosystems 10*: 1311–22.

van Houte-Howes, K.S.S.; Turner, S.J.; Pilditch, C.A. (2004). Spatial differences in macroinvertebrate communities in intertidal seagrass habitats and unvegetated sediment in three New Zealand estuaries. *Estuaries and Coasts 27(6)*: 945–957.

Veitch, C.R.; Habraken, A.M. (1999). Waders of the Manukau Harbour and Firth of Thames. *Notornis 46*: 45–70.

Villard, M-A.; Metzger, J.P. (2014). Beyond the fragmentation debate: a conceptual model to predict when habitat configuration really matters. *Journal of Applied Ecology 51*: 309–318.

Vooren, C.M. (1975). Nursery grounds of tarakihi (Teleostei: Cheilodactylidae) around New Zealand. *New Zealand Journal of Marine and Freshwater Research 9(2)*: 121–158.

Walsh, C.; Armiger, H.; Bian, R.; Buckthought, D.; McKenzie, J. (2017). Age composition of commercial snapper landings in SNA 8, 2015–16. *New Zealand Fisheries Assessment Report 2017/02*. 40 p.

Walsh, C.; Horn, P.; McKenzie, J.; Ó Maolagáin, C.; Buckthought, D.; Sutton, C.; Armiger, H. (2014a). Age determination protocol for snapper (*Pagrus auratus*). *New Zealand Fisheries Assessment Report 2014/51*. 33 p.

Walsh, C.; McKenzie, J.; Bian, R.; Armiger, H.; Rush, N.; Smith, M.; Spong, K.; Buckthought, D. (2014b). Age composition of commercial snapper landings in SNA 1, 2012–13. *New Zealand Fisheries Assessment Report 2014/55*. 62 p.

Walsh, C.; Parsons, D.; Armiger, H.; Bian, R.; Evans, O. (2019b). Length and age composition of commercial snapper landings in SNA 8, 2018–19. *New Zealand Fisheries Assessment Report 2019/73*. 35 p.

Walsh, C.; Parsons, D.; Bian, R.; Armiger, H.; Buckthought, D.; Smith, M.; Rush, N. (2019a). Age composition of commercial snapper landings in SNA 1, 2017–18. *New Zealand Fisheries Assessment Report 2019/45*. 62 p.

Waycott, M.; Duarte, C.M.; Carruthers, T.J.B.; Orth, R.J.; Dennison, W.C.; Olyarnik, S.; Calladine, A.; Fourqurean, J.W.; Heck, K.L.; Hughes, A.R.; Kendrick, G.A.; Kenworthy, W.J.; Short, F.T.; Williams, S.L. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences 106(30)*: 12377–12381.

Weinstein, M.P. (2005) Managing coastal growth in the 21st century: sustainability science and the paradox of the dual mandate. pp. 218–226. *In*: Feng, H. (Ed.) *Urban Dimensions of Environmental Change: Science, Exposures, Policies and Technologies*. Chinese Science Press, Shanghai, People's Republic of China.

Weinstein, M.P. (2008). Ecological restoration and estuarine management: placing people in the coastal landscape. *Journal of Applied Ecology 45(1)*: 296–304.

Weinstein, M.P.; Baird, R.C.; Conover, D.O.; Gross, M.; Keulartz, J.; Loomis, D.K.; Naveh, Z.; Peterson, S.B.; Reed, D.J.; Roe, E.; Swanson, R.L.; Swart, J.A.A.; Teal, J.M.; Turner, R.E.; Van der Windt, H.J. (2007). Managing coastal resources in the 21st century. *Frontiers in Ecology and the Environment 5*: 43–48.

Weinstein, M.P.; Reed, D.J. (2005) Sustainable coastal development: the dual mandate and a recommendation for ‘commerce managed areas’. *Restoration Ecology 13*: 174–182.

Widdows, J.; Pope, N.D.; Brinsley, M.D.; Asmus, H.; Asmus, R.M. (2008). Effects of seagrass beds (*Zostera noltii* and *Z. marina*) on near-bed hydrodynamics and sediment re suspension. *Marine Ecology Progress Series 358*: 125–136.

Wilcox, M.; Jeffs, A. (2017). Is attachment substrate a prerequisite for mussels to establish on soft sediment substrate? *Journal of Experimental Marine Biology and Ecology 495*: 83–88.

Wilcox, M.; Jeffs, A. (2019). Impacts of sea star predation on mussel bed restoration. *Restoration Ecology 27(1)*: 189–197.

Wilcox, M.; Kelly, S.; Jeffs, A. (2018). Ecological restoration of mussel beds onto soft sediment using transplanted adults. *Restoration Ecology 26(3)*: 581–590.

Wilcox, M.; Kelly, S.; Jeffs, A. (2020). Patterns of settlement within a restored mussel bed site. *Restoration Ecology 28(2)*: 337–346.

Williams, J.R.; Parkinson, D.M.; Bian, R. (2013). Biomass survey and yield calculation for the Coromandel commercial scallop fishery, 2012. *New Zealand Fisheries Assessment Report 2013/18*. 57 p.

Wilmshurst, J.M.; Hunt, T.L.; Lipo, C.P.; Anderson, A.J. (2010). High precision radiocarbon dating shows recent and rapid initial human colonisation of east Polynesia. *Proceedings of the National Academy of Sciences of USA 108 (5)*: 1815–1820. https://doi.org/10.1073/pnas.1015876108

Wilton, D. (2019). Cook landmarks at 'The Thames' (New Zealand), November 1769. https://thetreasury.org.nz/cookwilton/landmarks.htm

Worthy, T.H.; Holdaway, R.N. (2002). The Lost World of the Moa: Prehistoric Life of New Zealand. Bloomington/Christchurch: Indiana University Press/Canterbury University Press. 718 p.

Wood, A.C.L.; Probert, P.K.; Rowden, A.A.; Smith, A.M. (2012). Complex habitat generated by marine bryozoans: a review of its distribution, structure, diversity, threats and conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems 22*: 547–563.

Wotherspoon, L.; Lee, K. (2016). Dynamic characteristics of Auckland Central Business District reclaimed zones. *New Zealand Geomechanics News 92*.

Young, B.M.; Harvey, E.L. (1996) A spatial analysis of the relationship between mangrove (*Avicennia marina* var. *australasica*) physiognomy and sediment accretion in the Hauraki Plains, New Zealand. *Estuarine, Coastal Shelf Science 42(2)*: 231–246.

APPENDIX 1

Reproduced from Fitzsimons et al. (2020).

PRACTITIONERS' CHECKLIST

This table provides a high-level checklist for practitioners to help guide the establishment and delivery of shellfish restoration projects.

Know the system you are working in *(Chapters 1 and 2)*	Become familiar with the ecosystem in its local setting (e.g. consider its historical distribution*), causes for decline, current threats (including diseases), bivalve lifecycle and reproduction methods and associated community assemblages. Gather evidence of recruitment strength and timing from previous research, observation, aquaculture operators and settlement plates.
Develop a restoration concept and socialise with potential project stakeholders and supporters *(Chapters 1 and 2)*	Consider developing a short document that outlines project aspirations and potential approaches. Use this to receive feedback and support for establishing a more detailed feasibility plan and funding proposals. Include regulators in the outreach.
Establish a feasibility plan *(Chapter 3)*	Consider including the following in a feasibility plan: • Identification of reference ecosystems or reference models and derived targets • Clearly defined S.M.A.R.T. objectives • Identification of project stakeholders and supporters • Likely funding streams • Different restoration approaches • Availability and disease tolerance of broodstock and source of seed (if larvae limited)
Identify funding sources and secure funding *(Chapter 2)*	Consider linking ecosystem service outcomes to beneficiaries and targeting funding opportunities linked to ecosystem service outcomes. Explore opportunities to leverage and match initial support.
Establish project management systems *(Chapters 3, 4 and 5)*	Establish detailed project and implementation plans, communication plans, volunteer management, legal framework and contracts, detailed risk assessments, site management plans, tenders and quotes, etc.
Know biosecurity risks and permitting requirements *(Chapter 4)*	Identify biosecurity and disease risks to wild populations and to aquaculture and fishing industries. Understand requirements and development times to secure permits. Understand/address the potential threat of the harvest of shellfish from the restored reef.
Undertake habitat suitability assessments and pilot studies *(Chapters 3 and 5)*	Identify optimal places for restoration with the system using suitability assessments, history of the most recent shellfish reefs, and pilot studies.
Confirm technical approach(es) required to support recovery including reef designs *(Chapter 5, 6, 7 and 8)*	Does the ecosystem require reconstruction (e.g. addition of substrate and shellfish), assisted regeneration (e.g. addition of substrate or shellfish) or management to limit threats (e.g. sediment, disease or predation). What reef designs will be used to support these technical approaches?
Undertake restoration *(Chapter 5 and 6)*	Work with community volunteers, contractors and third parties to mobilise and deploy substrate, shellfish and reduce/remove threats.
Undertake monitoring, evaluation and reporting *(Chapter 7)*	Measure progress against predefined restoration targets and reference ecosystems and models. Measure universal indicators.
Effectively communicate outcomes of your project to stakeholders, practitioners and the research community *(Chapter 9)*	Plan for communication, do the basics and target visual mediums and social media.