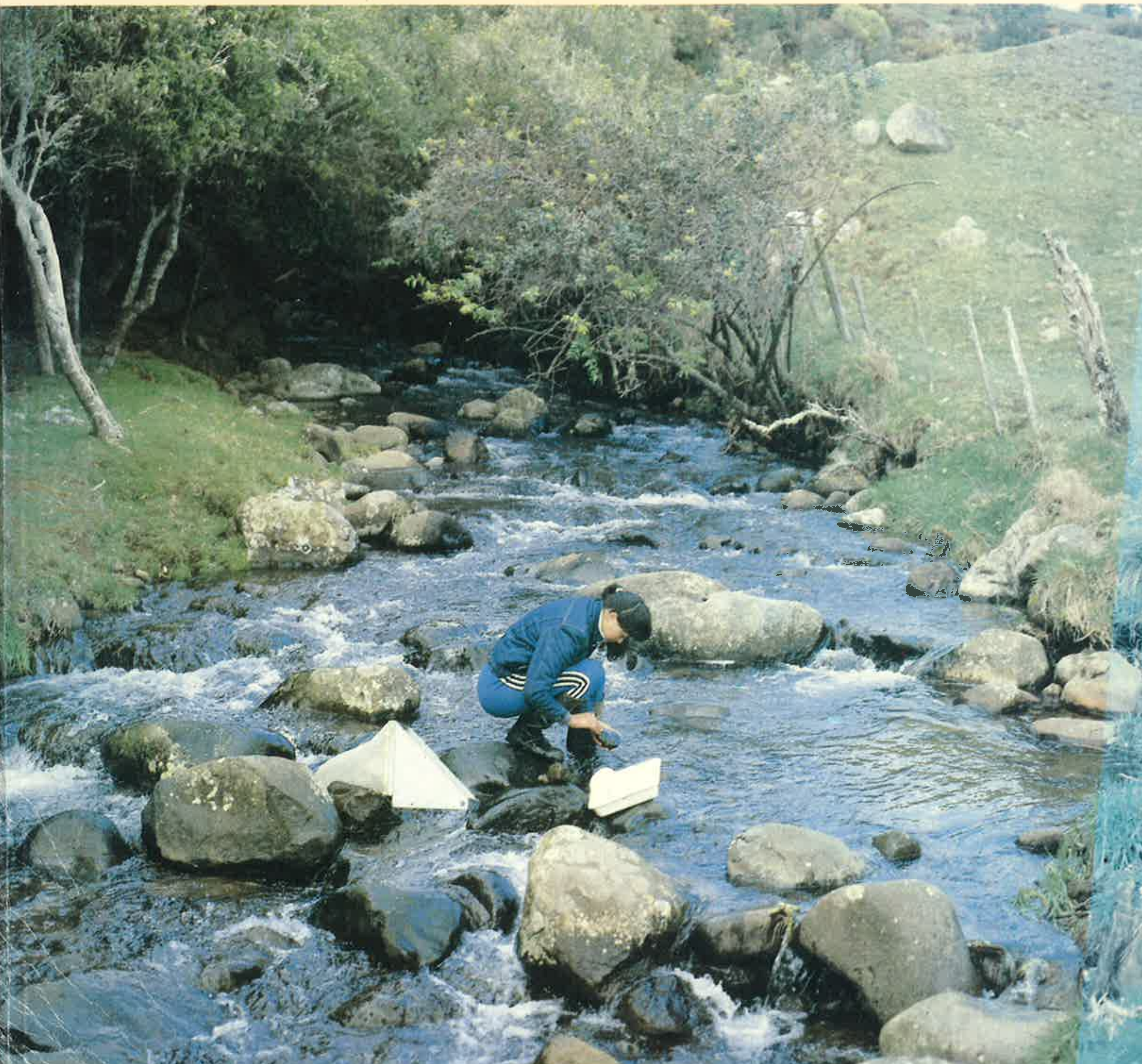
WATER AND SOIL MISCELLANEOUS PUBLICATION NUMBER 83

Biological Monitoring In Freshwaters

PART 2

PROCEEDINGS OF A SEMINAR, HAMILTON, 21–23 NOVEMBER 1984



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WATER AND SOIL MISCELLANEOUS PUBLICATION NO. 83

Biological Monitoring in Freshwaters: Proceedings of a Seminar, Hamilton, 21–23 November 1984 Part 2

Edited by

R. D. Pridmore and A. B. Cooper
Water Quality Centre
Ministry of Works and Development
Hamilton

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These seminar proceedings present information on the principles and techniques of biological monitoring in New Zealand freshwaters. Also discussed are desirable objectives for future biological research, particularly with regard to the further development of techniques for water quality management.

The proceedings are published in two parts. Part 2 presents information on two main topics: (1) Monitoring of macroinvertebrates and (2) Monitoring of fish.

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(Front cover)

Sampling for macroinvertebrates in Rangitukia Stream, Mt Pirongia, Waikato.

Photo: P. McIntosh, Photographic Department, University of Waikato, Hamilton.

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CONTENTS

FOREWORD

In November 1984, the Water Quality Centre (Water and Soil Directorate, MWD) hosted a seminar on "Biological Monitoring in Freshwaters" at Waikato University. The seminar was attended by 94 individuals representing 13 regional water boards, 8 government agencies, 6 universities, 4 acclimatisation societies, 3 consulting firms, 2 research institutes, and 2 industries.

The aims of the seminar were: (1) to present current knowledge on the principles and techniques applicable to biological monitoring in New Zealand freshwaters; and (2) to consider desirable directions for future biological research, with particular regard to the development of useful tools for water quality management.

Papers were presented by selected speakers on 6 main topics: (1) philosophy of monitoring; (2) biostatistics; (3) monitoring of aquatic plants; (4) monitoring of aufwuchs communities; (5) monitoring of macroinvertebrates; and (6) monitoring of fish. This publication contains 24 of the 26 papers presented.

In reading these papers, we have been impressed by the amount and quality of freshwater biological work which is going on in New Zealand, and by the conscientious attempts of virtually every worker to make his or her work applicable to freshwater management needs. We hope the reader will share this view and gain much from the publication.

We wish to thank the authors, who responded kindly to requests to present a paper and then make it available for publication; the chairpersons, for their pleasant and able control of each session; Drs M.E. Livingston, D.S. Roper, and J.D. Stark, for helping with both the selection of speakers and the organisation of the seminar; Miss J.E. Hewitt, for help with editing the manuscripts and draughting the figures; Mr C.J. Milmine, for his invaluable administrative assistance; and Judy Brighting and Mary Clarke, for the many hours of typing.

Rick Pridmore
Bryce Cooper
SEMINAR ORGANISERS

SESSION 5

MONITORING OF MACROINVERTEBRATES

Biological monitoring in freshwaters: proceedings of a seminar. Eds. R.D. Pridmore and A.B. Cooper. Water & Soil Directorate, Ministry of Works and Development for the National Water & Soil Conservation Authority, Wellington, 1985. Water & Soil Miscellaneous Publication 83.

RUNNING WATER ECOSYSTEMS

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ABSTRACT

Running water ecosystems are characterized by 2 contrasting functions; the transportation of materials and the processing of organic matter. Their relative importance depends partly on the presence of instream retention devices and is affected by geology, climate, catchment vegetation, and land use. Hard and soft water streams often differ in productivity while acidification may affect decomposition and production processes and is a subject of current concern.

Recent work has emphasized the processing function of stream ecosystems. This will be discussed with particular reference to New Zealand streams where stone surface organic layers are hypothesized to be major sites of carbon uptake and invertebrate feeding. The relative importance of physical factors and biological interactions in structuring benthic invertebrate communities may be largely a function of the "harshness" of the environment. This idea and the potentially useful intermediate-disturbance hypothesis will be considered.

INTRODUCTION

Streams and rivers are formed by erosional processes and serve to channel water and transport it to the sea. Organic and inorganic materials from a variety of sources also are entrained by stream water, and they provide the basic nutrients and sources of energy which are trapped and used by the aquatic biota.

The character of a stream is determined by conditions in its catchment, including geology, soils, climate, slope, vegetation, and land use. These

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factors can vary markedly and give streams their individuality which is reflected in community structure and productivity. Despite such differences, running water ecosystems possess a number of common features with respect to their functional organization (Fig. 1). The relative importance of different processes is a consequence of local factors.

CARBON SOURCES

The energy or carbon resources utilized by stream communities can be elaborated outside the stream (allochthonous) or produced within the aquatic system itself (autochthonous). In small, forested streams with closed canopies, inputs of allochthonous materials are large whereas instream primary production is small. This is because little incident light reaches the streambed. Not unnaturally, the biological communities of such streams are dependent on terrestrially derived organic matter as their principal energy sources (Fisher & Likens, 1972; Rounick et al., 1982).

In larger streams and rivers, more light reaches the bed and the potential for instream primary production is greater. An increase in the importance of autochthony downstream is a basic feature of most general models of river ecosystems (e.g., Cummins, 1975) and is apparent in the Devils Creek system, Reefton (Winterbourn et al., 1984) where increasing use of algal carbon at downstream sites was indicated by stable carbon analyses. Where headwater streams are not forested, however, different longitudinal patterns can be expected (Minshall, 1978; Naiman, 1981; Winterbourn et al., 1981).

ALLOCHTHONOUS CARBON SOURCES

Organic material entering streams can be separated into 2 broad categories; dissolved and particulate matter. Dissolved organic matter (DOM) is operationally defined as that component which can pass through a 0.45 µm filter and consequently includes a colloidal fraction (Lock et al., 1977).

Particulate organic matter (POM) is subdivided into a number of size fractions by most workers (see e.g., Boling et al., 1975), a basic division being into coarse (CPOM) and fine (FPOM) materials (i.e., >1 mm<). CPOM includes twigs, branches and leaves whereas major components of FPOM may be insect frass, invertebrate faecal pellets and finely

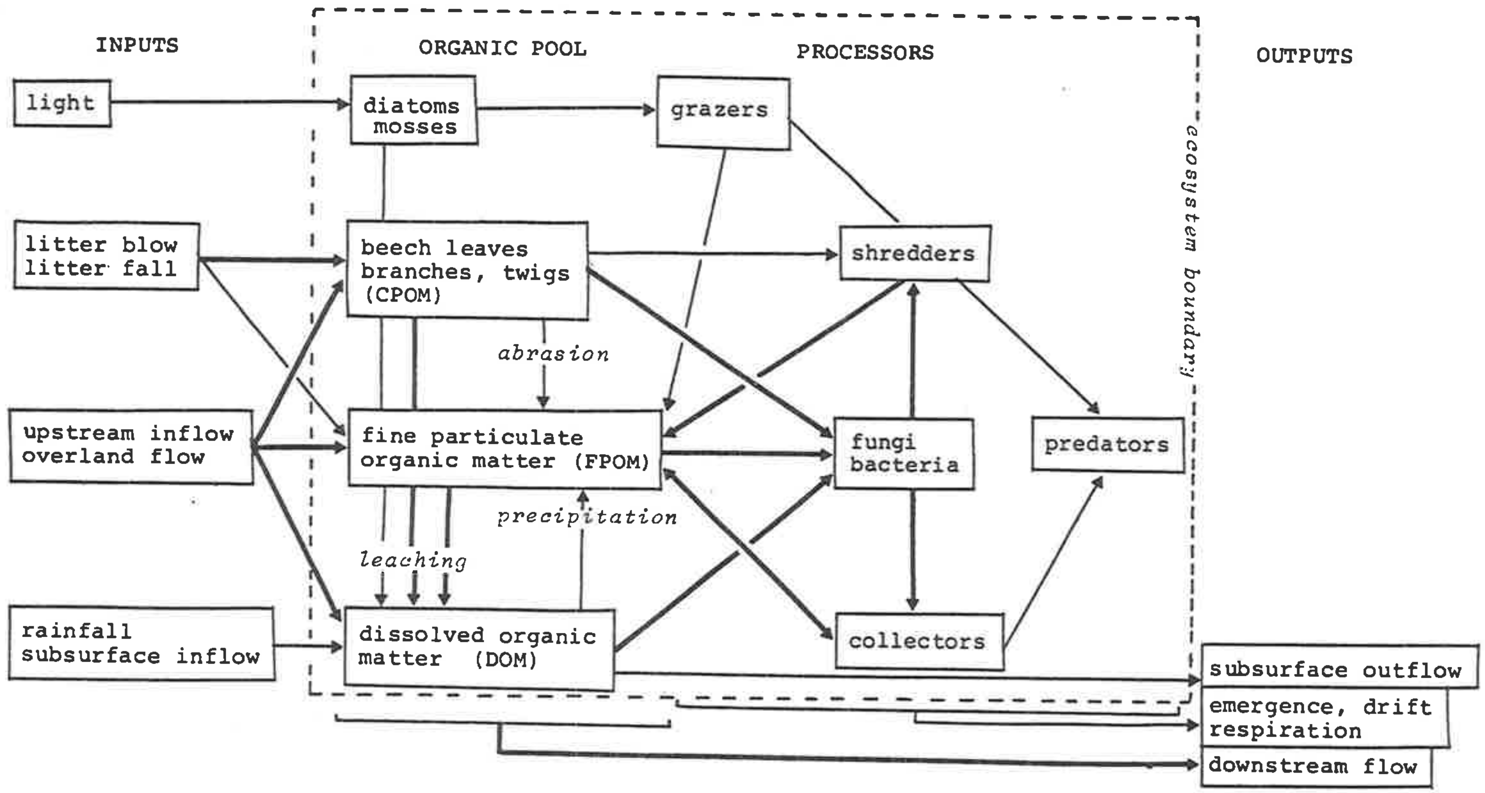

Fig. 1: A simplified model of a beech forest stream ecosystem showing the principal biological components, energy sources, and carbon pathways. After McCammon (1978).

decomposed litter derived from within and outside the water course. Most particulate material either falls or is blown into streams and it may exhibit seasonal input patterns.

Although less apparent, DOM is usually the major form in which organic carbon enters streams. For example, Fisher & Likens (1972) and McCammon (1978) calculated that it represented 46 and 47%, respectively, of total annual organic energy input to small forested streams, and Baekken et al. (1981) estimated that over 95% of the inputs to a weir basin in Norway comprised DOM in transport. A high porportion of DOM is present in groundwater which feeds into streams (Hynes, 1983). Leaf leachates may provide seasonal pulses of DOM in deciduous forest streams (McDowell & Fisher, 1976).

DOM in stream water has been poorly categorized chemically but it is common for a high proportion to be rather refractory, fulvic acid-like material of high molecular weight. Stream water concentrations typically lie between 1 and 15 g carbon m^{-3}, with those in many streams draining small, forested catchments being at the low end of this range. Diel and seasonal variations in concentration have been reported on occasions, but it is clear that streams differ considerably in this respect.

AUTOCHTHONOUS CARBON SOURCES

The principal groups of plants growing in running waters are algae, mosses and macrophytes. Attached algae and mosses predominate in forested streams, and where substrata are mainly eroding materials there is little opportunity for rooted plants to establish. Some diatom genera are typical of poorly lit habitats whereas green filamentous algae are most prevalent in open streams with higher nutrient levels. The importance of light (and sometimes nutrient availability) on algal production and community composition is often seen following the clearfelling of small forested catchments (see e.g., Lyford & Gregory, 1975; Murphy & Hall, 1981). Grazing by benthic invertebrates (Lamberti & Resh, 1983) and particularly abrasion and sediment entrainment during floods (Rounick & Gregory, 1981) also can have pronounced effects on algal standing crops and rates of primary production.

In addition to their obvious producer role, aquatic plants contribute to the particulate detrital pool following their death, and living plants are a source of DOM in the form of exudates. Kaplan et al. (1980) showed that algal exudates could bring about increases in the dissolved organic carbon concentration of stream water, and our stable carbon isotope studies (Winterbourn et al., 1984) indicate that algal exudates are taken up by heterotrophic microorganisms which in turn are ingested by invertebrates.

FATES OF ORGANIC MATTER IN STREAMS

Upon entering a stream, organic matter can be stored, transported or processed. Measures of all 3 are needed for the construction of organic matter budgets but most research effort has been expended on studies of organic matter processing.

Leaf processing

In small, forested streams, leaf litter usually forms the major particulate input and can represent a significant energy resource utilized by members of the invertebrate community. This is most likely to be true in low gradient streams which retain leaf material for long enough to enable decomposition to proceed. Leaf breakdown is brought about by a combination of physical and biotic processes. Abrasion, although difficult to assess, is likely to be an important fragmenting process especially in turbulent, stony streams. Feeding by insect larvae and other invertebrates, either by chewing or rasping surfaces, also results in particle size reduction, while decomposition is effected by colonizing microorganisms. Hyphomycete fungi have been shown to be the main colonizers of deciduous leaves in several North American studies (see e.g., Arsuffi & Suberkropp, 1984) but bacteria appear to be the primary decomposers of *Nothofagus* leaves in this country (Davis & Winterbourn, 1977). Breakdown and decomposition of leaves of other indigenous plants has not been examined in New Zealand streams but casual observations indicate that for many species it is almost certainly slow. Also, few invertebrates appear to feed on them perhaps in part due to their thick resistant cuticles and an apparent paucity of fungal colonizers which make leaves more palatable and nutritious to insect larvae. Fifty percent weight loss of *Nothofagus solandri* leaves held in streams at Cass and Craigieburn ranged from 10 weeks to over 22 weeks (Rounick & Winterbourn, 1983) compared with 6 weeks for willow leaves (*Salix babylonica*) kept in identical bags in the Avon River, Christchurch (Collier, 1984).

DOM uptake and stone surface organic layers

Primary sites of DOM uptake are the surfaces of stones, logs, leaves, and other sediments. Although most workers have emphasized biotic uptake mechanisms, principally by microorganisms, recent work by McDowell (1982) in Bear Brook, a moderately acid stream in New Hampshire, indicated that physical adsorption accounted for most DOM removal from the water column. Observations of surface and subsurface sediments in low to neutral pH streams in the Ashdown Forest, southern England lend support to this contention (Winterbourn *et al*., in press), but whether this is peculiar to acid streams or is a more general phenomenon has yet to be established.

Stone surface organic layers (biofilms, epilithon) are a combination of algal, fungal and actinomycete filaments, diatoms, bacteria, and their products, trapped plant fragments, silt, and "amorphous detritus". The relative and absolute abundance of different components depends on a variety of factors including light penetration, geology, water chemistry, physical substrate stability, temperature, and grazing pressure. Organic layers are dynamic entities (Fig. 2) which may play significant roles in nitrogen and carbon cycling, metal adsorption and bacterial digestion within stream ecosystems (Lock *et al*., 1984).

Stone surfaces also provide habitat for benthic invertebrates, particularly insect larvae, many of which feed by browsing or grazing over them. In fact, a central role for stone surfaces as sites of energy transfer in forested and open, upland streams has been hypothesized by Winterbourn *et al*. (1981). This is because they appear to be persistent, "reliable" carbon resources (Cowie, 1980) unlike non-attached particulate matter which is poorly retained by many mountain streams. In lowland, New Zealand streams, most of which are strongly modified, this scenario will be less applicable as low physical gradients result in slower flows and a greater occurrence of depositing substrata. As a result, fine, non-attached POM accumulates and provides food for a primarily deposit-feeding or suspension-feeding fauna.

INVERTEBRATE ASSEMBLAGES

The invertebrate faunas of stony streams in New Zealand are dominated by insect larvae. Winterbourn *et al*. (1981) considered that a nucleus of common genera and species tend to dominate the benthos of many streams, a claim which has been reinforced by subsequent studies in divergent localities and environments (Mt Egmont ring plain, Taranaki Catchment Commission, 1984; Otago hill-country tussock land, Ryder 1983; Kaimanawa Ranges, collections by P. Mylechreest, Wildlife Service, Department of Internal Affairs, Rotorua).

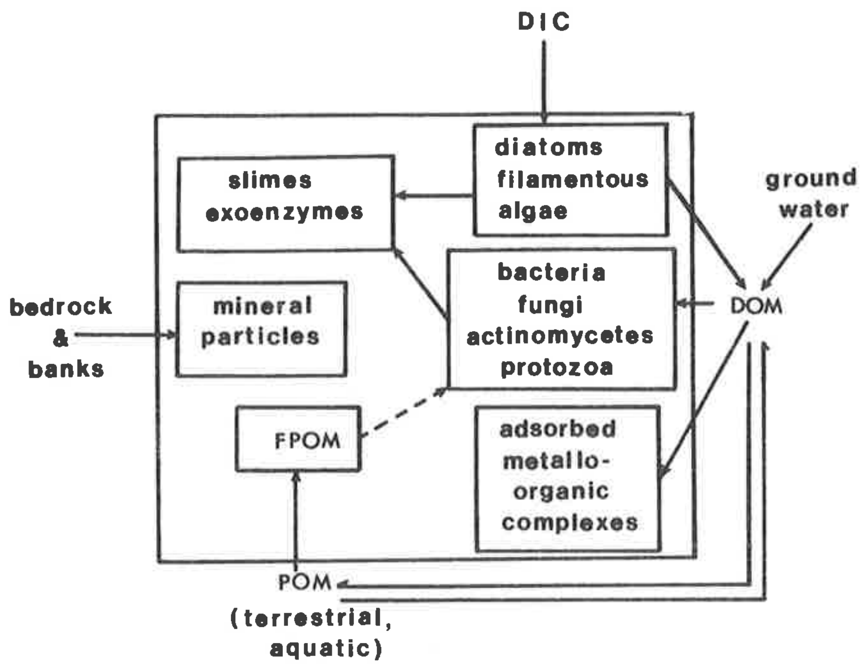

Fig. 2: A compartmental model of the stone surface epilithon showing proposed carbon sources and internal pathways. DIC = dissolved inorganic carbon.

The integrity of stream "communities" is a subject of some debate with views ranging from the highly integrated (organismic) concept advanced by Vannote et al. (1980) to the loosely associated ("the transitory result of many population dynamics") view of Reice (1980). Factors regulating populations may differ depending on the physical harshness of the environment (Peckarsky, 1983) such that biological interactions should be of least importance in unstable, climatically unpredictable streams like many in New Zealand. On the other hand, biotic interactions such as competition and predation may well be important in more "benign" and stable situations. Some support for this idea is provided by Hildrew et al. (1984) whose findings indicate that fish predation may play a significant role in structuring benthic communities in some small, low gradient English streams. Experimental studies are sorely needed to test the Peckarsky hypothesis and other theories of community organization.

In stream ecosystem studies where a functional or process-oriented rather than purely descriptive approach is desirable, it has become common for faunas to be analysed in terms of abundance, biomass or production of functional feeding groups (Cummins, 1973) or feeding guilds (Hawkins _et al_., 1982). These are defined primarily on the basis of feeding mechanisms and secondarily on type of food eaten, and include shredders (large particle detritivores), filter-feeders, scrapers, gougers, collectors and predators. A lack of uniformity in terminology used by different workers illustrates the inherent difficulty of defining feeding types explicitly and serves a warning that such classifications should be used with care. In fact, many stream insects have been described as generalists (Cummins, 1973) and depending on circumstances some can be fitted into different groups. We have used the term collector-browser to describe many of the common New Zealand species that inhabit stony streams and whose gut contents are dominated by fine, often amorphous detritus and diatoms. Animals included in this group include _Deleatidium_, _Nesameletus_, most of the small gripopterygid and notonemourid stoneflies and a number of the smaller, cased caddisflies. Depending on the habitat in question, the collector (gathering of loose fine particles) or browser (stone or leaf surface grazing) mechanism can be expected to predominate, while some nominal collector-browsers like _Olinga feredayi_, _Austroperla cyrene_ and _Zelandoperla fenestrata_ can also act as shredders. As a general rule, collector-browsers are the numerically dominant group in most New Zealand streams while shredders are often absent.

Where an aim of a study is to determine the relative importance of different carbon pathways within a stream ecosystem (i.e., the sources of organic matter supporting the benthic community) or to discriminate between the use of living and dead plant material, methods other than functional group analyses of fauna will need to be employed. The most obvious of these is gut content (or faecal) analysis, although stable carbon isotope analysis shows promise for allochthonous/autochthonous discrimination (Rounick _et al_., 1982).

MODIFIED STREAM SYSTEMS

As one moves down a river system, physico-chemical conditions change along with the size of the watercourse and the nature of riparian vegetation. The benthic fauna also changes in response to at least some of these (ill-defined) factors. Sometimes these changes in faunal composition are gradual as in Devils Creek (Cowie, 1985) although inflowing tributaries can act as significant modifiers of stream ecosystems (Bruns et al., 1984) and the presence of dams and reservoirs severely disrupts longitudinal patterns (Hauer & Stanford, 1982).

Man's activities in the catchments of streams and rivers can have varying effects on the structure and function of running water systems. Removal of riparian vegetation opens up the stream channel, alters the nature and pattern of allochthonous inputs, and frequently will result in increased instream primary production. Also, water temperature may rise and lead to changes in seasonal patterns and rates of decomposition as well as the distribution, development and life histories of invertebrates and fish.

Modifications to flow regimes and discharge patterns can alter the particle size distribution and physical stability of the streambed and, because many species of animals and plants are confined to a limited range of substrate types, changes in the benthic fauna can be expected (Hynes, 1970). The effects of silt deposition are likely to be particularly severe because of its smothering and infilling capacity. Although siltation is a major and expected consequence of dam building (Gray & Ward, 1982) and logging activities, particularly road construction (Griffiths & Walton, 1978), its effects may only be temporary as in the Maimai Experimental Area (Winterbourn & Rounick, in press). This will depend on the discharge regime, particularly the frequency and extent of storm events and subsequent management practices.

Finally, effects of agriculture on running waters include increases in bacterial and viral concentrations, soil erosion, elevated turbidity and enrichment resulting from nutrient runoff (Dance & Hynes, 1980). Improvements in soil drainage are reflected in altered stream flow patterns and both dissolved and particulate organics entering streams will change in quantity and "quality". The natural microbial communities on stones differ in their capacity to take up DOM from different sources (e.g., leaves, algae, soils) (Dahm, in press) and as a result changes in their composition and processing capacity can be expected. Experimental, process-oriented research on carbon pathways in streams is enabling stream ecologists to better understand the consequences of catchment changes.

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THE USE OF MACROINVERTEBRATES IN THE ASSESSMENT OF POINT SOURCE POLLUTION

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ABSTRACT

This paper illustrates the use of macroinvertebrates in the assessment of point source pollution in running water ecosystems. The effects of organic and toxic pollution on macroinvertebrates communities are described and contrasted using examples drawn from studies of the Wainuiomata River, Wellington, and the Waitaia Stream, Coromandel Peninsula. Structural and functional approaches to macroinvertebrate community analysis are compared.

INTRODUCTION

Macroinvertebrates are indicators of the type and relative magnitude of polluting inputs to freshwater ecosystems. Being fairly sedentary and easy to sample, macroinvertebrates are more useful than highly mobile fishes in assessing point source pollution. In addition, they are easier to identify than algae, and changes in their community structure caused by polluting inflows may persist for a considerable time, since many macroinvertebrates have a life cycle of a year or more. Thus semi-annual sampling is often adequate to detect significant changes in macroinvertebrate populations (Hirsch, 1958; Hawkes, 1964; Hynes, 1965; Gaufin, 1973; Penny, 1976; Welch, 1980; Winterbourn, 1981).

EFFECTS OF ORGANIC AND TOXIC POLLUTION ON THE MACROINVERTEBRATE COMMUNITY

The main effects of organic pollution are oxygen depletion of the water and sediments, increased levels of suspended organic material, deposition of organic material in the sediments, possible development of sewage fungus, proliferation of periphyton and macrophyte communities, and changes in

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species composition and abundance of consumer organisms, in particular macroinvertebrates. (Hynes, 1960; Mason, 1981).

Under septic conditions the macroinvertebrate community is largely eliminated except for tubificid and naiad worms. Where oxygen is depleted, but not exhausted, macroinvertebrate diversity is reduced, but the abundances of tolerant organisms increase in response to the increased food supply (Hynes, 1960; Mason, 1981). Table 1 shows changes in abundance of macroinvertebrate genera above and below an outfall discharging treated sewage effluent to the Wainuiomata River near Wellington. Immediately below the outfall (station 7) oxygen sensitive browsers of the algal film including the mayflies Deleatidum and Zephlebia and the cased caddis Beraeoptera and Helicopsyche were absent, whereas oxygen tolerant forms such as the pulmonate snail, Physa, the mothfly, Psychoda, and the purse caddis, Oxyethira, increased in numbers. Downstream at station 8, and below, large populations of snails, midges (chironomids) and elmid beetles flourished in the dense beds of macrophytes which grew in the organically enriched sediments. Once mineralisation of the wastes had been completed and the diurnal oxygen regime restored to saturation (at station 11) oxygen sensitive taxa such as Deleatidium returned to the community. Nutrient levels did not return to background levels, however, and the recovery zone was one of enhanced productivity in which the abundances of the cased caddis, Pycnocentrodes, and elmid beetles, Hydora, were higher than at station 3 (Gibbs & Penny, 1973; Penny, 1976).

The type and extent of the changes in the macroinvertebrate community which occur in response to toxic pollution are somewhat different from the effects of organic pollution. Toxic effects depend on the concentration and chemistry of the toxin in relation to the tolerance range of the species present and other environmental factors (Bryan, 1976; Welch, 1980; Mason, 1981; Hart, 1982).

As with organic pollution the overall effect of toxic pollution is to reduce the species diversity of the macroinvertebrate community and to change the relative abundance of tolerant organisms. At very low levels of toxicity, biomass may remain the same or even increase slightly. Some tolerant macroinvertebrates may increase due to a reduction in competition

TABLE 1 : Mean number of individuals in 27 macroinvertebrate taxa collected in the Wainuiomata River from stations above and below a sewage treatment plant outfall over 5 consecutive months (April-August 1971). Three samples were taken from each station with a Surber sampler. (Sample area = 0.3 m², Surber mesh = 750 µm). Data from Penny (1976).

CLASS or ORDER	Family	Species	3	6(RHS)	7(LHS)	8	11	Common name	Functional feeding group
		Station	3	6(RHS)	7(LHS)	8	11		
		Distance below outfall (km)	-3	0.02	0.02	0.8	12		
		Instream vegetation	gfa	gfa	gfa,sf	ec,gfa	Moss		
		Relative abundance	*	**	** ***	****	***		
GASTROPODA	Hydrobiidae	Potamopyrgus antipodarum	114	524	73	1461	205	snail	collector-browser
	Physidae	Physa spp.	2	27	831	591	114	snail	"
	Ancylidae	Ferrissia dohrnianus	4	13	52	88	<1	limpet	"
INSECTA									
EPHEMEROPTERA	Oligoneuridae	Coloburiscus humeralis	3	<1	0	0	<1	mayfly	filter-feeder
	Leptophlebiidae	Deleatidium spp.	82	9	<1	<1	14	"	collector-browser
		Zephlebia spp.	12	9	<1	<1	<1	"	"
PLECOPTERA	Grypopterygidae	Zelandoperla sp.	0	0	0	0	29	stonefly	"
		Zelandobius furcillatus	22	13	6	<1	<1	"	"
TRICHOPTERA	Hydropsychidae	Aoteapsyche colonica	63	63	<1	<1	63	caddisfly	filter-feeder
	Hydroptilidae	Oxyethira albiceps	46	96	200	138	40	"	algae-feeder
	Hydrobiosidae	Hydrobiosis umbripennis	1	2	1	3	2	"	predator
		H. parumbripennis	3	<1	0	0	<1	"	"
		Neurochorema confusum	6	<1	0	0	0	"	"
	Leptoceridae	Hudsonema amabilis	0	0	0	0	3	"	"
	Helicopsychidae	Helicopsyche albescens	37	<1	0	0	0	"	collector-browser
	Conoesucidae	Beraeoptera roria	17	<1	0	<1	<1	"	"
		Olinga feredayi	12	0	0	0	2	"	"
		Pycnocentria evecta	2	<1	0	0	5	"	"
		Pycnocentrodes aureola	42	41	11	14	433	"	"
COLEOPTERA	Elmidae	Hydora picea	19	76	39	145	190	beetle	"
DIPTERA	Chironomidae	(several species)	154	86	199	77	49	midge	"
	Simulidae	Austrosimulium tillyardianum	7	1	1	1	4	sandfly	filter-feeder
CRUSTACEA	Psychodidae	Psychoda spp.	0	3	173	3	0	mothfly	collector-browser
AMPHIPODA	Gammaridae	Phreatogammarus helmsii	0	14	1	0	62	amphipod	"
OSTRASCODA	Cypridae	Herpetocypris pascheri	8	12	7	28	2	ostracod	filter-feeder
OLIGOCHAETA	Turbificidae	(several species)	1	<1	7	17	1	annelid worm	burrowing scavenger
HIRUDINEA	Glossophoniidae	Glossophonia heteroclita	<1	0	<1	3	0	leech	predator
		Total Number	664	994	1602	2581	1223		

Key to instream vegetation :

gfa = green filamentous algae
sf = sewage fungus
ec = Elodea canadensis

*	sparse
**	moderate
***	abundant
****	very abundant

for food and space, and/or reduced predation, and this may compensate for the reduction in abundance of sensitive taxa. In most cases of toxic pollution, however, a reduction in diversity is accompanied by a reduction in biomass. This is in direct contrast to the large increase in biomass associated with mild forms of organic pollution (Welch, 1980; Mason, 1981).

This basic difference between organic and toxic pollution is illustrated in the ordination diagrams (Fig. 1) prepared using the method of Hocutt (1975). In these diagrams, control and impacted sites in an organically polluted river (Wainuiomata) and a toxically polluted stream (Waitaia Stream, Coromandel Peninsula; Penny, in prep.) are compared with respect to total number of taxa and organism abundance. The vectors dividing Quadrants I to IV represent the mean values for control data for these 2 parameters. Control coordinates tend to aggregate around the intersection of the vectors whereas impacted coordinates are confined to Quadrant I (organic pollution) and Quadrant IV (toxic pollution). Thus the pollutants exerted markedly different effects on the numbers of organisms present in the 2 examples, but in both there was a reduction in the number of species. In the Wainuiomata this was due to reduced oxygen and the inhibiting effect of blanketing algae, and in the Waitaia this was due to either an inhibiting or a toxic effect of acid mine drainage.

POLLUTION ASSESSMENT

An understanding of the type and severity of a particular pollution is gained by macroinvertebrate community analysis. The traditional approach is to partition the community into taxonomic groups, to species level where possible. The information is presented as a species inventory together with relative abundances, and is usually summarised as total number of species and total number of organisms. This information may be further condensed into a variety of biotic and diversity indices, (e.g., Beak, 1964; Wilhm & Dorris, 1968; Wilhm, 1972) and may be accompanied by a discussion of the community changes.

Severe pollution invariably reduces the number of species, but the main effect of mild pollution may be a change in relative abundance without a great loss of species. This latter effect is often associated with a change in the type and abundance of food rather than a direct inhibitory effect arising from oxygen depletion or concentrations of toxic substance.

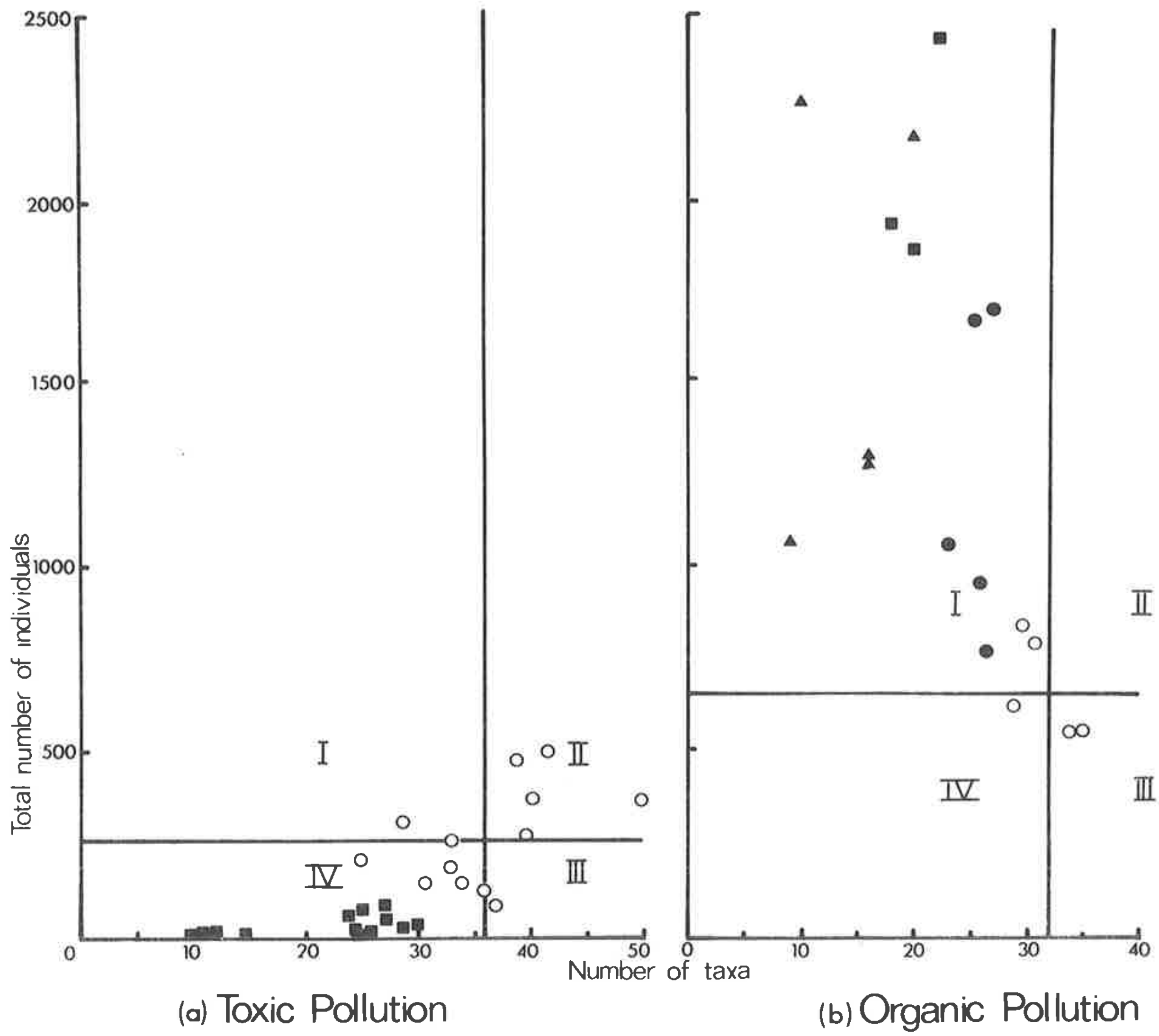

Fig. 1: Effect of toxic and organic pollution on macroinvertebrate abundance and species richness.

a Acid mine pollution in Waitaia Stream, North Island, New Zealand (toxic effect). o, control sites; ■, impacted sites.

b Treated sewage pollution in Wainuiomata River, North Island, New Zealand (organic effect). o, control sites; ▲, heterotrophic production; ■, autotrophic production; •, recovery zone.

These "food web" effects can be revealed by functional (or trophic) analysis. Macroinvertebrates are partitioned into functional feeding groups according to their mode of feeding. Some broad categories are : shredders (coarse particle feeders); collector-browsers (fine particle feeders including scrapers and algal grazers); predators; filter-feeders; burrowing scavengers.

The relative abundances of these groups and the number of taxa within each group provides some indication of the sources of food, the pathways for energy transfer, and the variety of opportunity for exploiting the food source.

TABLE 2 : Mean numbers and number of species of macroinvertebrates collected from stations above and below a sewage treatment plant outfall discharging into left hand side (LHS) of Wainuiomata R., Wellington Surber samples were collected monthly from April-August 1971. Data from Penny (1976).

Distance below outfall (km)	-3	0.02 (RHS)	0.02 (LHS)	0.8	12
Station	3	6	7	8	11
Number of samples	15	5	5	15	15
Number of collector-browsers	550	912	1586	2523	1143
Number of species	21	17	13	17	19
Number of filter-feeders	81	76	8	29	69
Number of species	6	5	2	3	6
Number of burrowing scavengers	1	<1	7	17	1
Number of species	NR	NR	NR	NR	NR
Number of predators	10	3	1	3	2
Number of species	11	5	3	3	3
Total number of macroinvertebrates	664	994	1602	2581	1223
Number of species	44	29	24	30	35

NR = not recorded

Thus in the Wainuiomata River (Table 2) the major effects were an increase in abundance and a decrease in species diversity within the collector-browser functional feeding group. Other functional groups within the stream decreased in abundance and diversity indicating that conditions favourable to some collector-browser taxa were inhibitory to other taxa within the group and to other types of feeders.

In the Waitaia Stream (Table 3), by way of contrast, abundance and diversity of all functional feeding groups decreased. Abundant detritus was available as food in this stream; thus it appeared that a direct toxic effect rather than lack of food caused the decrease or absence of shredders, collector browsers, and filter-feeders.

TABLE 3 : Mean numbers and numbers of species of macroinvertebrates at stations above and below the confluence of Control and Mine Branches of Waitaia Stream, Coromandel Peninsula. Samples were collected between March 1982 and January 1983 (Penny, in prep.).

	Control Stream		Confluence	Mine Stream		Mine
Distance from confluence (km)	1.0	0.01	0	0.01	1.0	1.8
Station	W3	W4	W6	W7	W8	
Number of samples	27	15	18	15	27	
Number of collector-browsers	198	273	100	105	9	
Number of species	30	25	25	22	14	
Number of filter-feeders	17	20	10	1	2	
Number of species	5	5	3	4	2	
Number of shredders	3	3	4	<<1	<<1	
Number of species	3	3	3	3	1	
Number of predators	8	15	8	6	<1	
Number of species	11	9	11	5	7	
Total number of macroinvertebrates	226	311	122	112	11	
Number of species	49	42	42	34	24	

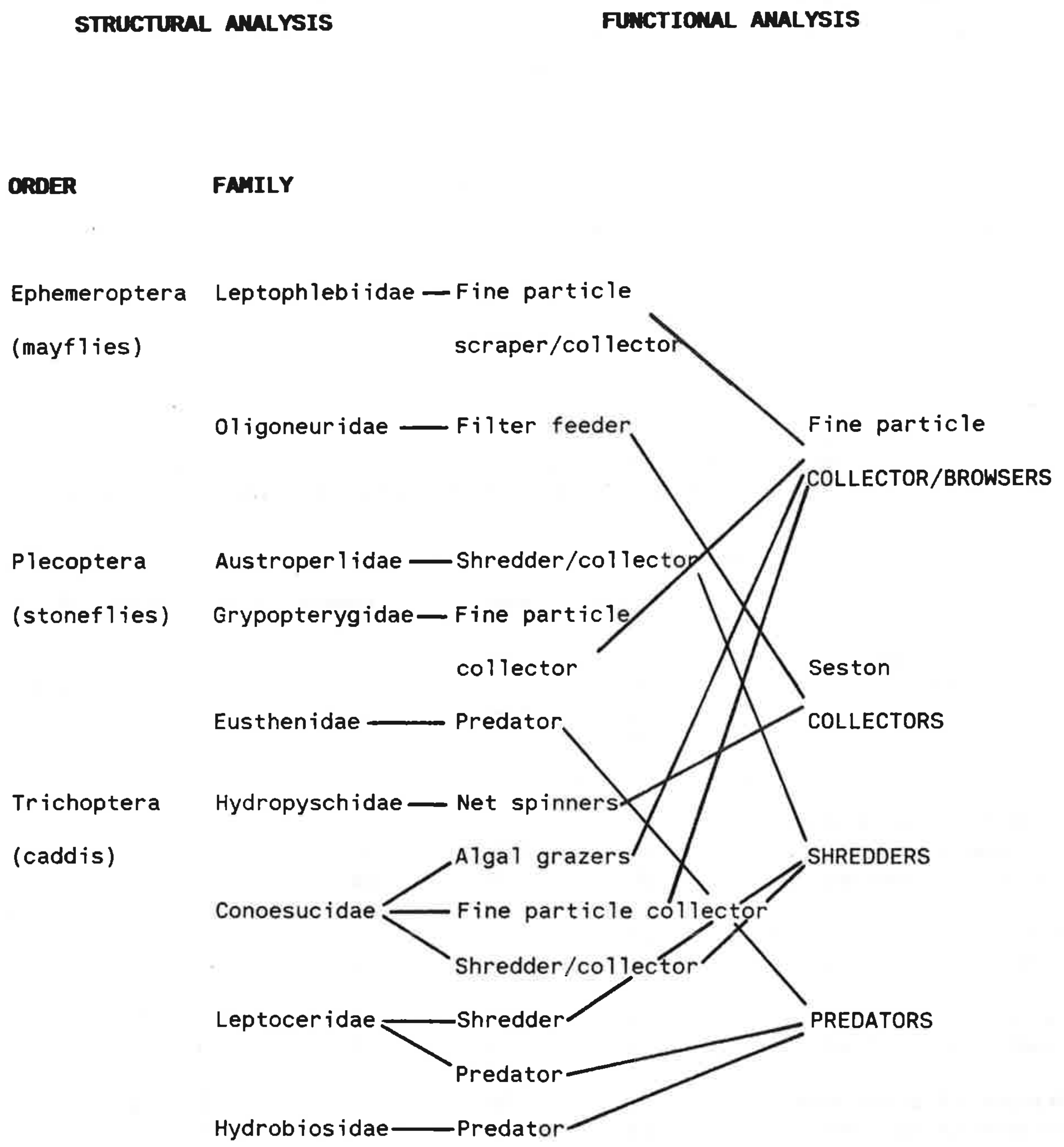

Fig. 2: Relationship between structural and functional analysis of 3 order of macroinvertebrates.

In most cases family groupings provide a reasonably consistent functional, as well as structural summary, of macroinvertebrate data (Fig. 2). Analysis to the family level of the large generalist collector-browser group is particularly useful since this group encompasses taxa with a wide variety of tolerance to organic pollution. Once broken down to family groups, sensitive and tolerant forms can be distinguished. In the Wainuiomata example, collector-browser families could be assigned as follows :

Tolerant	Sensitive
Physidae (snails)	Leptophlebiidae (mayflies)
Hydrobiidae (snails)	Conoesucidae (caddis)
Hydroptilidae (caddis)	Helicopsychidae (caddis)
Elmidae (beetles)	

Thus families may be the most convenient groups to use for showing the effects of moderate to severe pollution and may be adequate for routine compliance monitoring. Additional information is required, however, to fully understand the effects of mild organic pollution, or to distinguish sublethal toxic effects from other environmental variables. In these situations, the identification of individual species is necessary, and an understanding of their life history, ecology, and sensitivity to dissolved oxygen and toxins is helpful. Data for species which are tolerant of one type of pollution, but sensitive to another are particularly valuable for "forensic" investigations (Brinkhurst, 1974). Interpretation of such data also requires a catchment description and an assessment of the substratum (e.g., stability, particle size and proportion). It is recommended that a standard checklist be devised and used for this assessment.

CONCLUSION

In investigative baseline and impact studies all species in the macroinvertebrate community should be identified and counted where possible. For routine monitoring it is preferable that a species inventory be kept and changes in dominance recorded, but it may be sufficient to enumerate and make statistical comparisons at the family level.

It is suggested that in reports macroinvertebrate data be summarised in simple tables and graphs, accompanied by a short explanation of the community changes. These should use common names and functional feeding group terminology although the species inventory should be available for reference.

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THE IMPACTS OF FORESTRY UPON STREAM FAUNA

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ABSTRACT

Forestry operations such as roading, logging, and burning can have major impacts upon stream environments. These impacts include changes in the hydrologic regime (higher base flows, "flashier" flood peaks), changes in water temperatures, changes from an allochthonous to an authochthonous energy base, an increase in debris in the stream, and, perhaps most significantly, sedimentation. Excess sediment fills the interstitial spaces between stones and in doing so changes both the abundance and specific composition of the invertebrate community as well as smothering fish redds and eggs. There is some evidence that in New Zealand these changes are relatively greater in stable than in unstable streams.

In any given catchment the logging practices adopted are a major factor in determining impacts upon the stream environment. Leaving riparian strips alongside streams is a simple means of helping to limit such impacts. In New Zealand most catchments are logged using tracked bulldozers; this requires roads to be built around catchment contours and unless these are constructed carefully they can lead to excess sediment entering streams. Other practices such as root raking and burning can also affect the stream biota. One critical factor that cannot be controlled is the occurrence of storm events; most sediment enters streams following large storms and no matter how carefully logging is carried out, such events are still likely to have major impacts upon streams.

INTRODUCTION

About 25% of New Zealand's land surface is covered by forest, much of it protection forest. Exotic plantations account for about 8% of this area, or 2% of the total land cover. Forestry is currently our fifth largest

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export earning industry. At present some 18,000 ha of forest are logged each year; this will soon increase as much of the country's radiata pine resource will be logged during the 1990's.

This paper first outlines the main logging methods used in New Zealand before reviewing the various impacts of logging operations on stream invertebrate and fish populations [for a more thorough review of the impacts upon fish see Morgan & Graynoth (1978)]. This information is then used to draw general conclusions as to how sympathetic forestry practices and perhaps more stringent control of logging, can be used to reduce such impacts.

LOGGING TECHNIQUES

The choice of logging technique used depends primarily on land slope and stability, and economics. The main method used in this country involves either the tracked bulldozer or rubber tyred skidder. Tracks are constructed around the catchment boundary and at intervals along catchment contours. Logs are then towed out behind the skidders.

On particularly unstable or steep lands a skyline hauler may be used. A hauler pad is set up on the catchment boundary and heavy chains are used to haul logs to this pad, from which point they are transported out by trucks. The main benefit of this method, as compared to using a skidder, is that the land surface of the catchment remains relatively undisturbed. However, it is a more expensive logging method.

A number of other management decisions taken at the time of logging can also have important consequences for stream biota, e.g., will a riparian strip be left along the larger watercourses in a catchment, or will a catchment be selectively logged? Post-logging management decisions also can be critical, for instance whether or not "slash" is burnt or the catchment is root-raked prior to planting out. A discussion of how such practices affect the stream biota follows.

IMPACTS OF LOGGING

Impacts of logging upon the invertebrate and fish fauna arise largely from :

a changes in stream hydrologic regime;

b sedimentation;

c changes in stream morphology and debris dams;

d changes in the energy base to stream communities;

e alterations in water temperature.

Each of these is now discussed briefly.

Changes in stream hydrologic regime

Stream flows increase following logging because the quantity of water evaporated from the catchment, both by plant transpiration and through interception by plants, decreases (Bosch & Hewlett, 1982). Both base flows and flood flows increase, but whereas increased base flows could conceivably be beneficial to the fauna of small streams, increased flood flows are likely to be detrimental.

Sedimentation

Sediment entering streams following logging can have major impacts upon stream invertebrate and fish populations. By filling the interstitial spaces between stones, sediment can completely change the substratum character and therefore invertebrate habitat quality. Under such conditions the aquatic invertebrate community often changes from one dominated by mayflies, stoneflies and caddisflies to a situation where more tolerant and/or burrowing organisms such as chironomids and oligochaete worms are more prevalent (e.g., Graynoth, 1979; Cowie, 1984). Such changes are likely to be more marked in stable streams, which in near natural conditions generally have more diverse invertebrate faunas than do unstable streams (see Rounick & Winterbourn, 1982).

Amongst fish, salmonids in particular require silt-free gravels for spawning. Sedimentation can reduce the areas of good spawning gravels available to trout, as well as smother eggs and alevins within redds. In parts of New Zealand such as the Motueka River Catchment, substantial areas of which drain from Golden Downs State Forest, anglers have voiced concern that forestry operations and subsequent sedimentation may have detrimentally affected the spawning success of trout (Richardson *et al.*, 1984). In this context it is important to note that the impacts of sedimentation are not limited to the immediate stream environment - they can also affect larger watercourses further downstream.

Stream morphology

By modifying stream morphology, logging can have further impacts upon the stream biota. For instance, until 5 to 10 years ago in areas such as Golden Downs State Forest, South Nelson, it was common to leave massive debris jams in streams during and after logging (e.g., Graynoth, 1979). Such practices detrimentally affect fish passage and migrations, and perhaps invertebrate passage, and can have major impacts upon streams if the debris jams "burst" during floods suddenly releasing the water accumulated behind them.

Changes to the energy base

Logging a catchment can lead to a decrease in leaf litter inputs to a stream and an increase in light penetration. In combination these factors can result in a change from a primarily allochthonous (out of stream production) to autochthonous (instream production) energy base (Winterbourn & Rounick, in press).

The New Zealand stream fauna has a strong core element of about a dozen genera which are widely distributed in streams draining a variety of forest types, including indigenous plantations, and in open streams (Rounick & Winterbourn, 1982). This means that changing catchment vegetation may have relatively little impact *per se* upon stream invertebrate communities. Such changes may however affect levels of secondary production which are likely to be higher if the canopy is opened up.

The impacts of such changes upon fish communities are little studied in New Zealand.

Water temperatures

High water temperatures can be damaging to fish and invertebrate populations. Salmonids, for instance, are fish of cool temperate waters and temperatures of 22 to 28°C, depending on acclimation, can be fatal to trout (Church *et al*., 1979).

Water temperatures can increase markedly in small streams draining clear-felled catchments. For example, studies carried out in the Maimai Experimental Catchment Area near Reefton showed that on a warm summer's day water temperatures in a clear-felled catchment rose 6 to 8°C more than in an unlogged control catchment, and 5 to 6°C more than in a logged catchment with a riparian strip (pers. comm., C. O'Loughlin, FRI).

Burning of "slash" following logging can lead to water temperatures reaching levels lethal to aquatic life.

DISCUSSION AND CONCLUSIONS

Logging part or all of a forested catchment can be viewed as a major environmental perturbation which can potentially have major impacts upon the stream biota, at least in the short term. Such impacts can be minimised by careful logging practices.

a In skidder logged catchments roads should be carefully constructed to follow contours; this will help minimise sediment inputs to streams.

b In catchments with steep or unstable slopes hauler methods should be used in preference to skidders. Similarly, selective logging using skidders will help limit impacts on stream environments.

c Leaving a riparian strip is a simple and expeditious means of helping minimise the impacts of logging upon streams. Riparian strips may help to :

   i limit direct sediment inputs;
   ii protect streams banks and bank vegetation from logging induced erosion;
   iii prevent water temperatures from becoming critically high;
   iv help maintain the allochthonous energy base of the stream.

   A major problem with riparian strips is that in exposed localities they are very prone to wind-throw.

d Practices such as root-raking, which further disturbs the catchment land surface, burning of "slash" near streams, and leaving log debris in streams are likely to have serious consequences for the stream fauna and should be discouraged.

Three further points are also relevant to this discussion.

a No matter what means are undertaken to protect the stream biota during the logging of hill country catchments, major storms in the post-logging period are likely to lead to major sediment inputs to streams with consequent impacts upon the biota. Such storms can be of benefit, however, if they scour sediment from streams, as has happened in some of the Maimai catchments (pers. comm., M.J. Winterbourn, University of Canterbury).

b In the longer term, stream invertebrate communities seem able to recover from the short-term detrimental impacts of unsympathetic logging practices. For instance, Graynoth (1979) found that the invertebrate fauna of Gilbert Creek, Motueka River catchment was badly affected by logging in 1973-1974, yet by 1981 the invertebrate fauna had recovered to one typical of similar streams in that area (Cowie, in press).

c Finally, I believe that there is a good case for more stringent control of logging operations. A water right, with its associated objection and appeal procedures, is required for many ventures which often have minimal impacts upon soil and water values, e.g., a stormwater discharge operating discontinuously to a small, lowland stream. However, the logging of substantial exotic catchments, with potentially severe effects upon soil and water values, is often controlled only by the agency carrying out that logging. Although recognising that there is now a much increased environmental awareness within the forestry industry, I suggest that some independent appraisal of logging proposals could only be beneficial to soil and water conservation in New Zealand.

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LIFE HISTORY PATTERNS AND THEIR INFLUENCE ON MONITORING INVERTEBRATE COMMUNITIES

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ABSTRACT

Life history patterns are influenced to some extent by the composition of invertebrate communities. Communities in New Zealand lakes are species-poor with a relatively low contribution by insects, whereas those in streams are species-rich with a high proportion of insects. The types of life cycles in New Zealand invertebrate communities are described. Seasonal variation in New Zealand invertebrate communities is low and this is related to life history patterns. A model of seasonality based on northern hemisphere invertebrate studies is described and its applicability to New Zealand stream ecosystems is discussed. It is concluded that the predominance in New Zealand of poorly synchronised life cycles and poorly defined growth cohorts, while presenting difficulties for ecologists, provide resource managers in New Zealand with fewer sampling problems than in countries where seasonality is more pronounced.

INTRODUCTION

The more comprehensive accounts of the invertebrates in New Zealand lakes (Forsyth, 1978; Timms, 1982, 1983) and forest streams (Winterbourn, 1978; Towns 1976, 1978, 1979, 1981a, b, 1983; Cowie, 1980, 1983) cover a range of habitats, altitudes, and latitudes, but apart from some general comments by Winterbourn _et al_., (1981) and Towns (1983), no overview of New Zealand invertebrate life history patterns has hitherto been available. Recent reviews based on northern hemisphere examples include accounts of life history patterns in selected orders such as Plecoptera (Hynes, 1976), Trichoptera (Wiggins, 1977) and Ephemeroptera (Brittain, 1982), and models of how these patterns may have evolved in entire communities (Cummins &

Biological monitoring in freshwaters: proceedings of a seminar. Eds. R.D. Pridmore and A.B. Cooper. Water & Soil Directorate, Ministry of Works and Development for the National Water & Soil Conservation Authority, Wellington, 1985. Water & Soil Miscellaneous Publication 83.

Klug, 1979; Vannote & Sweeney, 1980). However, these accounts pre-dated most work carried out in New Zealand, some of them involve families not found here, and, in some cases, published New Zealand examples appear to have been overlooked. Accordingly, this contribution examines life history patterns from a New Zealand perspective by outlining the patterns found in this country, the ways these differ from patterns most frequently described in the literature, and the effects these patterns have on sampling periodicity.

The invertebrate communities of New Zealand streams are more diverse than those of lakes (see Forsyth, 1978; Towns, 1978; Cowie, 1983) so this review draws most heavily on examples from running waters. In addition, because this seminar is aimed at providing guidelines for sampling, coverage is restricted to macroinvertebrates, which represent the most easily obtained animals.

Because the types of life history patterns that can be encountered in aquatic ecosystems are a reflection of the types of invertebrates present, it is necessary to comment on the composition of these communities in New Zealand. Before doing so, 2 terms require clarification : life histories or life history patterns; and life cycles. Life histories are the overall patterns of growth, emergence and reproduction of populations of any one species (Cowie, 1980), and these patterns can show flexible responses to environmental conditions. Life cycles refer to the changes which occur as organisms grow and reproduce. These changes involve a series of life stages (e.g., larva, pupa, and adult), and are fixed genetically within groups of organisms.

COMPOSITION OF NEW ZEALAND INVERTEBRATE COMMUNITIES

The predominant macroinvertebrates in lakes and streams are oligochaetes, crustaceans, molluscs, and insects. The largest number of taxa recorded in a benthic lake community in New Zealand is 26 taxa from Lake Rotoiti (South Island) (Timms, 1982), of which 13 were insects. With an average of 11.4 invertebrate species in 7 Rotorua lakes (Forsyth, 1978) and 12.4 species in 20 South Island lakes (Timms, 1983), diversity of benthic invertebrates in New Zealand lakes is very low. Higher diversities (75 species) have been recorded from New Zealand lakes on submerged macrophytes (Stark, 1981), but all such faunas lack some groups found overseas, and have few representatives of some groups which are cosmopolitan. For example, fewer than 10 chironomid species have been recorded from most New Zealand lakes (Forsyth, 1978; Timms, 1982), whereas over 50 species have been found in some cool temperate lakes elsewhere (cf. Forsyth, 1978). Similarly, odonatans (dragonflies and damselflies) in New Zealand are represented by only a few species in any aquatic environment (McLellan, 1975; Winterbourn et al., 1981). On the other hand, New Zealand streams harbour a wide diversity of invertebrates, with the Insecta (80-92% of the fauna) (see Towns, 1978; Winterbourn, 1978; Cowie, 1983) represented by 11 aquatic or water-associated orders (Winterbourn & Gregson, 1981). The absence or poor representation of some families in New Zealand streams is apparently compensated by high diversity in other groups, such as caddisflies (Winterbourn et al., 1981). As a result, species diversity in relatively unmodified New Zealand streams can approach 200 species, which is comparable to the species richness recorded in some temperate northern hemisphere streams (Towns, 1978; Cowie, 1983). The invertebrate communities of lakes and streams in New Zealand show surprisingly few regional differences (Winterbourn et al., 1981; Timms, 1982; Rounick & Winterbourn, 1982). In fact, lentic invertebrate communities do not even reflect lake trophic status, although this commonly occurs elsewhere (cf. Timms, 1982). Some differences in sections of stream communities are apparent between the North and South Islands, and in relation to altitude (e.g., stoneflies appear to be more diverse in South Island than North Island streams), but little comparative work of this nature has been conducted.

Since only slight regional differences in community composition occur, general comments on life history patterns can be made with a reasonable degree of confidence.

INVERTEBRATE LIFE CYCLES

Knowledge of the types of life cycles of target organisms is required so that natural periodic absences are not interpreted as responses to environmental disturbance. For example, although numerically dominant in most aquatic ecosystems, very few insects have a fully submerged life cycle (Resh & Solem, 1978), and all New Zealand aquatic insects appear to have at least 1 terrestrial life stage. Because lakes support the highest proportion of non-insect groups, they tend to show a predominance of species with fully aquatic life cycles. These non-insect groups have life cycles varying in complexity (Table 1).

TABLE 1 : Some features of the life cycles found in New Zealand invertebrates.

Feature	Group
NON-INSECT INVERTEBRATES	
Asexual reproduction (fragmentation) and sexual reproduction (cocoons)	Oligochaetes
Brooding of young	Oligochaetes
Parasitic glochidium	Molluscs
Ovoviviparity	Molluscs, Crustaceans
Planktonic larvae	Crustaceans
Protandry	Crustaceans
INSECTS	
Incomplete metamorphosis	Hemiptera, Plecoptera Ephemeroptera, Odonata
Complete metamorphosis	All other orders

The shortest life cycles in fully aquatic groups involve vegetative (asexual) reproduction, as in naidid oligochaetes, which periodically bud off terminal body segments, or Lumbriculus variegatus (Lumbriculidae) which splits in half and regenerates each part (pers. comm., M.J. Winterbourn, Canterbury University). Like most annelids, naidid worms can also reproduce sexually, with the eggs deposited in cocoons. The most sophisticated form of reproduction in annelids occurs in glossiphoniid leeches (e.g., Glossiphonia heteroclita), which retain eggs in a brood case attached to the parent (Williams, 1980).

Particularly intriguing life cycle stages occur in 2 of the more common molluscs. The freshwater mussel, Hyridella menziesi, belongs to a family of bivalves which has a larval stage (glochidium) parasitic on fish (Williams, 1980), while one of the most common and widespread of all aquatic invertebrates, the snail, Potamopyrgus antipodarum, is ovoviviparous and facultatively parthenogenetic (Winterbourn, 1970). Various forms of ovoviviparity are found in several families of crustaceans, but a number of planktonic forms found in lakes have free-swimming larvae (nauplii) (Chapman & Lewis, 1976). The most bizarre life cycle is that of the freshwater shrimp, Paratya curvirostris, which has a planktonic larva that develops in estuaries, and is protandrous, changing sex from male to female as its size increases (Carpenter, 1978, 1983).

Insects do not show such a wide array of life cycles, but they do differ in their form of development and in the duration of the aquatic phase of various life stages. In the Hemiptera (bugs) and Plecoptera (stoneflies) development is gradual or incomplete, with newly hatched young resembling adults except for the absence of wings. Similar development occurs in Odonata and Ephemeroptera (mayflies), but the immatures do not resemble adults as clearly because of the presence of complex gills (mayflies) and mouthparts (odonates and mayflies). Immature stages in this group with gradual development frequently are termed nymphs. The remaining insects undergo complete metamorphosis, which consists of 3 distinct stages : an immature larva, a non-feeding pupa, and an adult (Barnes, 1968).

A generalised life cycle of an aquatic insect would involve a terrestrial adult laying eggs into water or onto a submerged substratum, and an aquatic larval and/or pupal stage present in the water for the major part of the insect's life. There are in fact numerous variations on this pattern, and several of them are found in Coleoptera. In some New Zealand beetles (e.g., Ptilodactylidae, Helodidae) only the larvae are aquatic. Others have aquatic larvae and adults, but terrestrial pupae (e.g., Elmidae, Dytiscidae), while some have terrestrial (or amphibious) larvae, but aquatic adults (e.g., Hydraenidae) (Williams, 1980). Because of this variation, a single benthic sample can produce an array of life stages (Fig. 1), some of which may have an aquatic phase of short duration, while others are present for 12 months or longer. For example, the elmid beetle, Hydora nitida, has a larval life of 2 years, but is present in streams as an adult for a maximum of 5 months (Towns, 1981b).

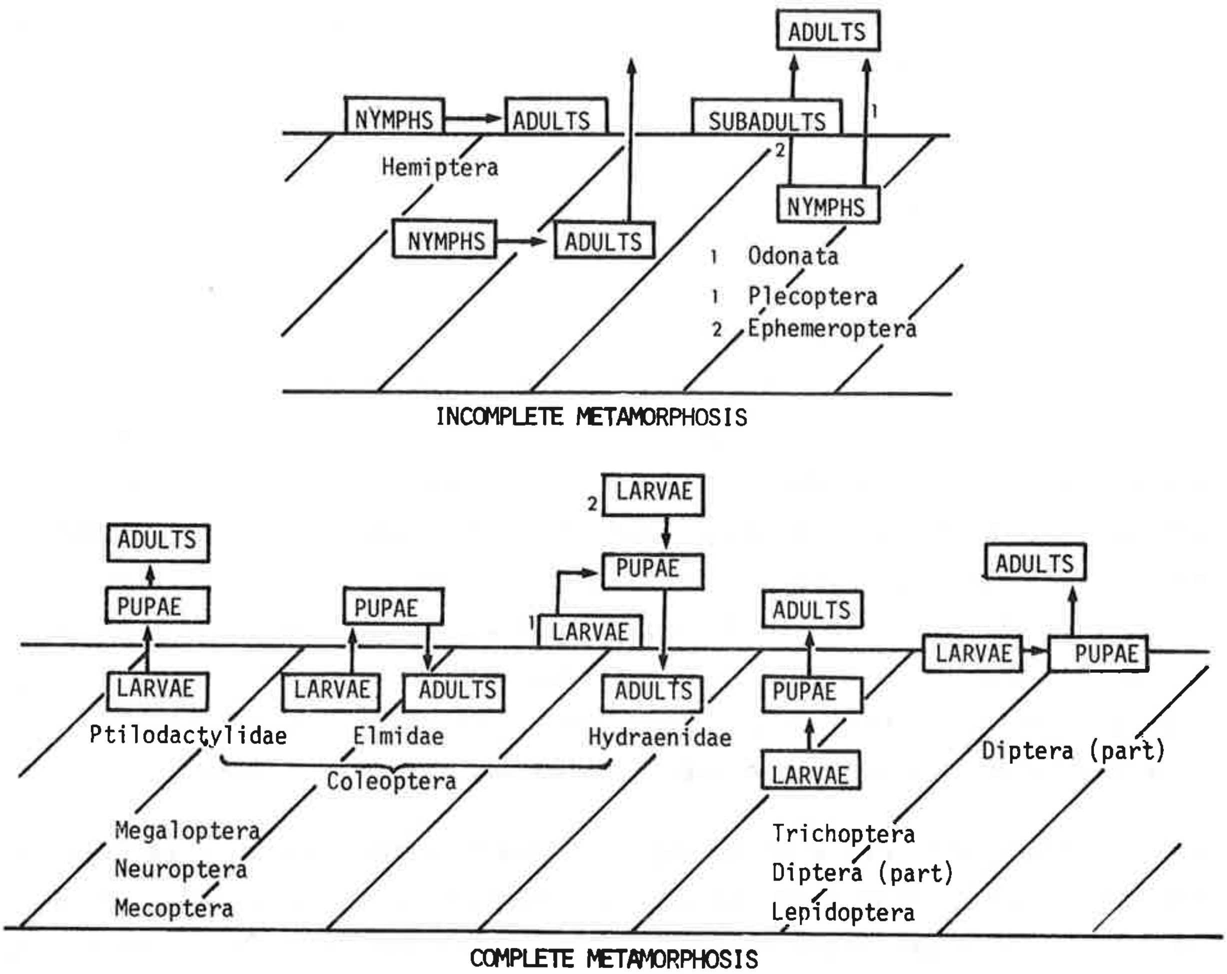

Fig: 1 **Life cycles commonly encountered in New Zealand aquatic insects, with the diagonally striped lower portion representing aquatic phases. Note : of the groups undergoing incomplete metamorphosis, only Ephemeroptera (2) have 2 terrestrial life stages; Hemiptera may be either surface dwellers or almost fully aquatic; some Diptera have aquatic larvae and pupae which breathe at the water surface; Hydraenidae larvae may be terrestrial (1) or amphibious (2); and Neuroptera have no fully aquatic members in New Zealand, but families with aquatic larvae occur elsewhere.**

SEASONALITY AND LIFE HISTORY PATTERNS

Because the life cycles of aquatic insects include terrestrial phases, some sections of the fauna may be absent from aquatic environments for quite long periods of time (cf. Hynes, 1970). In addition, closely related species may show sequential hatching, development and emergence periods, with little overlap in larval size distribution (cf. Cummins & Klug, 1979; Cowie, 1980; Towns, 1983). Although some seasonal variation in composition occurs in New Zealand aquatic invertebrate communities, the amount of variation is much lower than that commonly reported elsewhere. Suggested cues for the presence of high levels of seasonality, and the implications of seasonality for stream community dynamics have received considerable debate. At present, one school believes that abiotic factors and/or the effects of predation provide the primary determinants of life history patterns and habitat use, whereas the more conventional view espouses interspecific competition as a dominant and all-pervasive influence (e.g., Miller, 1984 vs Thorp, 1984). Actually, there is little empirical support for either position, although there is evidence that life history patterns may be fixed genetically and so possess little flexibility in some groups yet show considerable flexibility in response to environmental factors in others. The least flexible life cycles appear in species which undergo an obligatory diapause, which can occur in the egg, early instar nymph, or pupa (Hynes, 1970) (Table 2). Most species studied over a wide geographic range show some flexibility in their life history patterns. These patterns can be influenced by food quality, such as occurs in some caddisflies which produce larger numbers of generations of larvae when they have access to high quality food sources (Anderson & Sedell, 1979; Dall et al., 1984), or water temperature, which affects the time of emergence from quiescent stages, growth rates, and the number of generations produced (Table 2) (Hynes, 1970; Newell & Minshall, 1978; Humpesch, 1979; Mackay, 1984). A form of quiescence is one of the most important potential influences on the life history patterns of New Zealand aquatic invertebrates. This is delayed hatching of eggs, a process yet to be investigated in detail here, but apparently common in Australian stoneflies, some species of which are capable of producing hatchlings for all or much of the year from a single set of eggs (Hynes & Hynes, 1975).

Various models have been proposed to account for the "annual cycles" reported in northern hemisphere streams. The simplest of these was proposed by Hynes (1970), who divided life history patterns into 2 types : non-seasonal, in which individuals in a wide range of size classes are present throughout the year (e.g., species with long life cycles and non-insect groups); and seasonal, in which there is a distinct change in size distribution with time. The latter category can be subdivided into "slow" and "fast" seasonal patterns depending on the rate of development and the degree of quiescence or diapause in the early stages (Fig. 2).

TABLE 2 : Possible influences on life history patterns in insects.

INFLEXIBLE LIFE HISTORIES	RELEASER
Diapause - in the egg, early instar larva, or pupa	Temperature/daylength
FLEXIBLE LIFE HISTORIES	INFLUENCE
Quiescence - in the egg, early instar larva, or pupa	Temperature Genetic variability
Growth rate	Temperature Food quality

The life history patterns of a large number of New Zealand freshwater invertebrates are now known (e.g., Towns, 1976, 1981b, 1983; Cowie, 1980; Stark, 1981; Summerhays, 1983; and references cited in Winterbourn & Gregson, 1981), and seasonal cycles, such as those found in the northern hemisphere, are so rarely encountered that it has become more appropriate to refer our patterns to the categories of "well" versus "poorly" synchronised (Winterbourn, 1978; Winterbourn *et al*., 1981; Towns, 1981b).

This situation can be illustrated using mayflies as an example. Univoltine (single annual cycle) life history patterns and staggered emergence of congeneric species are reported frequently from temperate streams (references cited in Towns, 1983), but there are few examples known from New Zealand. Instead species assemblages here tend to show a mixture of well and poorly synchronised life history patterns, overlapping generations and size cohorts, and both long and short emergence periods (e.g., Towns,

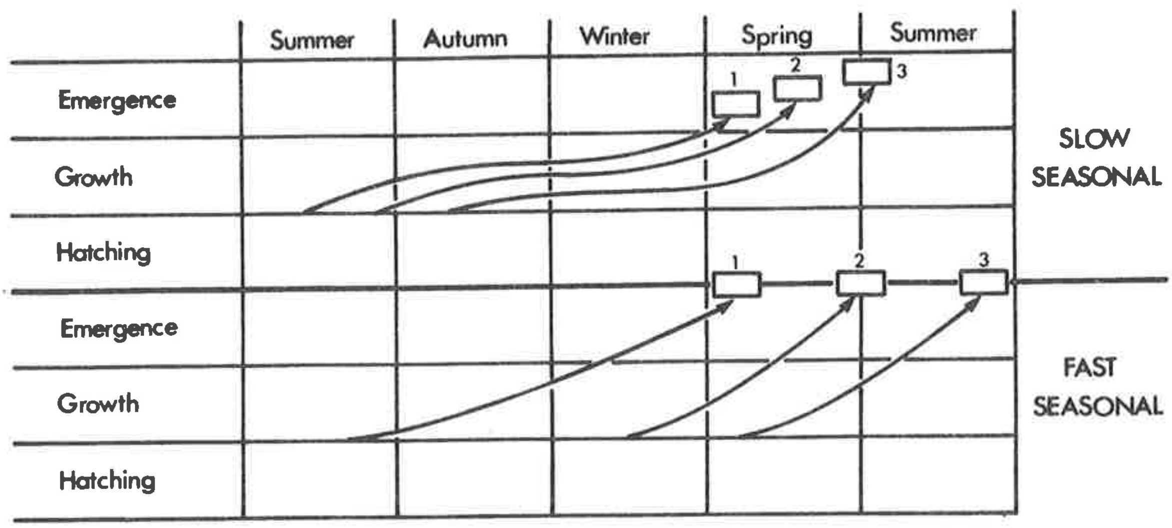

Fig. 2: The Hynes (1970) model of seasonal life history patterns (redrawn). Numbers refer to different cohorts.

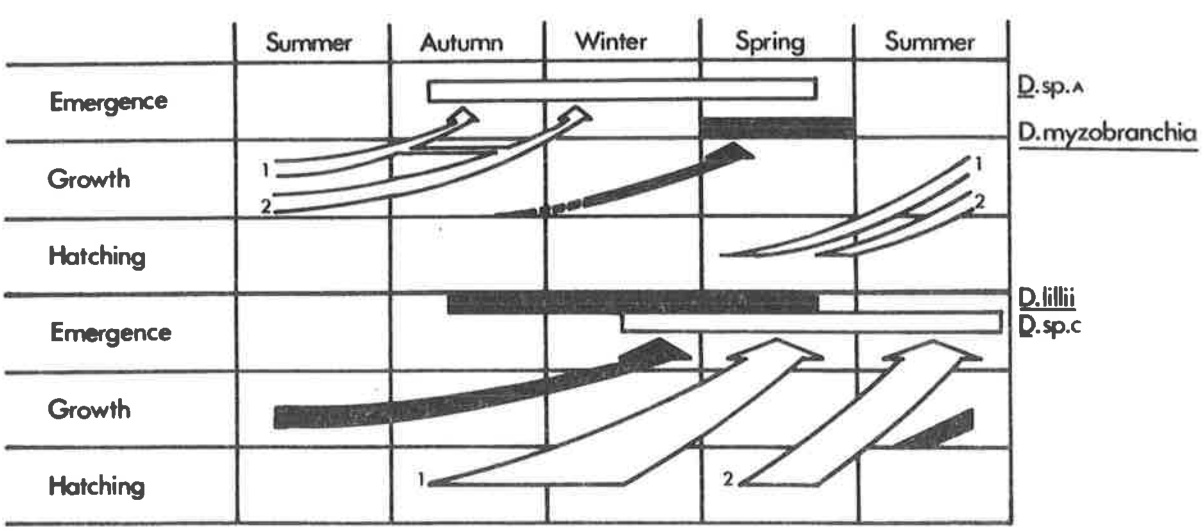

Fig. 3: Life history patterns of 4 sympatric New Zealand mayflies (Deleatidium), with species distinguished by presence/absence of shading. Arrows show hypothetical cohorts or generations. Deleatidium sp. C (shaded) and D. sp. A (unshaded) may have 2 generations (1 and 2) (after Towns, 1983).

1983; Summerhays, 1983) (Fig. 3). Even patterns shown by mayflies of cool southern streams appear to be no more clearly synchronised than those of warmer, northern streams (e.g., Winterbourn, 1974, 1978; Towns, 1983).

Possible reasons for the general lack of synchrony in the life history patterns of New Zealand aquatic invertebrates have been discussed by Winterbourn _et al_. (1981) who proposed that steep, unstable streams and unpredictability of the physical environment have favoured selection for opportunism. Curiously, these same life history patterns occur in the more stable northern streams (Towns, 1983; Summerhays, 1983), so additional influences may be involved. These could include the importance of fine particulate organic matter as a food resource which is utilised by a large proportion of the invertebrate community of New Zealand streams (Winterbourn _et al_., 1981) and wide climatic fluctuations over long geological time periods (Towns, 1976), both of which could favour selection for flexibility in an insular fauna.

THE STATUS OF STUDIES ON LIFE HISTORY PATTERNS

There is good reason to question the basis for much of the debate over the significance of life history patterns, because few studies have been conducted on the same species over a range of habitats, or on the same species for more than 12 months. A recent discovery by Resh (1982) indicates that long-term studies can be particularly instructive. Resh found that the caddisfly _Gumaga nigricula_ in a spring seepage in California had a poorly defined life history pattern in which early instar larvae and adults were found throughout the year - a pattern commonly encountered in New Zealand caddisflies (e.g., Towns, 1981b). A drought caused the stream to dry up and the caddisfly was eliminated. Two years later, once flow had recommenced, _G. nigricula_ reappeared, but this time as a single cohort population, with clear temporal succession from early to late instars and a narrow (two-month) adult emergence period. The species had shifted from a poorly synchronised, nonseasonal life history pattern to a well synchronised and highly seasonal one, apparently as a result of oviposition from a founder population present for a short time (Resh 1982). These findings have 2 important implications. First, they indicate that observed larval growth patterns may be related to constraints on the flight period of the adults, rather than to the widely assumed direct biotic or abiotic influences on the larval stage _per se_; and second, that life history patterns may be the result of some past random event.

This example illustrates well that we still lack a clear understanding of why certain life history patterns occur, to what extent they can fluctuate, and to what extent these fluctuations can be attributed to biotic or abiotic influences. There has even been some suggestion that clear temporal segregation of life history patterns in temperate northern hemisphere streams is less prevalent than would seem because only the well defined, clear, seasonal patterns are considered worthy of publication (pers. comm., B.L. Peckarsky, Cornell University).

IMPLICATIONS FOR STUDIES IN NEW ZEALAND

This comparison of studies on New Zealand invertebrates with the results of studies elsewhere indicates that 2 frequently encountered statements should be treated with suspicion :

1 "Many species in several orders of insects have a single generation each year and are to be found in the water only at a certain, often quite limited season" (Hynes, 1970, p.271).
2 "Temporal isolation of potential competitors .... seems to be a general rule among stream macroinvertebrates. When such species occupy the same general space in the stream their growth patterns are largely non-overlapping" (Cummins & Klug, 1979; p.162).

Faced by what appears to be the combined effects of confusion at a theoretical level over the significance of invertebrate life history patterns, plus the difficulties of interpreting the patterns found in New Zealand, resource managers would be justified in feeling that they are confronted by a particularly difficult problem. In fact, the poorly synchronised life history patterns prevalent in New Zealand aquatic ecosystems probably makes the use of invertebrate communities in pollution analysis less complicated here than elsewhere.

In those temperate northern hemisphere situations where there is an apparent predominance of invertebrates with slow or fast seasonal cycles, 1 or 2 sampling periods in a year could greatly underestimate total species richness. Indeed, without a thorough knowledge of the life histories of the species which might be present, erroneous assessment of the potential effects of intermittent pollution could be produced.

In New Zealand, hatching and emergence periods are often long and most species are present in streams for much of the year. Consequently, 2 samples, 1 in summer and 1 in winter in most situations should provide a close approximation of potential species richness (e.g., Towns, 1979). Because seasonal factors are minimised, the absence of key species or segments of the fauna should be related more closely to some environmental disturbance, rather than to peculiarities in life history patterns. In addition, because growth cohorts are often poorly defined, a wide range of size classes may be present at any one time. Consequently, the standard mesh sizes of 200-500 µm should capture some representatives of most species. Smaller mesh sizes are important where quantitative sampling is being attempted or where early instars or certain groups, such as chironomids and small oligochaetes, are required.

I will conclude this account with a note of optimism and one of caution. Complications in the life history patterns of aquatic invertebrates in New Zealand present analytical problems which need to be confronted by ecologists, but the same poorly synchronised patterns may be an advantage in pollution assessment. However, in situations where invertebrate diversity is low, either naturally, as in lakes (Timms, 1982), or due to subtle environmental effects, such as opening of the forest canopy (Towns, 1981a), responses to additional environmental change may not always be easily interpreted. The challenges posed by this situation are now being approached by Stark (1985), who is providing appropriate methods for the analyses of community structure.

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SAMPLING STREAM INVERTEBRATES

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ABSTRACT

Before embarking on a study of the benthos of a stream the biologist needs to ask 2 important questions: (1) What do I want to find out?, and (2) What procedures should I follow to accomplish my aim? This paper considers only the second of these questions. Two broad kinds of study can be defined. Faunal surveys aim to discover what species are present and perhaps their relative abundance. The objective of quantitative studies is to estimate numbers of organisms per unit area. Some form of the foot-kick method employing a hand net is suitable for faunal surveys. The main "quantitative" samplers used on hard substrata are the Surber and several kinds of box or cylinder samplers. Their attributes and disadvantages are discussed along with individual stone sampling, a method which shows some promise. Most samplers incorporate nets, and choice of mesh size is important. It can vary depending on the aims of the study and the conditions prevailing in the environment. Three other questions considered are: (1) Where should I sample?, (2) When should I sample?, and (3) How many samples should I take? Answers to these again depend on the study's objectives but inevitably will be a compromise between theoretical and practical considerations. Finally, methods of processing samples and reporting procedures are discussed.

INTRODUCTION

Changes in stream benthos populations are of special value in studies concerned with habitat changes because most benthic organisms have short life cycles and respond rapidly to habitat alteration (Raleigh & Short, 1981). Many methods have been proposed to sample benthic faunas and all have certain merits and detractions. These have been discussed at length by a number of authors, and the accounts given by Hynes (1970), Resh (1979), and Peckarsky (1984) are recommended.

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The fundamental question which must be asked by the practical biologist is "What do I want to find out?" Obviously, the objectives should guide the procedures selected - but is this often the case?

In his excellent handbook, Elliott (1977) distinguished 2 principal types of study, (1) faunal surveys, and (2) quantitative studies, the objectives of which demand different sampling procedures.

The aims of faunal surveys are to discover what species are present and often to estimate relative abundance. In contrast, the basic objective of a quantitative study is to estimate numbers of organisms per unit area.

Considerable pessimism has been expressed concerning the practicalities of quantitative sampling and is exemplified by the statement of Harper & Hynes (1972) that "The gathering of a representative sample from a population of nymphs is not presently feasible because of unsolved problems. Indeed, no net or apparatus has yet been devised to sample adequately the common gravel and cobble substratum." More constructively, Resh (1979) noted that errors in sampling derive from 4 main sources: (1) choice and operation of sampling devices, (2) physical features of the environment, (3) field and laboratory sorting procedures, and (4) biological features of the study organisms themselves. These must be kept in mind.

METHODS AND SAMPLERS

Samplers can be divided into 2 groups depending on whether they are designed to be used on hard or soft substrata. The latter include a number of coring devices, grabs and the Ekman dredge which is well known to lake biologists (see e.g., Biggs, 1983). In this paper I will not discuss their particular merits but it should be noted that many of the same problems are encountered in sampling soft substrate as in sampling hard substrata.

Samplers for use on hard substrata in flowing water fall into a number of categories. Most frequently used are handnets, Surber samplers, and enclosed box or cylinder samplers (Table 1). They will be considered below.

Airlift, drift and shovel samplers as well as electrofishing methods can also be employed but have rather fewer devotees. Individual stone sampling

Table 1 : Sampling methods used in 122 published studies of distributions and life histories of benthic stream invertebrates. The total is greater than 122 as more than 1 type of sampling was used in some studies.

Method	Number of Studies		Number of studies
Surber sampler	46	Corer	4
Hand net	34	Pump	2
Box or cylinder sampler	27	Airlift sampler	1
Grabs	8	Electrofishing	1
Individual stones	6	Open quadrats	1
Drift nets	5	Snag removal	1
Shovel sampler	4	Total	140

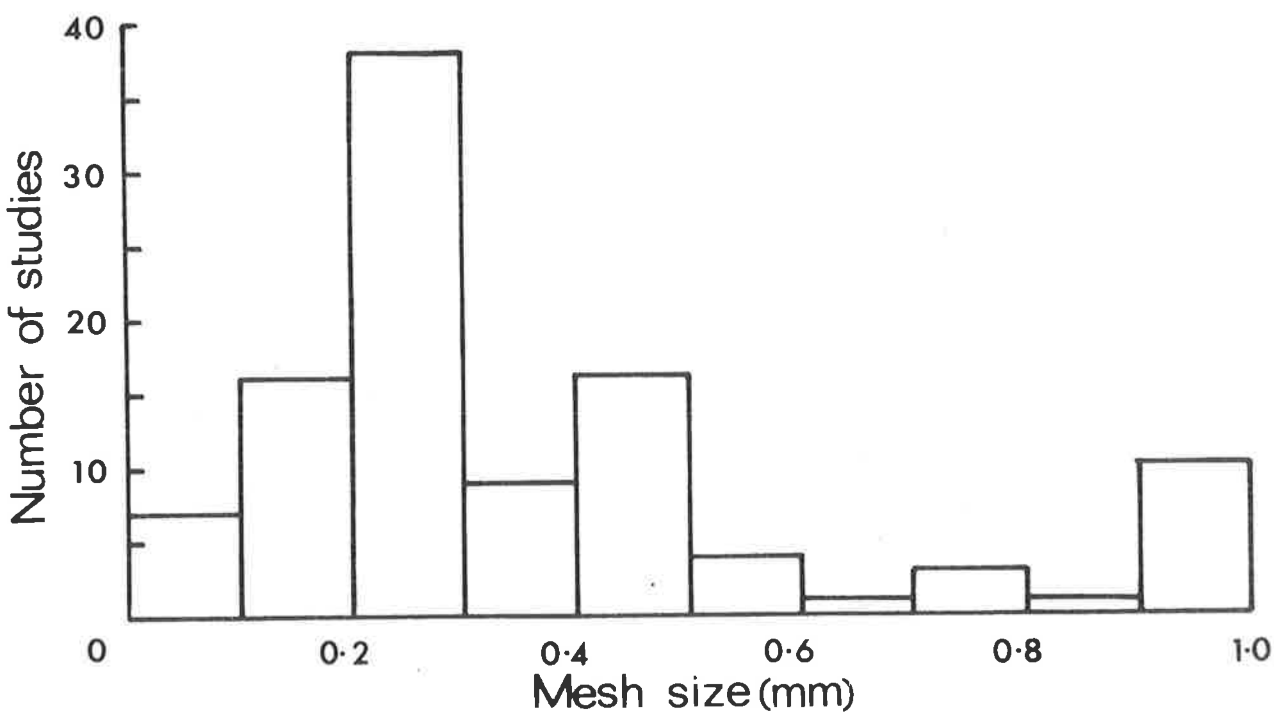

Fig. 1: Minimum mesh size of netting or sieves used in 105 published studies in invertebrate life histories and distributions.

has been advocated recently by Doeg & Lake (1981) and others and will be discussed.

Many sampling devices used in streams incorporate a catching net into which animals are swept by the current. Variations in catchability between samplers have been demonstrated by various workers and one reason for this is the mesh size used.

Mesh size

Meshes used on collecting nets vary widely in size (Fig. 1), and the selection of a suitable mesh should be guided by the requirements of the study. For accurate life history or productivity studies of many species, a very fine mesh (<200 μm) may be needed to retain all young stages. However, fine-mesh nets also gather and become clogged with fine sediments, algae, and detritus which make the subsequent processing of samples difficult. Also, fine-mesh nets tend to be inefficient in rapid and very slowly flowing waters. Williams & Hynes (1971) found that in very silty streams, a net with 20 meshes cm^{-1} choked so they resorted to a much coarser (8 meshes cm^{-1}) weave. In small slowly-flowing streams of the Maimai Experimental Area near Reefton, water would not flow satisfactorily through a 0.2 mm mesh net so a 0.8 mm mesh was used (Winterbourn & Rounick, in press).

Zelt & Clifford (1972) made a direct comparison of the capture efficiencies of 720 μm and 320 μm mesh nets in a Canadian stream and found that 50% of the aquatic insects captured by the latter passed through the coarser mesh. Nevertheless, they concluded that inferences concerning life history patterns of insects were not affected by the choice of mesh and that biomass estimates were reduced by only about 5% when calculated from coarse net samples. Biggs (1983) recommended that a mesh size of 200 to 210 μm be used routinely for survey work, but I consider that for most purposes a 500 μm mesh should suffice. This will capture specimens of most species including the later larval instars of small Chironomidae.

Non-quantitative sampling

A simple hand net is often used in survey work. The frame of the net can be circular (about 30 cm diameter), square or triangular (side about 30 cm), the latter being the most usual design. It is useful to attach the net to the frame by a sleeve of tough calico or plastic to reduce damage to the net. The net needs to be reasonably long (>50 cm is preferable) in order to minimize potential problems with clogging and backflow around the mouth. If soft, flexible material is used for net construction, emptying is made easy. Cowie (1985) found that using 2 triangular hand nets (1.0 mm and 0.2 mm mesh) "in tandem" enabled collections to be made effectively and subsequent sorting was simplified.

To operate a hand net, the substratum must be disturbed immediately upstream of the net. The exact distance depends on the flow regime and needs to be far enough away that sediment is not the dominant material caught! Stones are turned over and scrubbed by hand or with a brush, and/or the foot-kick method is used. The latter was formally described by Frost _et al_. (1971) and is convenient, effective, and an obvious choice for faunal surveys. Also, kick sampling is one of the few techniques available in rivers where the substratum is rough and unsorted (Cowie, 1985) or rocky, and where water levels vary considerably (Boon, 1978). Where such conditions prevail, it is likely that fixed-area samplers can be used only on atypical parts of the bed which limits their usefulness.

The effectiveness of kick-net sampling is affected by its duration, kicking intensity, behaviour of the fauna, mesh-size, and flow (Frost _et al_., 1971). In faunal surveys where as full a complement of species as possible is desired, a collection should cover a large area of the stream bottom (Elliott, 1977). Further, if an attempt is made to collect from all habitats roughly in proportion to their abundance, then useful comparisons can be made between stations and/or times, at least in terms of relative abundance of the more common species.

In an attempt to quantify the kick-net method, some workers have tried to standardize the time taken sampling or the number and intensity of kicks. This can provide only rough quantitative data and fixed-time sampling is not recommended by Elliott (1977).

Quantitative sampling

Quantitative samplers delimit a known area of streambed and animals within that area are collected. Several workers have demonstrated variations in catchability between collectors some of which can be attributed to the degree of enclosure of the device. The Surber is the most commonly used semi-enclosed sampler, although its suitability for community analysis has received considerable debate (see e.g., Bishop, 1973). Its reliability depends largely on the user (Pennak, 1977), and Hughes (1975) found that results obtained with it were comparable to those from an enclosed, cylinder sampler.

Surber sampling

The Surber sampler has a square, horizontal frame, usually 0.1 m^2 or thereabouts, and a collecting net slung from an identical sized vertical frame which forms the back of the sampler (see Biggs (1983) for further details). Triangular sides are usually present to prevent or at least reduce losses of fauna around the sides of the net. Sediments within the horizontal frame are disturbed, preferably to a predetermined depth. This is often 5 to 10 cm but depends on the nature of the bed. The disturbance procedure should be standardized and may involve digging into the bed with a trowel, stick or hands (look out for broken glass!) and/or brushing stones which are held in the mouth of the net. If stones are not scrubbed, some animals such as flatworms, caddisfly pupae and the larvae of _Helicopsyche_ and Hydropsychidae are likely to be missed.

Box samplers

Box or cylinder samplers are superior Surber samplers which enclose the entire area from which the collection is to be made. This prevents losses of animals or the entry of others from upstream. Well-known designs are the Neil, Hess, and Waters and Knapp samplers (see Welch, 1948; Schwoerbel, 1970; Waters & Knapp, 1961).

Box samplers sometimes have mesh-covered windows in their sides to enable at least a slow current to flow through the device. A net is attached to the downstream side and normally it leads to a removable collecting jar where the catch accumulates. If an "Agee" type ring is sewn into the apex of the net, collecting jars can be attached and removed readily. Alternatively, the collecting jar can be slid into a tight-fitting, elastic sleeve.

Some box samplers (and Surbers) have a toothed lower rim which helps screw the device into the stream bed. Operation of the sampler is as for the

Surber although a current may have to be produced by hand to ensure that animals float into the net.

Another kind of box sampler has no attached collecting net. Instead, the operator removes disturbed fauna from the water column inside the box with a hand net until no further animals are seen. This is likely to be a tedious procedure and some of the heavier organisms like snails and cased caddis probably will be missed.

Individual stone sampling

An alternative method of sampling which provides quantitative density data is to take individual stones as sampling units. This method could be useful in New Zealand streams especially where beds are irregular and rocky, and quantitative samplers cannot be used with ease.

Stones are sampled by placing a handnet of appropriate area and mesh-size immediately behind the stone which is lifted or rolled into it. In this way, organisms on and immediately below the stone are captured along with trapped detritus which floats free upon stone removal. It is possible that some active insect larvae (e.g., Nesameletus) may avoid capture by this technique (as when using a Surber sampler), and to prevent this a totally enclosed stone-lifting device was devised by Doeg & Lake (1981).

The size, number, and location of stones to be sampled will depend on the aims of the study and conditions in the particular stream. In a study of the population dynamics of a Danish stonefly, Madsen (1976) took at least 10 stones of greatest dimension 5 to 12 cm per station on each sampling day. Lake et al. (in press) collected 15 randomly selected rocks (size not given) per month in a benthic community study.

Results are expressed in terms of stone surface area which can be calculated in several ways. The most appropriate measure of surface area is unclear as in practice only a portion is likely to represent actual habitat available to and used by aquatic invertebrates. In any event, individual stone area cannot be equated with the area delimited by a Box or Surber sampler.

The simplest, crudest, yet most practical measure of "inhabitable surface" is obtained by multiplying stone length by width. Alternatively, an approximation of total surface area can be calculated as $S = \pi(LW+LH+WH)/3$ where L, W and H are stone length, width and height, respectively. Total area also can be determined by wrapping stones in "Gladwrap" or aluminium foil. The wrapping is cut to size, dried (if necessary) and weighed, and the weight converted to area using an empirically established relationship.

McElhone & Davies (1983) argued that for organisms attached to stones (like some larval Trichoptera) that are strongly influenced by surface area, quantification in terms of numbers per unit area of stone surface is potentially a more meaningful measure than abundance (i.e., numbers per unit area of stream bed). However, for species not strongly influenced by rock surface area, conventional quantification in terms of abundance is appropriate.

Whether the New Zealand aquatic fauna includes species whose distributions are correlated with stone size is not known, and the representativeness of stone samples for community analysis has yet to be demonstrated. Nevertheless, for survey work and life history studies the individual stone method is worth considering as it is convenient and provides clean, rapidly processed collections. It is interesting to note that some recent studies of net-spinning Trichoptera have used snags and submerged branches as sampling units and density, biomass, and production have been expressed per unit area of branch surface (Cudney & Wallace, 1980; Benke et al., 1984). Results expressed in such units are difficult to equate with those from conventional streambed samples unless the bed area covered by snags is known.

WHERE SHOULD I SAMPLE?

The distributions of benthic invertebrates are determined by a number of interacting factors of which substratum, current, and food have been emphasized by Cummins & Lauff (1969).

Where biological data are required for water quality assessment purposes, comparable sampling sites are needed at all stations being compared. This reduces the "competing" influences of factors other than water quality. As a general rule, riffles are favoured sites, and if flow, depth, and substratum particle size are similar so much the better. Communities in riffles are often species rich and dominated by insect larvae, whereas fewer species typically occur in pools. Slack waters are also depositional areas where fine sediments and organic debris accumulate, and as a result burrowers and deposit feeders can be expected. These include various "pollution tolerant" taxa (e.g., some Chironomidae and Oligochaeta - see Winterbourn, 1981) which are useful as organic pollution indicators only when associated with hard substrata.

Where life history information is required for particular invertebrate species, sampling of a single habitat is unlikely to be sufficient. This is because the larvae of many species change their habitat requirements as they grow older. For example, young larvae of the New Zealand mayfly *Oniscigaster wakefieldi* commonly inhabit sand and fine gravel bars in shallow water, whereas older larvae occur in deep pools, and emergence is on stones at the heads of the pools (McLean, 1970). Larvae of the stonefly *Acroperla trivacuata* migrate towards the sides of streams as they develop and late larvae are semi-terrestrial (Winterbourn, 1966). Larvae of other species may change their vertical distributions within the streambed at different times of year. *Spaniocercoides cowleyi* is likely to do this since late-instar larvae and adults are commonly seen in Westland streams during spring and early summer but smaller larvae are rarely found (Cowie, 1980).

Where non-quantitative data are sufficient for a life history study, hand net sampling should attempt to encompass the whole range of habitats present at a station (see above). Where quantitative samples are wanted their placement needs to be decided with care. Stratified-random programmes (i.e., random placement of samples within the various habitat types) are recommended. Cross-stream transects do this with samples placed at equal intervals so the habitat types are sampled roughly in proportion to their occurrence. The number of sampling points will depend on the width of the river, the heterogeneity of the environment and the commitment of the biologist. Dudgeon (1982) recommended 5 or multiples of 5; Resh (1977) managed 70! A sensible practice is to lay a diagonal transect and to work across the stream starting at the downstream end. This will ensure that the operator does not accidentally disturb sites that have yet to be sampled.

HOW MANY SAMPLES SHOULD I COLLECT?

This simple question has no simple answer. It should depend on the object of the study and according to Resh (1979) "it is generally determined by experience or intuition and then modified by cost considerations". In practice, this means as few as possible which is not unreasonable considering that the collection and processing of samples is a lengthy and tedious business. Raleigh & Short (1981) give a figure of 130 hours to collect and process 47 Surber samples.

More formally, where data on the numerical density of invertebrates are required, the number of sampling units needed is a function of the size of the mean, the degree of aggregation of the population, and the desired precision of the mean estimates. Elliott (1977) considered that a standard error of ± 20% (95% confidence limits of about ± 40%) was a realistic degree of precision to achieve in a stream sampling programme given the heterogeneity of the environment and the known tendency for clumping by the benthos.

To determine the minimum number of sampling units ($\underline{n}$) required to achieve a specified degree of precision, a preliminary sampling programme must be undertaken from which $\underline{n}$ can be calculated as follows :

$$\underline{n} = \frac{s^2}{D^2\bar{x}^2} \qquad \text{(Elliott, 1977)}$$

where s^2 = the variance

D = the desired standard error as a proportion of the mean

$\bar{x}$ = the mean

Inevitably, this provides only a general guide since the number of samples required to achieve a given degree of precision for one species, or one point in time is unlikely to apply to another. In practice, such an exercise is rarely carried out and the number of samples to be taken is decided subjectively. A perusal of papers in my reprint collection showed that in 85 "quantitative" studies of stream benthos the number of samples taken per station ranged from 2 to 70, with 5 or less in over half of them (Fig. 2).

It is worth noting here that most relatively undisturbed stream faunas include a few abundant species and a large number of less common ones

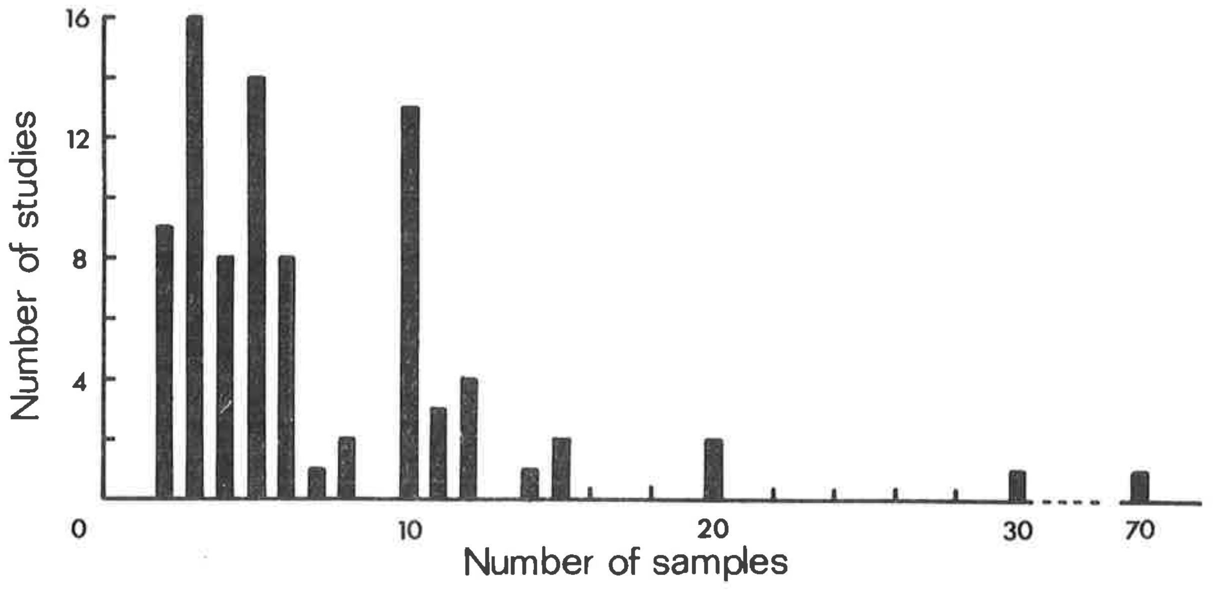

Fig. 2: **The number of samples (sampling units) taken per station in 85 published studies based on quantitative sampling.**

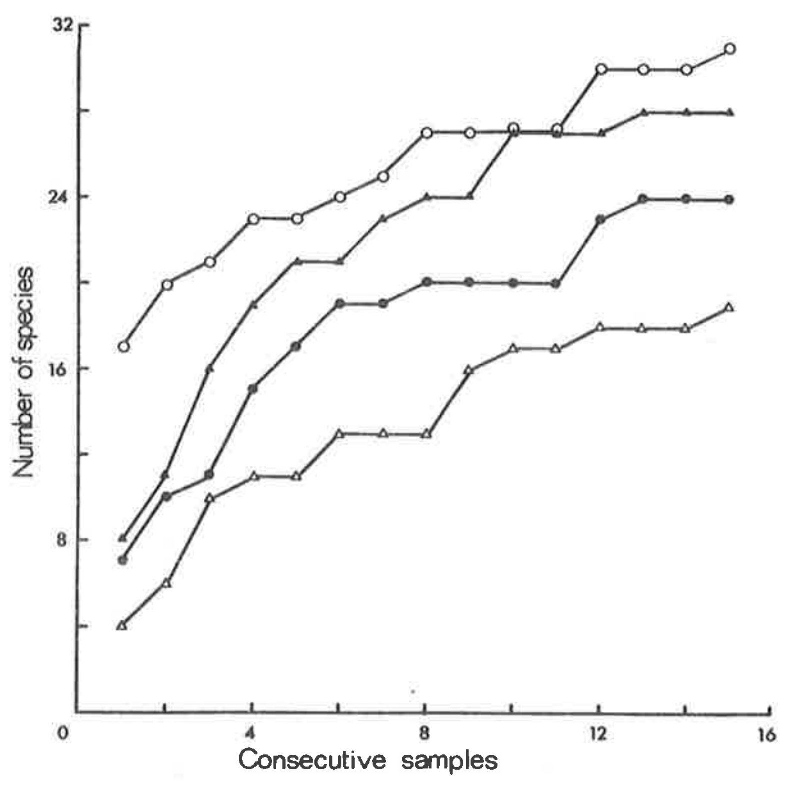

Fig. 3: **Cumulative numbers of species present in consecutively collected Surber samples (0.1 m^2) from Middle Bush stream (•, o) and Craigieburn Cutting stream (▲, Δ) on 2 occasions.**

(Table 2). This means that as sampling effort is increased the species list should increase as rare species are encountered (Fig. 3). This must be borne in mind when comparative studies using species presence/absence data are contemplated.

In order to make valid comparisons of species richness and faunal similarity with indices such as those of Margalef, Sorensen and Jaccard (see Winterbourn, 1981; Cowie, 1985), sampling effort needs to be as uniform as possible.

TABLE 2 : Numbers of invertebrate species in 4 relative abundance classes at 4 sites on Devils Creek, Reefton (from Cowie, 1980). Note: Over 50% of individuals were contributed by 1-3 species whereas 72 species never made up more than 0.1% of total numbers at any site.

Percentage of total numbers	Numbers of species Site 1	Site 2	Site 3	Site 4
Over 10	3	3	2	1
1-10	10	9	8	12
0.1-1	34	17	18	23
Under 0.1	35	47	68	52
Total species	82	76	96	88

WHEN TO SAMPLE

Once again, the aims of the study should dictate the time and frequency of sampling. In New Zealand, there does not seem to be any one time of year which is demonstrably "better" than any other for making "one-off" surveys (see e.g., Stark, in press). The poorly synchronized life histories of many invertebrates in this country mean that their larvae are present most of the time, although there may be changes in their relative abundance within a community. These may be consequences of life history phenomena or other events, for example, responses to environmental factors like flooding or drought which may or may not be predictable. Some stoneflies, including species of Zelandoperla and Acroperla, are exceptions to the "rule" that most species occur in streams year round. Their main period of larval growth is winter and spring and few larvae are seen in summer. If any one time is better than any other to carry out a survey then it is probably autumn. This follows the major emergence periods of insects and a new-generation of larvae can be expected to be present.

In contrast to surveys, life history and productivity studies require sequential sampling over a long period - perhaps a year or more. Traditionally, fixed-interval sampling (often monthly) has been employed, but the appropriateness of the monthly sampling mentality has been questioned by Waters (1979). Instead, he and Resh (1979) both advocated that the intensity and frequency of sampling should be greatest at times when the population of interest is changing in size and structure fastest. This requires prior knowledge of the general form of the life cycle which may not be available, however.

PRESERVING AND PROCESSING SAMPLES

Upon collection, samples are best transferred directly from nets (or whatever) to containers for transport to the laboratory. Initial transfer to a tray is a waste of time (and may result in losses), unless the collection needs to be perused for a particular reason.

To transfer net contents efficiently, concentrate them into the apex of the net by sweeping the net briskly through the open water, against the current. Then, invert the net into the container. If the latter contains some fluid, most adhering materials will float off the net. Repeat the procedure if necessary and check for "intransigent animals" clinging to the mesh. Larvae of the stonefly Zelandoperla, hydropsychid caddis larvae and oligochaete worms are notably hard to dislodge and may have to be picked off by hand.

Also add preservative to the sample in the field. If not done then, all samples will have to be reopened in the laboratory which adds an unnecessary step. Recommended preservatives are 70% methanol (cheaper than ethanol), and 4% formalin. The latter is rather unpleasant to use but preferable if animals are wanted for gut analyses or are to be dry weighed. This is because alcohol extracts plant pigments (e.g., chlorophyll from algal cells in guts) and lipids from animal tissues.

Following preservation in 70% alcohol for a month, the dry body weight of Deleatidium and Stenoperla larvae fell by about 27% and that of Zelandopsyche by 40% (Winterbourn, 1974; Winterbourn & Davis, 1976).

Plastic bags make convenient sample containers for field use. They are light, easily closed by tying a knot in the bag, and can be packed together in a small space as long as most air is squeezed out. Make sure they are not leaking though - formalin fumes make for an unpleasant drive home! As an alternative to bags, various kinds of glass or plastic jars with tight-fitting lids make convenient collection holders. A label with collection details - at least the date and the location should be placed inside each container. Use a pencil or non-soluble felt pen to write labels; ink runs. Labelling with felt pen on the outside of bags is also a good idea and enables particular samples to be located quickly.

Sorting is carried out conveniently in shallow, well lit white trays. Remember though that some animals, e.g., some leeches, amphipods and ostracods show up best against a dark background. If they are likely to be present transference of the sample to a black tray at some stage can be useful. If a sample contains a lot of sediment, wash it through a sieve or sieves to remove fine material. This procedure also removes preservatives whose presence can make sorting unpleasant. Select a sieve which will retain all animals needed in the study and remember that if the finest sieve used in the laboratory has a 1 mm mesh, then there is no point in using a 0.5 mm mesh net in the field. Work through tray samples methodically. Sort through small quantities of debris at a time and swirl the contents of the tray about at intervals. Remove animals with fine forceps (watchmakers forceps are best) or perhaps a fine paint brush or eyedropper, and place them into vials of clean perservative. Often this will be 70% methanol, but remember the choice of preservative again will depend on the ultimate use of specimens concerned.

A number of aids to sorting are available and may be worth considering. If very small insect larvae need to be recovered, a dissecting or travelling microscope is useful to provide greater magnification. Dyes such as phloxine B and Rose Bengal (Mason & Yevich, 1967) can be used to stain animals and so increase sorting efficiency. Blyth et al. (1983) used a combination of Rose Bengal and Lugols iodine to stain animals and Chlorozol

Black E to darken silt and algae. Flotation methods are advocated by some authors and if used properly will separate many animals (not snails with shells or cased caddis) from sediments and plant debris which remain on the bottom. Sugar and salt solutions are often recommended although the former are incredibly messy. I have had success with saturated solutions of $CaCl_2$ (see Anderson, 1959).

REPORTING SAMPLING PROCEDURES

Reporting of all studies involving benthic sampling should include complete information on methodology as a matter of course. A perusal of the scientific literature shows that this is not always the case and consequently the results of some studies cannot be evaluated with ease or confidence.

Descriptions of methodology should always include the type and dimensions of the sampler, net mesh size, number of samples taken and their arrangement (e.g., random, stratified-random, on a transect, in midstream), and depth of substratum disturbance. Further, accounts of quantitative studies should provide a statement as to the precision of means presented (standard errors or 95% confidence limits), at least in the text if not in relevant tables and figures.

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ARTIFICIAL SUBSTRATA : THEIR USE, ADVANTAGES AND DISADVANTAGES IN AQUATIC INVERTEBRATE MONITORING

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ABSTRACT

The use of artificial colonising substrata for invertebrate sampling and monitoring is discussed with reference to their use in New Zealand. Artificial substrata can provide favourable habitats of uniform and reproducible substratum composition and area for invertebrate colonisation. They permit the collection of a diverse fauna at stations of choice, and facilitate comparison of populations collected at different stations regardless of natural bottom conditions. Disadvantages of use include loss or damage by floods, debris or interference, and the collection of fauna that may not be accurately representative of that occurring naturally. Useful applications of artificial substrata in biological monitoring of New Zealand freshwaters are considered to be in short-term intensive surveys of large or difficult to sample rivers, in monitoring irregular point discharges in small streams, and in augmenting information collected by other sampling techniques.

INTRODUCTION

The macroinvertebrate community existing in any given situation is a product of its environment, and the principal community determinants within the environment are substratum type and water quality. Field studies of benthic invertebrates as indicators of water quality are often handicapped by the effects of natural substratum variability on community composition. In addition, and particularly in deep or swift rivers, the major invertebrate community component may be located in situations inaccessible to many common sampling techniques, such as Surber, kick-net or grab sampling.

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Substratum type and water quality obviously are not the only variables influencing invertebrate community composition (and hence influencing any index or diversity values). There are a number of other biotic (e.g., life cycle stage) and abiotic (e.g., geographical location, current velocity) factors which may have to be taken into account when interpreting a series of comparative invertebrate samples.

In developing a biological monitoring programme which incorporates sampling of benthic invertebrates, a question the investigator should ask is: "How can a series of samples be collected easily in a way that effects of natural substratum variability are reduced and effects of water quality highlighted?" And, in addition: "If the sampling locality is not amenable to application of the aforementioned sampling techniques, are there any other suitable techniques?"

Artificial substrata appear to provide some of the answers, particularly if a distinction is made between their use to measure detailed biological parameters and their use to assess the effects of water quality. In the former instance it is often essential that natural bottom conditions be duplicated as closely as possible. To assess water pollution, exact duplication of natural conditions is of less importance than ability to obtain replicate samples at different times and in different locations.

A review of the literature has shown that the development and use of various types of artificial substrata has facilitated the collection of comparative and quantitative samples and tackled the problem of sample collection and monitoring in situations where other techniques are not appropriate, or are used only with difficulty. Because of the variability inherent in invertebrate sampling it is important that those variables affected by sampling difficulty and substratum differences be reduced as much as possible.

The objectives of this paper are to briefly describe some of the types of artificial substrata that have been developed, and where appropriate to make reference to their use in New Zealand. In addition, the advantages and disadvantages of artificial substrata compared to more conventional invertebrate sampling techniques are discussed, and some conclusions are made regarding their use in biological monitoring of freshwaters.

Throughout the text the terms biological monitoring or invertebrate monitoring refer to the collection and examination of invertebrate samples to assess the broad based effects of water quality.

TYPES OF ARTIFICIAL SUBSTRATA

Two types of artificial substrata, basket samplers and multiplate samplers, are used commonly in the United States, and routinely by such organisations as the Federal Water Quality Administration and the Pollution Survey Division of the Environmental Protection Agency.

Basket samplers

Basket samplers were developed for use in large rivers in the United States (Mason et al., 1967). They consist of a cylindrical or rectangular wire mesh basket which is filled with limestone chips, rocks, plastic, debris or brush (e.g., Scott, 1958); the contents of the sampler can be varied to include elements of the natural substratum and/or material that permits some quantification of artificial substratum area. One side of the basket is hinged to permit replacement of substratum material and examination of the invertebrate sample. The assembled samplers can be suspended at the desired depth from a suitable platform or buoy arrangement (e.g., Mason et al., 1967) or may be placed on the natural substratum where access and river conditions permit.

Basket samplers were recently used during an intensive survey of the Tarawera River (pers. comm., J.D. Stark, Taranaki Catchment Commission). In this deep, fast moving river the natural substratum is comprised of mobile coarse and fine pumice material. Basket samplers containing similar numbers of similar sized rocks were deployed at 7 sites (with 5 replicates) over a single immersion period of 5 to 6 weeks. They were suspended from buoys anchored to the river bed. Some 76 taxa and very large numbers of individuals were collected. The samplers proved successful in distinguishing differences in invertebrate community composition that may be attributed to the effects of different discharges.

Multiplate samplers

The multiplate sampler was developed in the United States by Hester & Dendy (1962) and consists of a number of 7.5 cm^2 hardboard or ceramic plates held together on a central bolt and separated by uniform or various sized spacers. The assembled sampler can be weighted and suspended in the water at any depth (e.g., Fullner, 1971; Mason et al., 1973), or a number can be suspended below a buoy (e.g., Wefring & Teed, 1980). Alternatively they may be attached to a standard or metal bar driven into or lying on the river bed. Multiplate samplers have been used with some success in sampling in the Waikato River (Davenport, 1981), where the river bed is mainly mobile, duning pumice and sand, supporting few organisms. The invertebrate fauna is concentrated in the littoral fringes, and temporary and shifting weed beds. These areas are subject to large fluctuations in water level. Using multiplate samplers, it was found that the majority of organisms inhabiting the weed beds were sampled in reasonable numbers, and some additional taxa such as Ephemeroptera and Trichoptera which were not often collected in net sweeps of the weed beds were regularly obtained.

Multiplate samplers attached to metal stakes driven in to the river bed were used successfully in sections of the Patea River where the natural substratum is primarily mud and mudstone (pers. comm., B. Zuur, Waikato Valley Authority). Here the major component of the invertebrate fauna was restricted to submerged logs and other debris; these substrata were not amenable to rigorous sampling by other techniques.

The performances of the basket and multiplate artificial substrata described above in collecting invertebrates are quite similar. When constructed with equal substratum surface area and placed in similar environments, comparable samples, in terms of both numbers of individuals and numbers of genera, can be obtained (Fullner, 1971; Mason et al., 1973). Species preferences for one or other sampler type may become apparent, a factor due mainly to differences in substratum geometry and the feeding, respiratory and light requirements of particular species. In assessing the comparative performance of basket and multiplate samplers Fullner (1971) concluded that both collected a sufficient diversity of benthic fauna to be useful in assessing water quality. It is suggested, however, that prior to initiating a surveillance programme using artificial substrata some brief comparison be made between the fauna collected by the chosen substratum type and that present on the natural substratum.

In addition to the basket and multiplate samplers there are a number of other types of artificial substrata, each of which may be appropriate to certain river situations and sampling strategies.

Web or fibre samplers

These consist of lengths of plastic or other synthetic web, rope or fibre attached to a suitable float, or buoy and anchor arrangement (e.g., Cairns & Dickson, 1971). They are suitable for faunal sampling where filamentous algae and other plant material are permanently or temporarily a predominant component of the natural substratum. In common with other artificial substrata they may be deployed at any depth and immersed for various periods.

Wooden blocks

The use of these has most recently been reported by Winterbourn (1982). In a forest stream in the South Island, grooved wooden blocks placed on the stream bed were used to sample and study the invertebrate fauna colonising artificial substrata on which fine particulate material was deposited. The samplers were lifted and examined at monthly intervals, and at least 31 invertebrate species were sampled during the study.

Sewage filter media

Cylindrical, plastic sewage filter media, arranged as a series of cylinders and termed the Standard Aufwuchs Unit, have been evaluated for routine use in different rivers in the United Kingdom (James & Evison, 1979). Groups of these cylinders may be weighted and suspended at the desired depth, or attached directly to the substrata. The principle merit of this type of sampler is that it affords a large surface area relative to total size; however, certain organisms may be excluded due to the smooth plastic surface of the media. They have been used in a variety of small stream situations in New Zealand. Challis (1983) used plastic sewage media with some success in a study of the effects of low flows on invertebrates in the Ngakoroa Stream, near Auckland. In this situation up to thirteen species were recorded, and seasonal changes in numbers of individual organisms and species diversity were obvious. Sewage media samplers were also used to good advantage in a study of small streams of the upper Waitemata catchment (pers. comm., D. Challis, Auckland Regional Water Board). Here the invertebrate fauna was diverse and numerous, but restricted to sticks and debris in extensive river sections where the natural substratum comprised mud and silt. It was found that the sampling units were colonised very heavily and sub-sampling became necessary to reduce processing time.

The above list of the types of artificial substrata is not definitive, and other samplers have been used in particular circumstances, e.g., litter trays, concrete blocks, pipes filled with rocks to study depth colonisation, and so on. However, the types described constitute those that have some direct invertebrate monitoring application. The reader is referred to the additional list of references for further details of use and comparative performance of individual artificial substratum types. There are, however, a number of factors relevant to their use in a biological monitoring role that will be highlighted here.

THE USE OF ARTIFICIAL SUBSTRATA IN MONITORING PROGRAMMES

Artificial substrata can reduce or eliminate the problems associated with trying to sample highly variable substrata or sampling in difficult conditions. However, consistent practices in sampler installation are necessary in any monitoring programme in order that similar habitat types are made available for colonisation by the organisms of each station.

Samplers should be placed in similar environmental conditions, especially with respect to water velocity, light, depth and proximity to the natural substratum.

In common with other biological sampling techniques an appropriate number of replicates should be considered. In evaluating the use of basket samplers for sampling invertebrates, Dickson et al. (1971) found that, in clean water, 4 samplers were required to estimate the true mean number of taxa and individuals with an error of 25%; a greater number of replicates was required for statistical precision in clean water situations than in polluted conditions. Mason et al. (1973) found that 3 to 4 basket sampler replicates were adequate in providing estimates of total numbers with commonly accepted levels of statistical precision.

In an invertebrate monitoring programme adjacent to the Huntly Thermal Power Station on the Waikato River it was found that 3 multiplate replicates were generally sufficient for statistical precision (this author, unpublished material).

Placement of artificial substrata on the river bottom results in loss of control of substratum composition. Such control is necessary in comparing communities collected by samplers at stations subject to differing water quality. Mason et al. (1973) found that basket samplers placed on the bottom accumulated more sediment and showed greater variability between samples than those sited near the surface.

Immersion period, or colonisation time, is also important in an artificial substratum sampling programme. Mason et al. (1973) noted that although depth could affect invertebrate sample compostion, sampling duration had more effect on the numbers of organisms collected.

Winterbourn (1982), in using wooden block substrata to study invertebrate fauna associated with fine particulate matter in the Middle Bush Stream, considered that monthly intervals between sampling times should have enabled 'equilibrium level' invertebrate populations to establish. In examining the factors affecting the performance of artificial substrata, Mason et al. (1973) found that in the Little Maimi River (Cincinnatti) total numbers of organisms collected with different exposure periods became more pronounced with seasonal decline in water temperature. Significantly greater numbers of individuals occurred in samplers immersed for 8 weeks than those immersed for 4 or 6 weeks in winter.

In the Waikato River it was found that in winter multiplate samplers suspended mid-water collected more taxa and more individuals when immersed for 8 weeks than when immersed for 4 or 6 weeks (this author's unpublished observations).

A minimum colonisation time in New Zealand waters would appear to be 4 weeks. This permits development of a periphyton community on the artificial substratum (a minimum of 2 weeks) and subsequent colonisation by macroinvertebrates.

ADVANTAGES OF ARTIFICIAL SUBSTRATA

The main advantages of application of artificial substratum monitoring techniques have been alluded to. Essentially they provide a standard reproducible colonising substratum for the collection of invertebrate samples in comparative surveys. They are appropriate for use in rivers and at sampling stations that cannot be sampled effectively by other techniques such as kick nets, Surber samplers or grabs.

There are a number of other advantages in the use of artificial substratum monitoring methods.

In some situations artificial substrata can generate higher invertebrate community base diversities. This is the case in rivers with little or no stable substratum, e.g., in the Patea River and upper Waitemata tributary stream examples cited earlier. In the lower Waikato River at Huntly the invertebrate community generally is confined to littoral submerged plant beds, or to areas protected from strong currents where there is a stable soft mud and sand bottom. Some organisms, however, are restricted to hard substrata (logs and debris) in flowing water. In an invertebrate monitoring programme designed to examine the effects of thermal effluent from the Huntly Power Station on Waikato River invertebrate fauna, 3 sampling methods are employed (Waikato Valley Authority, 1982). Net sweep samples are taken to examine the weed-bed fauna, and grab samples taken to sample the sediment infauna. Multiplate samplers positioned adjacent to weed beds and above the mobile substratum have been used to sample the invertebrate fauna of flowing water. This fauna includes some taxa (e.g., Ephemeroptera, Trichoptera) which may be more sensitive to thermal pollution than, for example, the predominantly oligochaete fauna collected in grab samples. The role of artificial substrata here, in generating higher base diversities, is one of augmenting the information obtained with other sampling methods.

In some situations, and particularly in small streams, artificial substrata may be used on a more or less continuous basis in monitoring irregular point or diffuse discharges. In many cases some discharges cannot be identified or characterised by spot physico-chemical or biological sampling. In these instances artificial substrata can facilitate quantification of the effects of water quality on benthic community structure and abundance. In such long-term monitoring programmes more than one investigator may be required for the collection and analysis of samples. The use of artificial substrata can reduce help to reduce variation produced by different sampling techniques or reduce effort on the part of different investigators.

Some artificial substrata, in particular the multiplate type, can reduce sample processing time. They are simply disassembled, and animals are easily dislodged with a minimum of damage. This type of substratum is lightweight, quick and inexpensive to manufacture, and in an extensive monitoring programme provide a means whereby a large number of samples can be collected in a short time. The only field work required is replacement of the artificial substratum.

A characteristic of artificial substrata is that generally they do not provide a substratum similar in all respects to natural bottom conditions, and hence they do not collect an invertebrate community fully representative of that occurring naturally. This factor was highlighted by James & Evison (1979) as being the principle drawback of the Standard Aufwuchs Unit. In many instances this problem may be overcome to some degree by using artificial substrata that incorporate at least some of those materials forming the natural substratum of the river or site to be sampled. The wooden settlement blocks used by Winterbourn (1982) were constructed of mountain beech, the predominant timber surrounding the study site. Basket samplers can be assembled using rocks, gravel, or debris from the sampling locality, and multiplate samplers can be constructed of naturally occurring timber, or pretreated by immersing in water prior to use. In any event it is inevitable that some of the drift debris and sediment in any given situation will become lodged on or within the samplers.

The use of artificial substratum samplers for collection of comparative samples does not require that all species that may be present be sampled, or that the collected community is representative of natural communities nearby. Provided that a broad spectrum of species with a good population size is collected, comparison between sites can be made on the basis of community diversity.

DISADVANTAGES OF ARTIFICIAL SUBSTRATA

A very real disadvantage stems from the fact that artificial substrata must be anchored or secured in position for considerable periods. They are, therefore, prone to loss or damage caused by drifting debris, flood events, or gross movement of mobile river bed substrata. In addition to being securely anchored they must be marked in such a fashion that they can easily be relocated by the investigator. This normally requires the use of marker or floatation buoys, guide ropes or some other device. Such devices not only increase likelihood of loss due to drifting debris, but also act as very strong attractants to those in society with either inquisitive or vandalistic tendencies.

The above problems proved to be quite major ones in the previously mentioned invertebrate monitoring programme at the Huntly Power Station. Positioning and servicing samplers by diving went some way towards alleviating vandalism and drift damage, but created additional problems of relocating sample sites in turbid water or after gross movement of the river itself. As a result the sampling became labour intensive and time consuming. In addition the loss of all replicate samplers at any one sampling station meant the loss of data from a sampling period covering 8 weeks. The artificial substratum component of the monitoring programme has now been reduced to an annual survey in which the number of replicates has been increased to cater for losses.

The problems encountered in using plastic sewage filter media substrata in sampling the tributary streams of the upper Waitemata catchment included the retention of large quantities of silt, making sample sorting difficult. In addition, very large numbers of individuals were collected, necessitating some subsampling and requiring long periods of sorting, identification and counting (pers. comm., D. Challis, Auckland Regional Water Board). It must be noted that the collection of large numbers of animals is not necessarily a disadvantage.

Finally, a minor disadvantage of artificial substratum sampling is the dislodgement and loss of organisms on removal of the sampler. This problem can be readily overcome by placing the sampler in a net or mesh bag prior to or during retrieval, and then examining the net for any dislodged animals.

SUMMARY

A variety of artificial substrata have been developed for sampling and monitoring aquatic invertebrates. Basket and multiplate samplers are the most widely used, primarily in rivers where conventional sampling techniques are inappropriate due to factors of depth, current, changes in water level, mobile substrata or limited suitable natural substratum. However, samplers designed for specific sites and specific purposes can be and have been used.

Artificial substrata permit the collection of a numerous and diverse fauna at sampling stations of choice, regardless of bottom conditions. They provide favourable areas for invertebrate colonisation, on a uniform or reproducible habitat on which period of colonisation can be controlled. The provision of a standard substratum thus provides a basis for the collection of comparative samples, against which the effects of the water quality component of the environment may be reasonably assessed.

A minimum colonisation time of 4 weeks is suggested, but long immersion periods render artificial substrata susceptible to damage or loss due to floods, drift debris or vandalism. In some instances they do not provide a substratum similar to that occurring naturally. Coupled with the need to deploy them above the natural substratum, this can result in the sampling of a community that is not fully representative of the naturally occurring invertebrate community. This is not a problem in comparative, as opposed to descriptive or baseline studies.

Artificial substrata have been used successfully in New Zealand streams and rivers. Based on the accounts of their use presented in this paper, and on the author's experience with artificial substrata in the Waikato River, their most useful application to biological monitoring appears to be in the following contexts :

a in short-term (i.e., 1 or 2 immersion periods) intensive surveys of large rivers;
b in monitoring of point or diffuse discharges in small streams;
c in augmenting the information collected by other invertebrate sampling techniques.

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ANALYSIS AND PRESENTATION OF MACROINVERTEBRATE DATA

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ABSTRACT

This paper provides a brief introduction to a variety of techniques for the analysis and presentation of survey or monitoring data on macroinvertebrate communities. Methods of data analysis discussed include descriptive methods, diversity indices, biotic indices and community similarity/ dissimilarity indices. Techniques for the presentation of data, ranging from species lists and percentage community composition to dendrograms, ordinations and principal components analyses are outlined. Examples of the uses of a number of these techniques are given. Reference is made to literature that may be consulted for details of the application of these, and other, techniques.

INTRODUCTION

Biological surveys or monitoring normally result in the collection of complex biotic and environmental data from which patterns and relationships need to be extracted. Clearly, it is not possible to use and comprehend all of the data in its original form, and the problem is then how to reduce or condense that data and to present it clearly and concisely (Hellawell, 1977). A confusing variety of techniques are available for the analysis of such data. To date, in New Zealand, most studies concerned with the biological assessment of water quality using macroinvertebrates essentially have been descriptive (e.g., Hirsch, 1958; Winterbourn et al., 1971; Gibbs & Penny, 1973; Winterbourn & Stark, 1978), perhaps with the use of selected diversity indices. Few studies have attempted to use comparative (e.g., Suckling, 1982) or biotic (Taranaki Catchment Commission, 1984; Stark, in press) indices.

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In this paper, I hope to provide a brief introduction to a variety of methods that can be used for analysis and presentation of macroinvertebrate data, such as that collected from biological monitoring programmes. A complete coverage, or detailed description of techniques is not attempted, but references are given that may be consulted for further information and explanation.

Sound design of the sampling programme is the basis for successful biological monitoring and inherent weakness in sampling cannot be compensated for by subsequent data manipulation (Hellawell, 1977). For the purposes of this paper, I make the assumption that adequate samples have been collected via a well-designed sampling programme (useful discussions concerned with the design of sampling programmes are given by Elliott, 1977; Green, 1979).

Various methods of data analysis and presentation are illustrated in this paper using data from the South Branch, Waimakariri River (Winterbourn *et al*., 1971), the Taranaki Ring Plain Water Resources Survey (Taranaki Catchment Commission, 1984; Stark, in press) and various stream surveys undertaken by Hirsch (1958).

DESCRIPTIVE METHODS OF DATA PRESENTATION AND ANALYSIS

As Winterbourn (1981) stated, descriptive analysis of macroinvertebrate communities in the context of their physico-chemical and biotic environment must form the basis of studies that use macroinvertebrates for assessment of water quality. I believe that this should be the case, also, for most reports detailing the results of biological monitoring programmes. Indeed, an experienced biologist can make a useful assessment simply by inspecting a list of species and their relative and absolute abundances. This assessment certainly is improved when environmental data are available also. However, the results obtained and the conclusions reached must be conveyed to others in a concise and understandable manner.

Taxa lists, taxa-by-site matrices containing presence/absence data, relative abundances, counts (Table 1), percentage composition, or biomass data should be presented as a reference database. If data are extensive, they may be best as appendices to the report (or referenced as being lodged in a computerised data-base). However, data in this form may be relatively

TABLE 1 : Mean numbers of macroinvertebrates per 0.1 m^2 at 6 sites on the South Branch, Waimakariri River (7-24 December 1970). Data from Winterbourn *et al*. (1971)

	Taxon No.	SITE NUMBER 1	2	3	4	5	6
EPHEMEROPTERA							
Deleatidium sp.	1	39.0	95.0	88.6	-	-	-
Coloburiscus humeralis	2	-	0.3	0.6	-	-	-
TRICHOPTERA							
Aoteapsyche sp.	3	-	4.6	-	-	-	-
Helicopsyche sp.	4	-	9.6	11.0	-	-	-
Olinga feredayi	5	2.0	11.0	30.6	-	-	-
Pycnocentria evecta	6	0.6	3.3	0.3	-	-	-
Pycnocentrodes aureola	7	4.6	70.3	44.6	-	-	-
Hudsonema amabilis	8	1.0	1.6	1.0	-	-	-
Oxyethira albiceps	9	68.6	-	0.3	-	-	0.3
Polyplectropus sp.	10	0.6	-	-	-	-	-
Hydrobiosis sp.	11	1.0	1.3	2.0	-	-	-
Psilochorema bidens	12	7.6	2.3	0.6	-	-	-
Neurochorema confusum	13	3.6	1.0	-	-	-	-
DIPTERA							
Tanypodinae	14	1.0	-	-	-	-	-
Maoridiamesa sp.	15	1.0	-	1.0	-	-	-
Orthocladiinae	16	30.0	-	-	-	-	3.0
Chironomus zealandicus	17	2.3	-	0.3	61.0	34.6	0.6
Limnophora sp.	18	-	-	-	-	-	0.5
CRUSTACEA							
Paracalliope fluviatilis	19	13.6	0.6	-	-	-	-
Ostracoda	20	1.6	-	-	-	-	-
MOLLUSCA							
Potamopyrgus antipodarum	21	159.0	111.0	104.0	-	-	-
Physa sp.	22	0.6	-	0.6	-	-	0.6
Gyraulus corinna	23	3.0	-	-	-	-	-
HIRUDINEA							
Glossiphonia sp.	24	1.3	-	-	-	-	-
PLATYHELMINTHES							
Cura pinguis	25	1.6	-	0.3	-	-	-
Rhabdocoela	26	-	-	-	-	-	2.0
OLIGOCHAETA							
Tubificidae	27	9.6	38.6	16.3	308	4780	15.0
Lumbriculus variegatus	28	14.0	1.0	2.3	787	32781	143
Eiseniella tetraedra	29	7.3	0.3	-	-	-	-
NUMBER OF TAXA		24	16	17	3	3	8

incomprehensible to the non-biologist. Therefore, in the main body of the report, graphical representation of data (e.g., kite diagrams or bar graphs showing numbers of taxa, percentage community composition (Fig. 1), or density data) may be used to illustrate changes between sites and the effects of pollution. Common names of taxa may be used also, since many people find long scientific names detract from their comprehension of a report. Wilhm (1972) and Hellawell (1978) provide good examples of the many ways that biotic data may be presented in a clear and understandable manner.

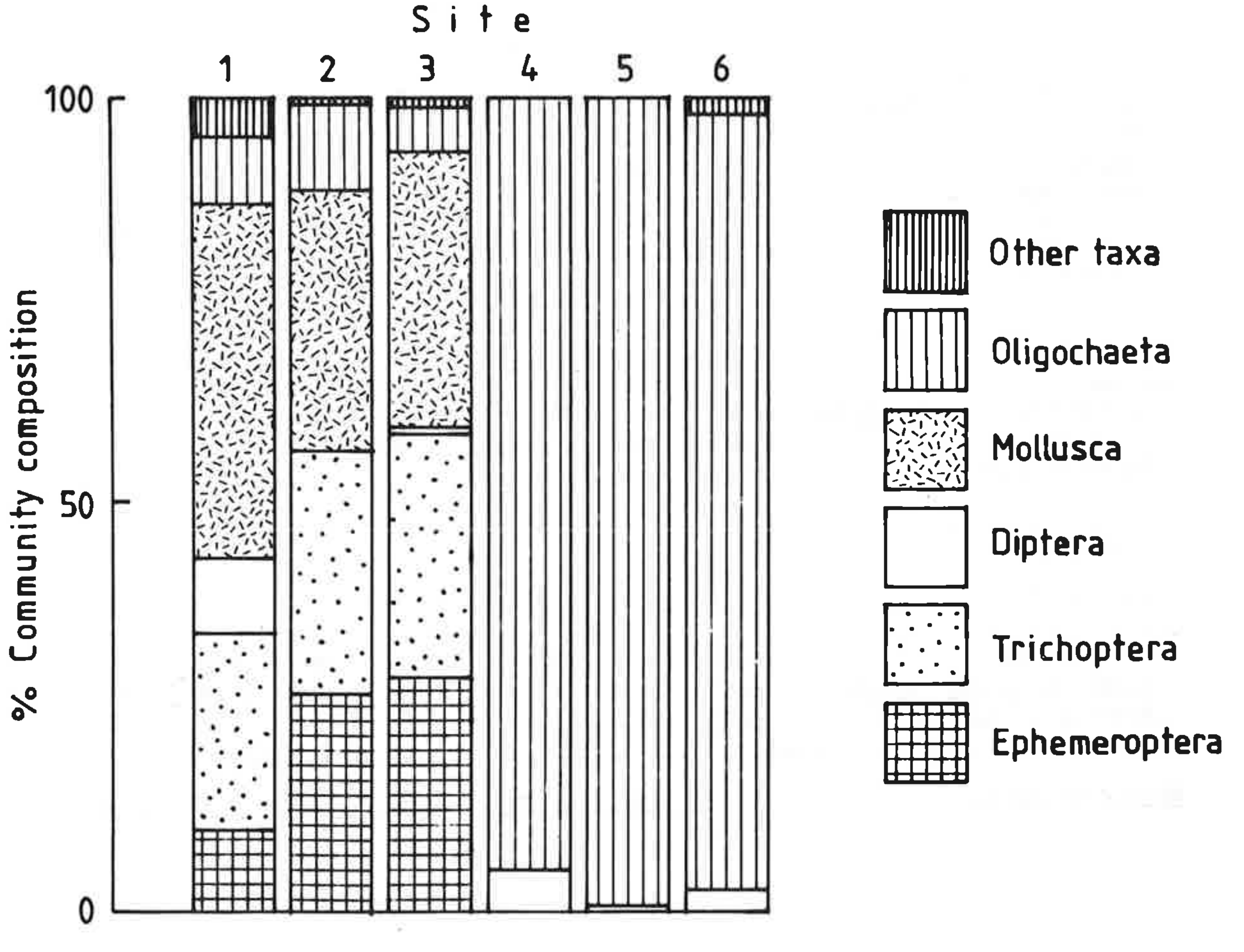

Fig. 1: Percentage community composition, by major macroinvertebrate groups, for 6 sites on the South Branch, Waimakariri River (7-24 December 1970). Freezing works discharges enter between sites 3 & 4 and 4 & 5. Data (Table 1) from Winterbourn *et al*. (1971).

SINGLE-SAMPLE AND MULTIVARIATE DATA PROCESSING

Biological monitoring programmes may generate large volumes of data, which are best stored and analysed using computers. The selection of appropriate methods for the analysis of these data has received considerable attention with the result that there is available a bewildering array of techniques. References that are worth consulting on this matter include Wilhm (1972), Clifford & Stephenson (1975), Hellawell (1977, 1978), Green (1979) and Washington (1984).

Two primary categories of methods for the analysis of biomonitoring data may be identified :

Single-sample or univariate techniques

These methods include biotic and diversity indices and are applied to data from single sites/samples without reference to data collected from other sites/samples.

Multivariate techniques

Multivariate techniques, such as community similarity indices, ordinations and principal components analysis, are desirable for the assessment of spatial or temporal changes by comparing 2 or more populations or community structures.

UNIVARIATE METHODS OF DATA ANALYSIS

Diversity indices

A variety of approaches have been used to formulate diversity indices, but there is general agreement that diversity (in this context) must have both a species richness component and an evenness component (i.e., a measure of the relative abundance of individuals within species) (Washington, 1984). Diversity indices derived from information theory are the best known and most commonly used indices (especially that of Shannon in Shannon & Weaver, 1949), but, at present, these are being questioned as to their biological meaning (Washington, 1984). Some workers (e.g., Margalef, 1958, 1968, 1969, 1972; Wilhm & Dorris, 1966; Wilhm, 1967, 1968, 1970; Dills & Roger, 1974; Pielou, 1975; Cairns, 1977) have found Shannon's H useful or promising, but others (e.g., Hurlbert, 1971; Goodman, 1975; Hilsenhoff, 1977; Hughes, 1978) have criticised its use. This criticism has not been countered by proponents of the information theory indices (Washington, 1984). Washington (1984) concluded that the continued use of H seems to be because of its entrenched nature rather than any belief in its biological relevance and it should be classified as a dubious index.

In summary of diversity indices Washington (1984) concluded that there is a need to find an index that has a meaningful biological explanation and includes both species number and abundance of individuals within species. Hurlbert's PIE (1971) and Keefe & Bergersen's TU (1977) were suggested to offer the most promising approaches to diversity but both require further investigation and evaluation.

Hulbert's PIE (probability of interspecific encounter) (1971)

$$PIE = [N/(N-1)]\ [1- \sum_{i=1}^{s} p_i^2]$$

Keefe and Bergersen's TU (1977)

$$TU = 1 - [n/(n-1)]\ [\ \sum_{i=1}^{s} p_i^2 - 1/n]$$

where s = number of taxa in the sample or population,

N = number of individuals in the population or community,

n = number of individuals in the sample, and

p_i = n_i/n = the fraction of a sample of individuals belonging to species i.

An important point to note concerning the application of diversity indices is that some are meant to be used to estimate the index value from a sample (e.g., TU) and others to calculate the exact value when the sample is treated as a population (i.e., a complete collection) (e.g., PIE). Many indices have versions that may be used in both circumstances.

Biotic indices

Whereas diversity indices may describe biological quality in terms of the structure of the macroinvertebrate community, biotic indices make use of the indicator organism concept. Biotic indices tend to be pollution-type

TABLE 2 : Scores allocated to invertebrate taxa collected from stony riffles, during the Taranaki Ringplain Biological Survey, for use in calculation of Macroinvertebrate Community Index (MCI) values. (After Stark, in press).

EPHEMEROPTERA	
Ameletopsis	10
Nesameletus	9
Coloburiscus	9
Zephlebia	7
Deleatidium	8
Atalophlebioides	9
Mauiulus	5
Austroclima	9
PLECOPTERA	
Stenoperla	10
Megaleptoperla	9
Zelandoperla	10
Zelandobius	5
Acroperla	5
HEMIPTERA	
Microvelia	5
MEGALOPTERA	
Archichauliodes	7
COLEOPTERA	
ELMIDAE	6
PTILODACTYLIDAE	8
STAPHYLINIDAE	5
HYDROPHILIDAE	5
HYDRAENIDAE	8
DYTISCIDAE	5
DIPTERA	
Limonia	6
Aphrophila	5
ERIOPTERINI	9
PSYCHODIDAE	1
Mischoderus	4
TANYPODINAE	5
PODONOMINAE	8
Maoridiamesa	3
ORTHOCLADINIINAE	2
Polypedilum	3
Chironomus	1
TANYTARSINI	3
DIPTERA (Cont.)	
CERATOPOGONIDAE	3
Austrosimulium	3
STRATIOMYIDAE	5
EMPHIDIDAE	3
MUSCIDAE/ANTHOMYIIDAE	3
TABANIDAE	3
TRICHOPTERA	
Orthopsyche	9
Aoteapsyche	4
Polyplectropus	8
Hydrobiosis	5
Psilochorema	8
Neurochorema	6
Hydrochorema	9
Costachorema	7
Tiphobiosis	6
Oxyethira	2
Paroxyethira	2
Pynocentria	7
Beraeoptera	8
Pycnocentrodes	5
Confluens	5
Conuxia	8
Olinga	9
OECONESIDAE	9
Helicopsyche	10
Triplectides	5
CRUSTACEA	
AMPHIPODA	5
OLIGOCHAETA	1
HIRUDINEA	3
PLATYHELMINTHES	3
MOLLUSCA	3
Potamopyrgus	4
Physa	3
Latia	3
ACARINA	5
NEMATODA	3

and geographically specific (Washington, 1984) and so are never likely to be universally applicable. The concept of the indicator organism, whose presence indicates pollution, has been disputed since these taxa may be found in clean streams also (Chandler, 1970). However, it has been suggested that associations or communities of benthic macroinvertebrates may provide a more reliable assessment of pollution than the occurrence of specific taxa (Wilhm, 1970; Winterbourn, 1981; Stark, in press).

Washington (1984) evaluated 19 biotic indices but, of these, only 4 may be applied directly to freshwater macroinvertebrate communities in New Zealand. These indices, essentially, are ratios of tubificid oligochaetes to other taxa (Goodnight & Whitley, 1960; King & Ball, 1964; Brinkhurst, 1966) or, simply, densities of oligochaetes (Wright & Tidd, 1933). Such indices are likely to be too simplistic for widespread application in this country due to the complexity and variability of stream environments (Winterbourn, 1981). However, the principles of those indices considered promising could be applied to New Zealand conditions. Washington (1984) considered that Chandler's (1970) and Chutter's (1972) approaches to development of a biotic index had most merit. Included also in this class of index are Hilsenhoff's Biotic Index (1977), the Biological Monitoring Working Party Score system (BMWP, 1978; Armitage et al, 1983), and the only comprehensive biotic index adapted specifically for New Zealand conditions - viz. the Macroinvertebrate Community Index (MCI) of Stark (in press).

The MCI involves allocation of scores (from 1 to 10) to taxa (mostly genera) with pollution intolerant taxa gaining higher scores than pollution tolerant ones (Table 2). The MCI is applied by summing scores for each taxon present at a site to obtain a site score, dividing this by the number of scoring taxa and multiplying by 20 (a scaling factor). The index ranges from 20, when all taxa present score 1 point each, to 200, when all taxa present score 10 points each. For stony riffles in rivers and streams on the Taranaki ringplain, MCI values greater than 120 indicate clean, unpolluted conditions; values between 100 and 120, slight-to-moderate nutrient enrichment; and values less than 100, moderate-to-gross organic pollution. For detailed methodology concerning the application of the MCI, discussion of its characteristics, further examples of its application, and a version of the index that uses quantitative data (QMCI), reference should be made to Stark (in press).

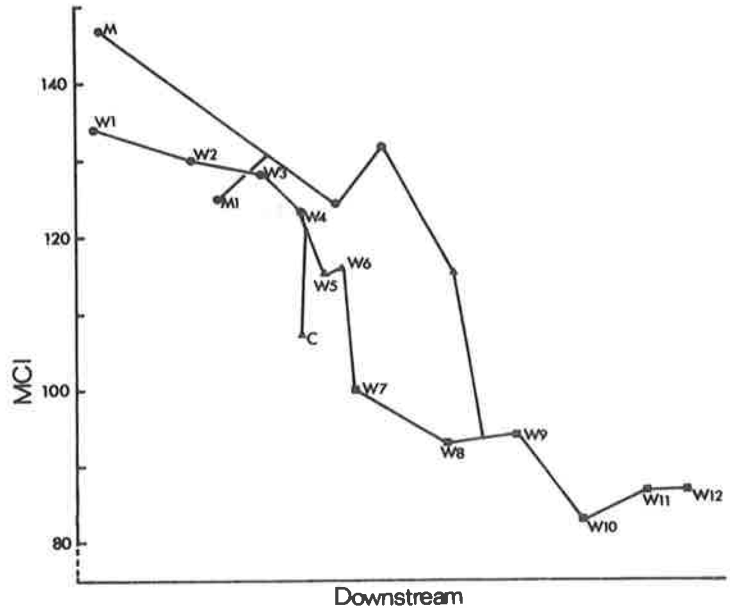

Fig. 2: Macroinvertebrate Community Index values for selected sites on the Waingongoro Stream (W), Mangatoki Stream (M), Mangatoki-iti Stream (M1), and the Climie Stream (C) in the Waingongoro catchment, Taranaki (20-21 July 1983) (after Stark, in press).

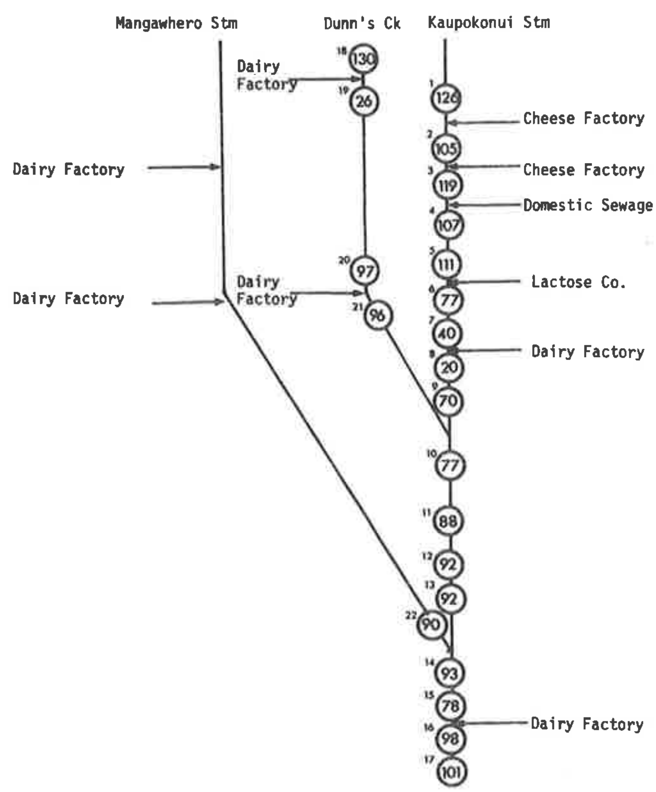

Fig. 3: Diagrammatic representation of the Kaupokonui Stream catchment, Taranaki showing main pollution inputs and Macroinvertebrate Community Index values for 22 sites (March 1957). Data and site numbers from Hirsch (1958). N.B. Not to scale (after Stark, in press).

The results of MCI analysis (or indeed any single-sample statistic) may be tabulated, depicted graphically (Figs. 2, 3) or shown on a map of the study area (Fig. 4).

It must be stressed that biotic indices (and indeed, diversity indices), which can be invaluable in summarising complex biological information into a form understandable by water-managers, must not be regarded as the be all and end all of biological monitoring. There is great danger in collecting information of a quality that is only just sufficient for an index to be applied, or in never analysing data in more sophisticated ways. It is only by detailed quantitative ecological and taxonomic studies that we will learn more about the habitat requirements and tolerances of aquatic taxa. It is this more detailed information that could form the basis for more reliable or more refined biological methods of pollution assessment.

Many authors have developed, evaluated or compared various biotic indices, and for more information on this topic the reader may consult Hellawell (1977, 1978), Mason (1981), Washington (1984) and references therein.

Fig. 5 shows one way of depicting changes in a number of diversity and biotic indices downstream in the South branch of the Waimakariri River. As Mason (1981) has noted, there are situations where various indices may convey little more information than the number of taxa (s)!

Biotic indices should never be compared to diversity indices, especially not as indicators of water quality (Washington, 1984). Diversity indices measure change in community structure, whereas biotic indices measure pollution (usually organic) specifically based upon the reactions of physiologically sensitive organisms. Presence/absence or abundance of sensitive organisms may alter without a major change in community structure. In such cases, there would be a poor correlation between the 2 types of indices. This is not to say, however, that biotic and diversity indices never yield similar interpretations (see Fig. 5). Despite this, a number of authors have compared biotic and diversity indices (e.g., Chutter, 1972; Hilsenhoff, 1977; Mason, 1981) but there appears to be little consensus as to which indices are preferred.

MULTIVARIATE METHODS OF DATA ANALYSIS

Although single-sample analysis is essential in aquatic community

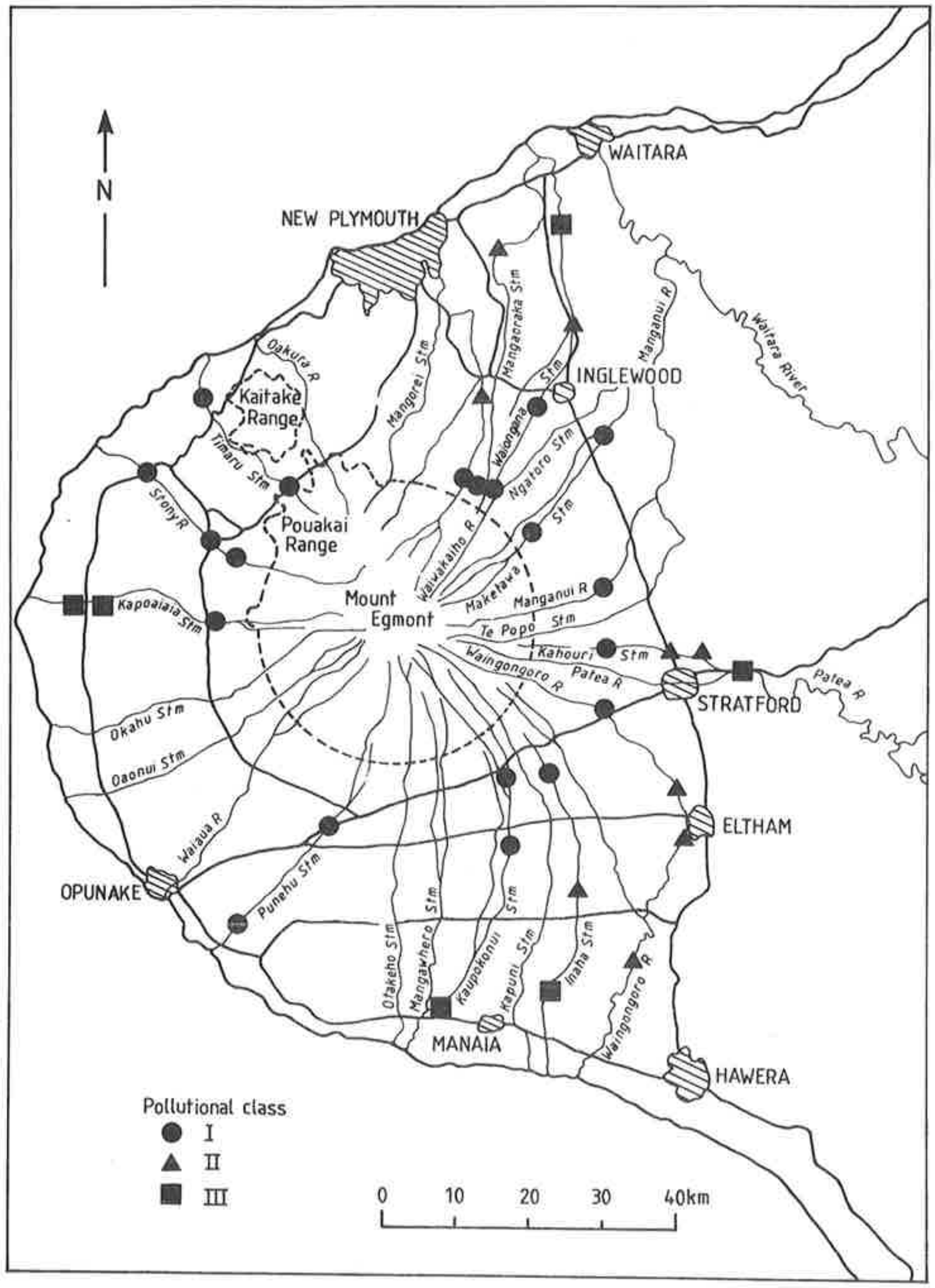

Fig. 4: Pollutional classification of Taranaki ringplain sampling sites based upon Macroinvertebrate Community Index values calculated from combined winter, spring and summer samples. I = pollution insignificant; II = slight - moderate pollution; III = moderate - gross pollution. After Stark (in press).

biomonitoring, analysis often is incomplete unless comparisons are made between samples (e.g., control vs affected site comparisons). The ability to discriminate between samples and to identify the cause(s) of sample differences is a goal of biomonitoring (Herricks & Cairns, 1982).

Multivariate analyses may be performed on quantitative (density), coded abundance, frequency, biomass, or presence/absence data (Field et al., 1982). However, comparisons are valid only if similar sampling and sample analysis procedures are used. Hurlbert (1969) suggested that, since abundances are influenced readily by extraneous factors, presence/absence data may give a less ambiguous measure of association. I don't believe that the situation is that simple; rather the choice of methods must be considered in relation to the aims of a study and the habitat/river types involved. Often there are financial or time constraints that dictate the use of more rapid presence/absence techniques. In any event, one must appreciate that the choice of methods, inevitably, will influence the results and possible conclusions of a study. For this reason, methods must be chosen carefully by suitably experienced persons to ensure that they are appropriate to the aims of the investigations.

Green (1979) has identified 2 broad categories of multivariate analysis, viz. cluster analysis and ordination. Both procedures allow visual interpretation of relationships between samples and may be used in combination to increase analytical capabilities (Herricks & Cairns, 1982).

Walker et al. (1979) identified 3 approaches that could be used for the multivariate analysis of survey data :

1 Search for patterns amongst the biological variables and then attempt to interpret them in terms of environmental data.
2 Search for patterns of relationships between biotic and environmental data simultaneously, e.g., canonical analysis (Cassie, 1972).
3 Search for patterns in the environmental variables followed by a search for related patterns in biotic data.

The third approach requires prior knowledge of which physical variables are likely to be dominant so that they can be measured, but there seems to be most support for the first approach (Day et al., 1971; Green & Vascotto, 1978; Green, 1979; Field et al., 1982). The first strategy

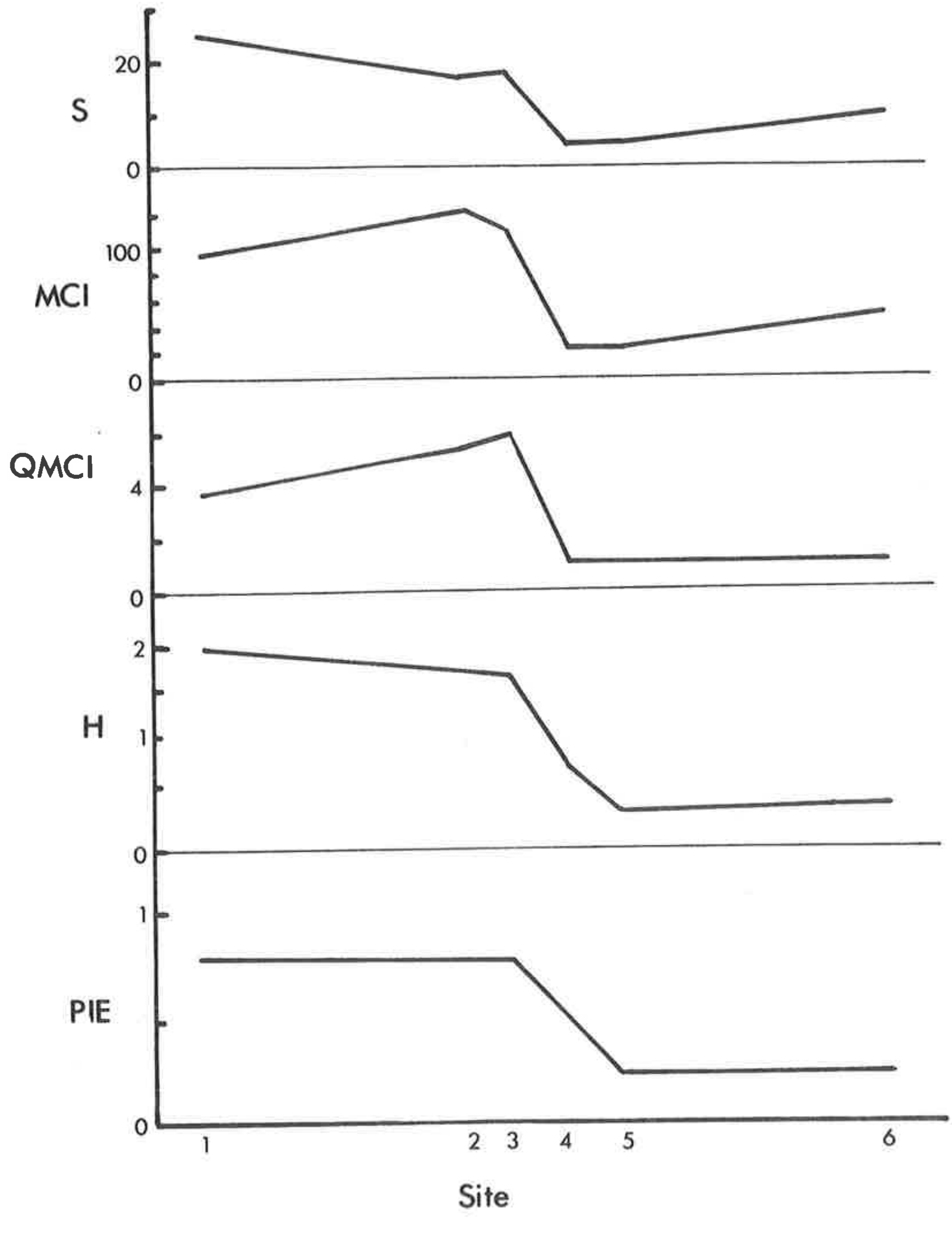

Fig. 5: **Changes in various biotic and diversity indices downstream in the South Branch, Waimakariri River. S = number of taxa; MCI = Macroinvertebrate Community Index; QMCI = MCI using quantitative data; H' = Shannon-Weaver Species Diversity; PIE = Hurlbert's Probability of Interspecific Encounter. Data (Table 1) from Winterbourn *et al*. (1971).**

avoids the influence of previous assumptions about the relationships between environmental and biotic variables, and minimises the danger of circular arguments in seeking to deduce relationships (Field et al., 1982).

Two types of analysis may be performed :

1 Site group analysis (otherwise termed normal, or q-type analysis) in which samples or sites are arranged into groups according to the similarity of their biotic communities (Fig. 6).

2 Taxa group analysis (otherwise termed inverse, or r-type analysis) in which taxa are grouped according to the similarity of their distributions over the sites (Fig. 7).

Data transformation and standardisation

Prior to multivariate analyses, and depending on the characteristics of the data, it may be desirable (or essential) to transform or standardise data. Clifford & Stephenson (1975) and Sokal & Sneath (1963) discuss a number of methods that may be used and the conditions they are appropriate for.

Transformation, which is not required for presence/absence data, alters the entry for each taxon for each sample without reference to the range of scores in the rest of the data. Most commonly used is the logarithmic transformation but square root transformations may be used also.

Standardisation is not required for normal (site group) analysis but may be desirable prior to inverse (taxa group) analysis when quantitative data are used. This is because perfectly correlated taxa, which always occur together, might be separated from one another because their scores were not the same. One recommended standardisation for taxa group analyses involves replacing each data value by :

$$Y_{ij} = 100\left[X_{ij} \Big/ \sum_{j=1}^{n} X_{ij}\right] \qquad \text{(Field et al., 1982)}$$

Comparative indices

Measures of distance, information, correlation, similarity and dissimilarity have been used to summarise the overall similarity between 2 samples/sites taking all taxa into consideration. Many of these indices

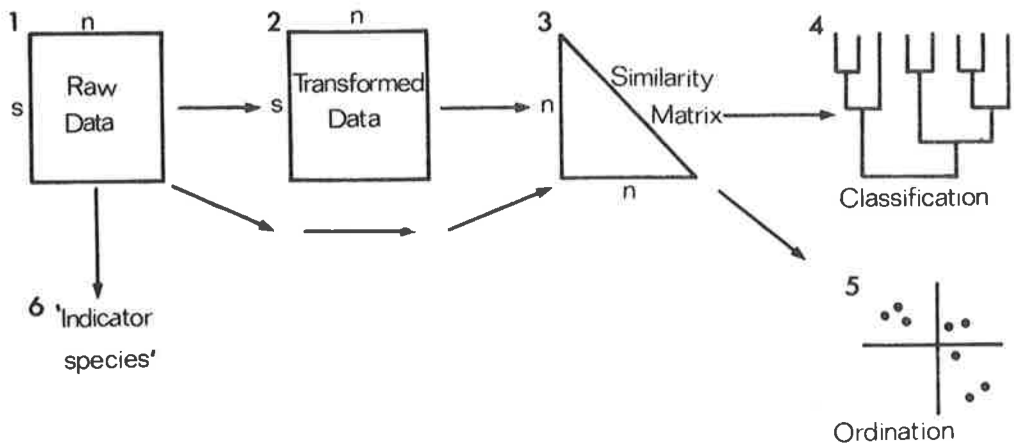

Fig. 6: Site-group (normal or q-type) analysis : summary of stages leading to classification and ordination of samples. Raw data (stage 1) are represented in a matrix of n samples by s species. Data may require transformation (stage 2). The similarity matrix (stage 3) is derived and classified (stage 4) or ordinated (stage 5) to yield complementary pictorial summaries of the relationships between the 8 samples in this diagram. Indicator species (stage 6) are obtained directly from the raw data and aid in interpretation of sample groupings (after Field et al., 1982).

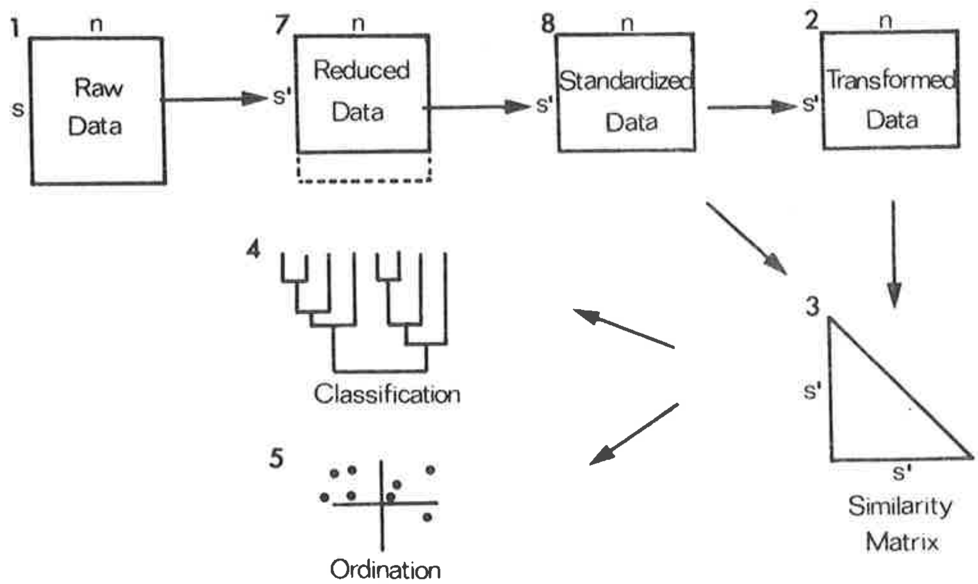

Fig. 7: Taxa-group (inverse or r-type) analysis : summary of stages leading to classification and ordination of species (cf., Fig. 6). Additional steps (7 and 8), data reduction and standardisation, may be required. The classification suggests that there are 2 main clusters of 4 species each; the ordination indicates which individual species are close or distant in distribution (after Field et al., 1982).

and their properties have been summarised by Sokal & Sneath (1963), Sneath & Sokal (1973), Clifford & Stephenson (1975), Hellawell (1977, 1978), Perkins (1983) and Washington (1984). Few different indices seem to have been used in aquatic ecosystems (Washington, 1984).

Qualitative comparisons

Of the indices available for use with presence/absence data, the index of Jaccard (1908) (termed CC or Coefficient of Community by Whittaker & Fairbanks, 1958) is the oldest, simplest, and one of the most satisfactory of those in general use (Clifford & Stephenson, 1975).

$$CC = c/(a + b - c)$$

where a = number of taxa at site A,
b = number of taxa at site B, and
c = number of taxa in common

Jaccard's index ranges in value from 0 (when 2 sites have no taxa in common) to 1 (when both sites have the same taxa present). Sometimes the index is expressed as a percentage (Washington, 1984).

Quantitative comparisons

Comparative community similarity/dissimilarity indices that use quantitative data, and which have been used in aquatic ecosystems, include the Percentage Similarity of Community (PSC) (Whittaker & Fairbanks, 1958), the Bray-Curtis (1957) measure with various later modifications (see Clifford & Stephenson, 1975), Pinkham & Pearson's index (B) (1976), and Euclidean Distance (Clifford & Stephenson, 1975).

Brock (1977) compared PSC and B and concluded that Pinkham and Pearson's B was too sensitive to rare taxa and not sensitive enough to dominants. PSC showed a greater response to changes in dominance and sub-dominance, and was considered less affected by normal sampling error. PSC was preferred for identification of structural/functional differences between communities.

$$PSC = 100-0.5 \sum_{i=1}^{s} |a-b|$$

where a and b are, for a given taxon, the proportion (or percentage) of the total samples A and B which that taxon represents.

Perkins (1983) compared a number of community comparison indices in macrobenthic community bioassays and, although no single index was preferred, it was suggested that community comparison indices hold promise as a group.

In summarising community similarity indices, it would appear that further evaluation of their performance is desirable (Perkins, 1983; Washington, 1984) and that the use of 2 (or more) indices (e.g., Jaccard and PSC), which have different areas of sensitivity, is likely to provide more reliable interpretations (Brock, 1977).

Presentation of results of comparative indices

Comparative community similarity indices may be applied between adjacent sampling dates (to depict changes in community composition between times) or between adjacent sites (e.g., sequentially downstream). Results normally are best represented graphically (similar to Figs. 2 or 5 but with points plotted midway between sampling dates or sites).

More usually, comparative indices are applied to taxa x site data matrices where every site (for site-group analyses) or taxon (for taxa-group analyses) is compared with every other. Results may be presented as a similarity matric or trellis diagram (Fig. 8), dendrogram (Figs. 9, 10), or ordination (Figs. 11, 12).

Similarity matrices and trellis diagrams

Matrices of similarity (or dissimilarity) indices formed from site/taxa group analyses are very informative to one used to interpreting them. However, they rarely are suitable for presenting results to others in an understandable manner. Differential shading of the similarity matrix (termed a trellis diagram) is the simplest technique for recognising, at one glance, groupings among the sites/taxa (Fig. 8). Groups may be found by rearranging rows and columns to obtain highest similarity values (darkest shading) along the principal diagonal of the matrix. Discontinuities in the matrix are identified by searching for blocks of dark or light shading (i.e., high or low similarities) which may be delimited as site/taxa groups. However, this process becomes quite unwieldy with large data sets (Sokal & Sneath, 1963).

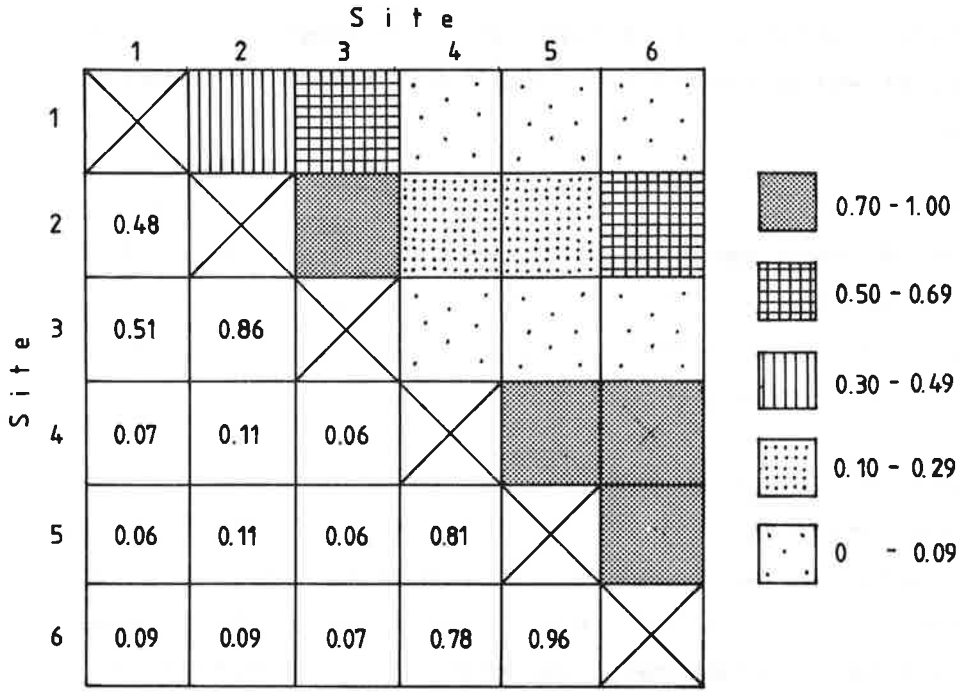

Fig. 8: Trellis diagram (upper half) and matrix of PSC similarity coefficients (lower half) for 6 sites on the South Branch, Waimakariri River. These data are plotted as a dendrogram on Fig. 9c. Data (Table 1) from Winterbourn et al. (1971).

Cluster analysis

A similarity matrix normally is represented graphically as a dendrogram. Although small dendrograms may be constructed without the use of computers using methods such as those described by Hellawell (1978), the calculations are tedious rather than difficult and are best undertaken using a computer. Computers are essential for more complex analyses. BASIC computer programs for dendrogram analyses are given by Orloci (1975) and Spencer (1984).

A variety of clustering strategies are available to produce a dendrogram from the similarity matrix (Sokal & Sneath, 1963; Sneath & Sokal, 1973; Clifford & Stephenson, 1975). The most successful method appears to be the average linkage clustering strategy, which joins 2 groups of samples together at the average level of similarity between all members of one group and all members of the other (Clifford & Stephenson, 1975; Field et al., 1982). This method also is known as 'group average', 'arithmetic average' or 'UPGMA' (unweighted pair group method using arithmetic averages) (Sneath & Sokal, 1973). Just as the similarity or dissimilarity

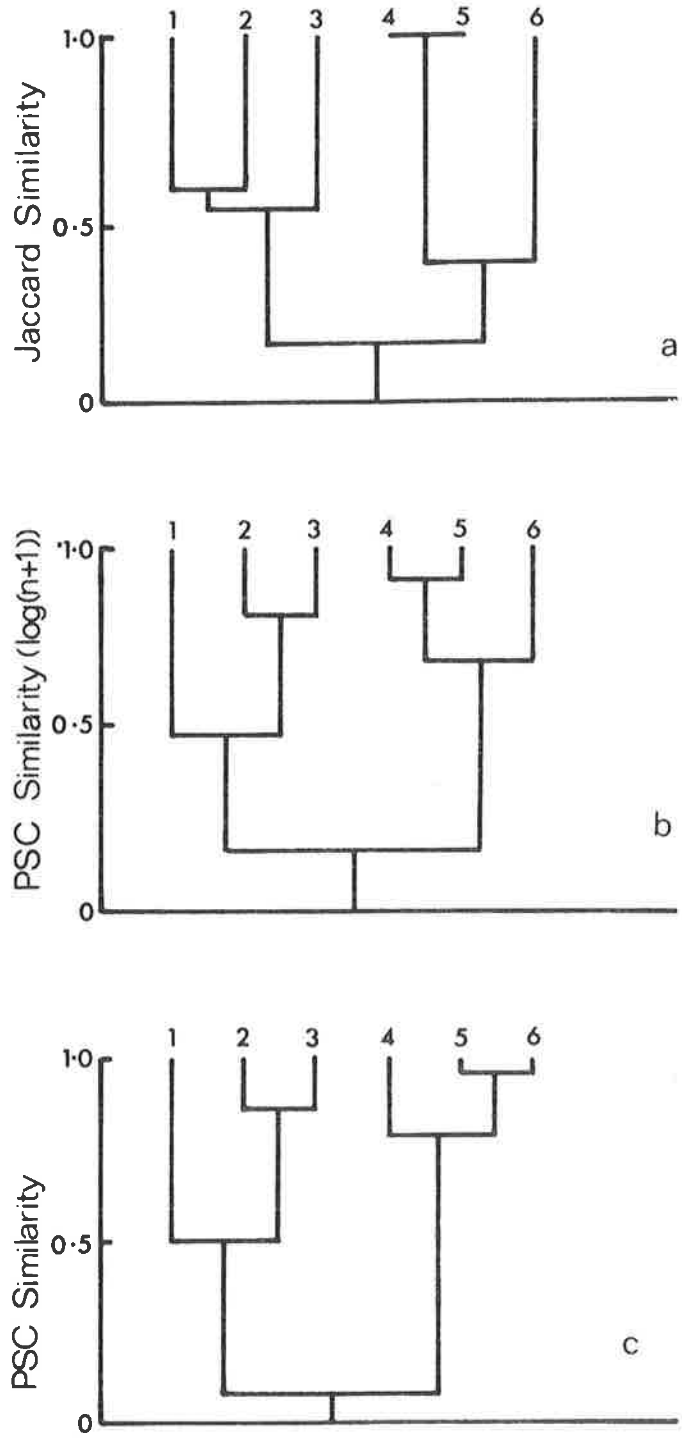

Fig. 9: Average linkage clustering dendrograms of sites in the South Branch, Waimakariri River using (a) presence/absence data, (b) log(n+1) transformed counts and (c) counts of macroinvertebrates. Data (Table 1) from Winterbourn *et al*. (1971).

index chosen can affect the results of an analysis, the choice of clustering strategy will influence the appearance of the dendrogram (see Clifford & Stephenson, 1975).

Dendrograms have the advantage of being simple to understand. Samples are clustered into distinct groups, although the cut-off levels between groups are arbitrary and depend upon convenience. However, Field et al. (1982) listed 4 main disadvantages to dendrograms :

1 The hierarchy is irreversible - once a sample has been placed in a group its identity is lost.
2 It then follows that dendrograms only show inter-group relationships - the level of similarity indicated is the inter-group value.
3 The sequence of individuals (sites, samples or taxa) in a dendrogram is arbitrary and 2 adjacent samples are not necessarily the most similar. This can be illustrated by visualising an upside-down dendrogram as a suspended mobile with threads holding samples/sites/taxa which are free to rotate in a horizontal plane. (Often, however, this sequence can be modified slightly so that it does correspond with some environmental gradient).
4 Cluster analysis will always produce discrete groups of samples/sites/taxa even where there is no pattern in the data (i.e., dendrograms may over-emphasise discontinuities).

In view of these disadvantages, it can be advisable to confirm the results of classification using some other method. Sometimes agreement between a dendrogram and the pattern in the data is so obvious that further confirmation is not really required (as in the Waimakariri River example presented in this paper). In other cases a complementary method (such as an ordination) may be used to determine whether or not the groups delimited on the dendrogram are 'real' (i.e., when the 2 methods agree and the results make ecological sense) (Field et al., 1982). If results are at variance, then interpretations may need to be tempered accordingly.

OTHER TECHNIQUES OF MULTIVARIATE ANALYSIS

Classification or cluster analysis results in the production of groups of samples/sites/taxa from a given set of data. In order to achieve such

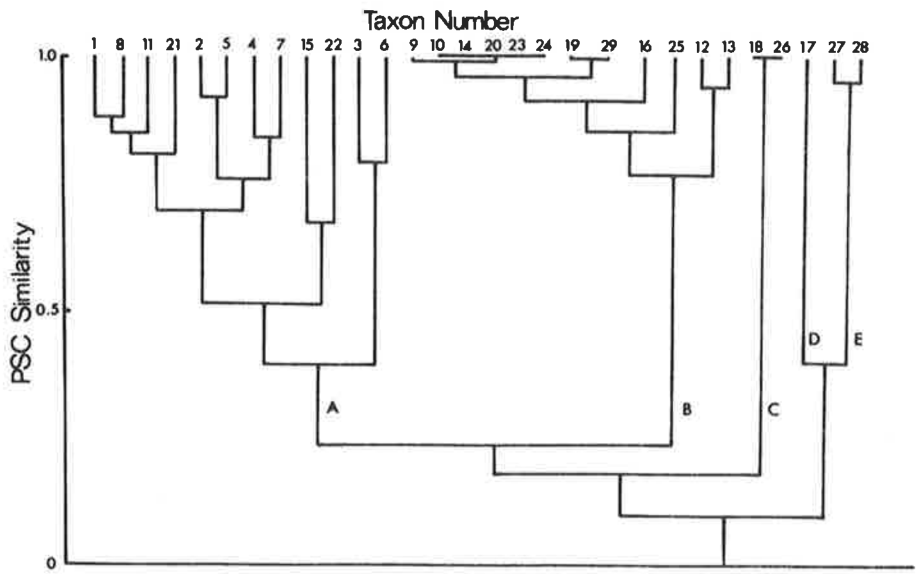

Fig. 10: Average linkage clustering dendrogram of taxa in the South Branch, Waimakariri River. Taxa numbers are given in Table 1. Five taxa groups (A-E) have been delimited. Data (Table 1) from Winterbourn *et al*. (1971).

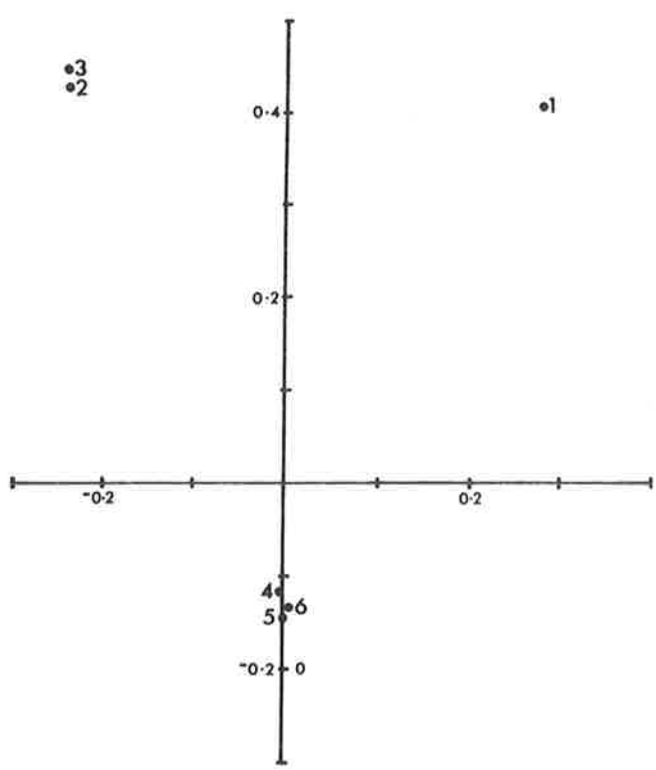

Fig. 11: Reciprocal averaging ordination of 6 sites in the South Branch, Waimakariri River (cf., Fig. 9c). Data (Table 1) from Winterbourn *et al*. (1971).

simplicity (e.g., a two-dimensional dendrogram) there has been a considerable loss of information (Clifford & Stephenson, 1975). To avoid this loss, considerable attention has been given to determination of the best condensation of multidimensional relationships by projection on to a reduced number of axes or planes (usually 2 or 3). Amongst these techniques are various kinds of ordination [principal components analysis (= summarisation), principal coordinates analysis, reciprocal averaging, multidimensional scaling, trend-seeking], and canonical analysis (= multiple discriminant analysis). Discussion of the characteristics and uses of these methods is beyond the scope of this paper, suffice to say that the aims of a study will influence the choice of method. Reference should be made to Clifford & Stephenson (1975), Orloci (1975) and Green (1979) for such discussion.

Examples of reciprocal averaging ordination using data from the South Branch, Waimakariri River (Winterbourn et al., 1971) are given on Figs. 11 and 12. In this case, groupings evident on the site and taxa ordinations are consistent with those obtained on the corresponding dendrograms (Figs. 9, 10).

Ordination has a number of advantages over clustering. Firstly, individual sites (or taxa) retain their identity, and their distances apart on the ordination represent their relative similarity. Secondly, the axes of an ordination are described by particular dominating environmental variables. For the reciprocal averaging ordination of sites for the South Branch (Fig. 11), for example, the 'Y' axis could represent the degree of pollution (increasing downwards), as measured by a suitable water quality variable (e.g., conductivity, BOD). However, of the environmental data collected by Winterbourn et al., (1971) (current velocity, substratum, dissolved oxygen, BOD, conductivity, pH, turbidity and temperature), only current velocity appears to correlate with the X-axis (increasing left).

On the other hand, in the present example, the ordinations took about 5 times longer to run on the computer than the dendrograms.

Determination of taxa characteristic of site groups or sites characteristic of taxa groups

Once the distribution patterns have been summarised in a classification or

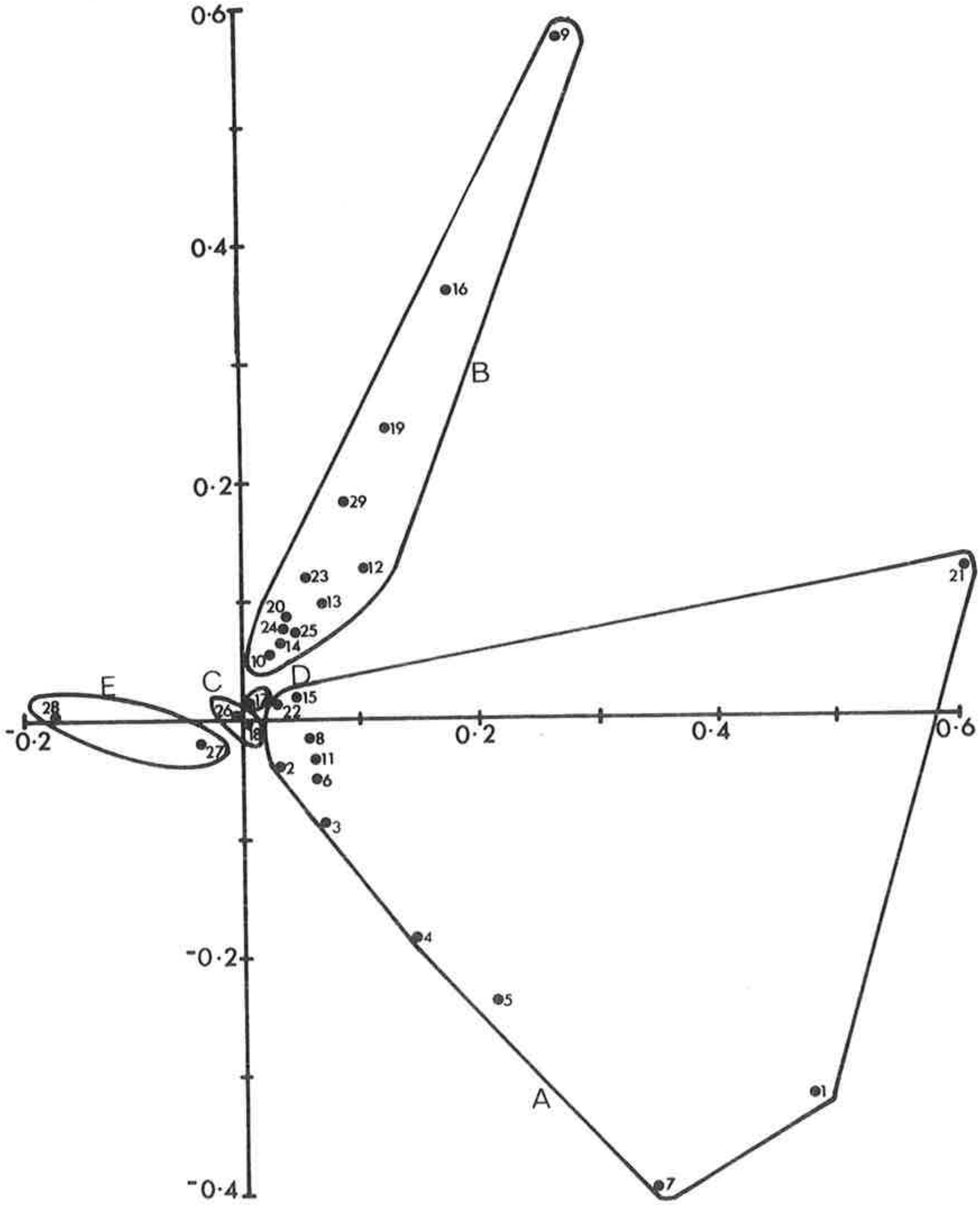

Fig. 12: Reciprocal averaging ordination of taxa in the South Branch, Waimakariri River. Taxa codes are given in Table 1. Taxa groups A-E indicated are those delimited on the PSC similarity dendrogram (Fig. 10). Data (Table 1) from Winterbourn <u>et al</u>. (1971).

an ordination, the taxa differences that caused the patterns have been obscured. This information can be regained only by returning to the taxa-by-site matrix of raw data or frequencies derived from the raw data (Field et al., 1982). Rearrangement of a taxa-by-site matrix with taxa and sites reordered according to their orders on the axes of taxa-group and site-group dendrograms (or ordinations) respectively and then marking in divisions between site and taxa groups can help one to identify taxa or taxa groups characteristic of site groups by means of their presence, or high/low abundance (Table 3). A number of 'statistical' tests are available to determine which taxa are characteristic of one group of sites but absent from another (the Information Statistic; Field et al., 1982), or which taxa differ most in their distribution amongst all site groups simultaneously (the 'pseudo F-test'; Stephenson et al., 1977). These tests are a very useful way of re-examining data in practice, but, since several assumptions are not met, they must not be regarded as statistical tests of significance.

Relationship to environmental data

The aim of most investigations, once groupings have been delimited in the biotic data (and their biological characteristics discussed), is to find environmental factors that are likely to be responsible for the patterns found. One simple method is to construct a trellis diagram showing between-sample biotic similarities in one half and similarities of an environmental variable (e.g., substratum composition) in the other. A mirror image effect suggests a strong environmental relationship (see examples in Green, 1979).

The mapping of biological (e.g., site groups from a dendrogram) and environmental variables onto spatial and/or temporal dimensions can provide an effective display of pattern, for example, related to effects of discharge (Green, 1979).

Several statistical techniques are available for testing relationships between environmental and biotic data (t-test, Mann-Whitney U-test, ANOVA, multiple discriminant analysis)(Green, 1979; Field et al., 1982). The simplest tests involve comparing all observations of an environmental variable for one site group with those for another. The variables that differ significantly are noted as being possible factors responsible for

TABLE 3 : Taxa-by-site matrix reordered according to output from dendrogram analyses (Figs. 9c and 10) showing percentage community composition at 6 sites on the South Branch, Waimakariri River (7-24 December 1970). Site and taxa groups are delimited. Data from Winterbourn *et al*. (1971).

	Taxon No.	SITE NUMBER 1	2	3	4	5	6
Deleatidium sp.	1	10.4	27.0	29.0	-	-	-
Hudsonema amabilis	8	0.3	0.5	0.3	-	-	-
Hydrobiosis sp.	11	0.3	0.4	0.7	-	-	-
Potamopyrgus antipodarum	21	42.5	31.6	34.2	-	-	-
Coloburiscus humeralis	2	-	0.1	0.2	-	-	-
Olinga feredayi	5	0.5	3.1	10.1	-	-	-
Helicopsyche sp.	4	-	2.7	3.6	-	-	-
Pycnocentrodes aureola	7	1.3	20.0	14.6	-	-	-
Maoridiamesa sp.	15	0.3	-	0.3	-	-	-
Physa sp.	22	0.2	-	0.2	-	-	0.4
Aoteapsyche sp.	3	-	1.3	-	-	-	-
Pycnocentria evecta	6	0.2	0.9	0.1	-	-	-
Oxyethira albiceps	9	18.3	-	0.1	-	-	0.2
Polyplectropus sp.	10	0.2	-	-	-	-	-
Tanypodinae	14	0.3	-	-	-	-	-
Ostracoda	20	0.4	-	-	-	-	-
Gyraulus corinna	23	0.8	-	-	-	-	-
Glossiphonia sp.	24	0.3	-	-	-	-	-
Paracalliope fluviatilis	19	3.6	0.2	-	-	-	-
Eiseniella tetraedra	29	1.9	0.1	-	-	-	-
Orthocladiinae	16	8.0	-	-	-	-	1.8
Cura pinguis	25	0.4	-	0.1	-	-	-
Psilochorema bidens	12	2.0	0.7	0.2	-	-	-
Neurochorema confusum	13	1.0	0.3	-	-	-	-
Limnophora sp.	18	-	-	-	-	-	0.3
Rhabdocoela	26	-	-	-	-	-	1.2
Chironomus zealandicus	17	0.6	-	0.1	5.3	0.1	0.4
Tubificidae	27	2.6	11.0	5.4	26.6	12.7	9.1
Lumbriculus variegatus	28	37.0	0.3	0.8	68.1	87.2	86.7

the biotic groups (Field, 1971). This approach has certain disadvantages inherent in repeated significance tests (see Green, 1979; p. 76), and, because comparisons are pairwise, the overall effect of the environmental variables on the observed patterns may not be clear. Multiple discriminant analysis is considered by Green (1979) to be the best method for evaluating the significance of group separation and interpretation.

SUMMARY AND CONCLUSIONS

A monitoring or surveillance programme, which is to be used in water management (i.e., where the results must be understood by non-biologists), should comprise:

a Descriptive analysis of macroinvertebrate communities.

b Site descriptions and measurement of physico-chemical factors likely to influence benthic macroinvertebrate community composition.

 A checklist or questionnaire approach could be used to facilitate this process.

c Pictorial methods of data presentation with raw data included as appendices to the report (or referenced as being stored in a computer data file).

d Single-sample techniques of data analysis such as biotic or diversity indices to reduce complex information to a single number, which may be used to rank sites or, given a suitably calibrated index, to evaluate the pollutional status of a site.

e Multivariate techniques of data analysis (e.g., dendrograms and ordinations), which are preferred for the assessment of spatial or temporal changes, by comparing, for example, control vs affected sites.

 NOTE: If possible, at 2 least complementary methods of data analysis should be used. For example, Jaccard and PSC similarity indices, dendrograms and ordinations. The use of complementary methods may not be necessary when the first method chosen is used merely to depict obvious pattern in the data.

f Identification of environmental factors responsible for the observed biotic pattern.

g Translation of the results of the investigation into practical recommendations that will have direct application in water resource management.

Management should be presented with relevant information, interpretation and recommendations, either in a brief report tailored to these purposes or in specific management-oriented sections as part of a more comprehensive report.

I believe that the onus is on the biologist to provide management with a useful interpretation of the study findings. In turn, I believe that management have an obligation to make best use of their technical/ scientific support by giving such findings due consideration.

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SESSION 6

MONITORING OF FISH

Biological monitoring in freshwaters: proceedings of a seminar. Eds. R.D. Pridmore and A.B. Cooper. Water & Soil Directorate, Ministry of Works and Development for the National Water & Soil Conservation Authority, Wellington, 1985. Water & Soil Miscellaneous Publication 83.

FISHERY VALUES AND WATER QUALITY

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ABSTRACT

Management of water resources requires a knowledge of all values of water, and how these can be protected by controls on water quantity and quality. One of the most important instream values of water is habitat for fish. The current status of work identifying values of New Zealand's freshwater fish stocks is outlined. Fish stocks are defined by species and catchment, and fishery values, which are related to the use or benefit of identifiable fish stocks, are classified into ecological, biological/ scientific, and exploitable categories. The combined fishery value of a catchment may therefore involve several stocks of fish and several categories of values, each of which can only be evaluated with reference to similar fish stocks in other catchments. Once fishery values in a catchment are known, the water quality and quantity limits needed to conserve them can be determined. This is a complex problem and, at present, existing knowledge of the effects of water quality or quantity changes on fish populations can only provide general guidelines. It is therefore recommended that water managers should emphasise trial-and-error approaches to setting water right limits. This will require scientifically determined first-approximations to water quality and quantity needs of fish populations, as well as monitoring of fishery values to assess the scope for further water use, or compensation.

INTRODUCTION

Regional water boards are increasingly turning to water management plans as a basis for managing their water resources. The process involved in producing these plans is likely to involve wide public participation to identify current values of water. Once the values are identified, water

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quality/quantity limits needed to sustain them would be determined and used as a basis for both decisions on water use, and later for monitoring, to uphold the decisions.

In some ways, the preparation, advertising and 5-yearly review of water management plans will follow town and country planning procedures. These procedures will put water management on a more democratic basis than exists at present. But by relying on the definition and monitoring of water quantity/quality limits to manage the resource, water management plans also seek to put decision making on a firmer scientific and objective footing. This approach is commendable, although where fishery values are concerned, it may be coming before its time. The study of links between water quality and fisheries values is in its infancy and it is dangerous to assume that water quality/quantity limits needed to sustain fishery values can (a) be determined and (b) practically monitored to sustain fishery values at present. For example, reviews of biological methods for the assessment of water quality in New Zealand have dealt primarily with bacteria, viruses, plants and invertebrates. In short, all major groups of aquatic flora and fauna except fish (Standing Biological Working Party 1979, 1981). This is not surprising in view of the limited and/or generalised information on the water quality requirements of fish (e.g., Church et al., 1979). Monitoring water quality, either directly or indirectly by biological parameters, is an important part of water management, but the emphasis on monitoring water quality to protect instream values is misplaced. Pridmore (1983) discussed the role of biological monitoring in water management and pointed out that whereas many regard biological monitoring as an alternative to physico-chemical monitoring, it is not. Biological parameters reflect the degree of ecological change or imbalance, whereas physico-chemical measurements are needed to identify the cause of such change. Both forms of monitoring are needed for effective water management and I believe that in many circumstances the quantification and monitoring of fishery values holds more promise as a water management tool, than water quality monitoring. In this paper I intend to spend some time explaining why, and to then describe and explain how the Ministry of Agriculture and Fisheries (MAF) is approaching the identification, measurement and evaluation of fishery values in New Zealand's freshwaters. Emphasis is given to fisheries values, rather than to the water quality limits related to them, because the identification of fisheries values has to precede related water quality studies, and also because the current status of knowledge on water quality requirements of fish populations is so general as to only be of use in framing guidelines. As a result, MAF's current approach to the identification of the water quality limits needed to protect fishery values is to begin identifying the fishery values requiring protection. This is the field of resource assessment which is an important aspect of Fisheries Research Division's Freshwater Resource Assessment and Research Group or FRARG.

IMPORTANCE OF MONITORING FISHERY VALUES

Fishery values need to be monitored by water managers because the effects of changes in water quality or quantity on these values cannot be readily predicted.

Fish populations are generally components of the tertiary trophic level of river or lake ecosystems and depend on production at the primary and secondary levels. In addition, fish are behaviourally more complex than invertebrates or lower orders of life. They interact with a wide range of physical variables and are highly mobile in 3 dimensions. As a result, predictions of fish behaviour, distribution or population dynamics (e.g., growth, mortality, recruitment) often need to be site-specific, and qualified by a large list of constants, which rarely hold for long in nature. This means that whereas fisheries science can sometimes predict the direction (positive or negative) of change in relation to a proposed modification of water quantity/quality, the degree of change is less certain. The importance of this principle was recently demonstrated in a long-term Canadian study. Hecky et al. (1984) identified differences between 2 impact prediction studies and actual experience in Southern Indian Lake after river diversion and impoundment occurred. Predictions at the primary (algal) trophic level were qualitatively correct, but predictions, even qualitative ones, decreased in accuracy with increasing trophic level. The fishery collapsed because of shifts in fishing patterns forced by complex changes in fish distribution and feeding behaviour. These changes were related to alterations in the main species of zooplankton, as well as to increased water turbidity. In retrospect, changes in zooplankton species composition could have been predicted if an overall reduction in water temperature had been correctly predicted, but changes in the fishery would still have been unpredictable because of a lack of information on fish feeding behaviour. Compensation was retrospective and contentious because of failure to consider a decline in the fishery and to agree on principles for compensation before development occurred.

This example illustrates the difficulties of predicting impacts on fish. The current inability of biologists to adequately predict all reactions of fish to a change in water quality/quantity means that, in situations where fisheries values are likely to be affected by cumulative water rights, water managers need to adopt a conservative, trial-and-error approach to setting water quality/quantity limits. Fisheries biologists would still advise water managers on what water quality limits should be used as a starting point and in turn water managers would couple such provisional limits on water rights with both monitoring of fishery values and short term reviews of water rights. Where fisheries values show no change over time, the water rights could be liberalised or further rights allowed. In the frequent situations where a trial-and-error approach is not appropriate (e.g., dams, large irrigation schemes), fisheries values could be monitored to assess the degree of decline below a certain agreed level so that compensation could be awarded.

This approach to water management involves monitoring fishery values directly, rather than indirectly through water quality/quantity limits. The process is illustrated in Fig. 1 where the existing and additional monitoring loops, needed for checking effects of water use on both water quality (e.g., compliance with water rights) and fisheries (impact of right) are compared.

Fishery values also need to be monitored because they constitute a common, widespread and sensitive instream use of water, which is most often in conflict with other water uses. For example, most New Zealand rivers now contain trout fisheries and, whereas most of the actual angling may only take place in middle reaches, the upper tributaries are important for spawning, and lower reaches often provide important feeding areas. A rather localised fishery can therefore depend on a fish population which utilises virtually the whole catchment and requires high water quality throughout. Furthermore, eels and galaxiid species forming the whitebait fisheries, are also present in most New Zealand rivers. Their climbing ability enables them to penetrate to the furthest tributaries so that they are present in most of the smaller streams from the estuary to headwaters. Such cosmopolitan distributions are useful facets for indicator species. Monitoring fisheries values could therefore provide a practical basis for approaching water management in New Zealand until research on water quality links and fisheries values is more advanced. The following is a description of how fishery values in New Zealand are defined and an overview of investigations related to the identification and assessment of fishery values.

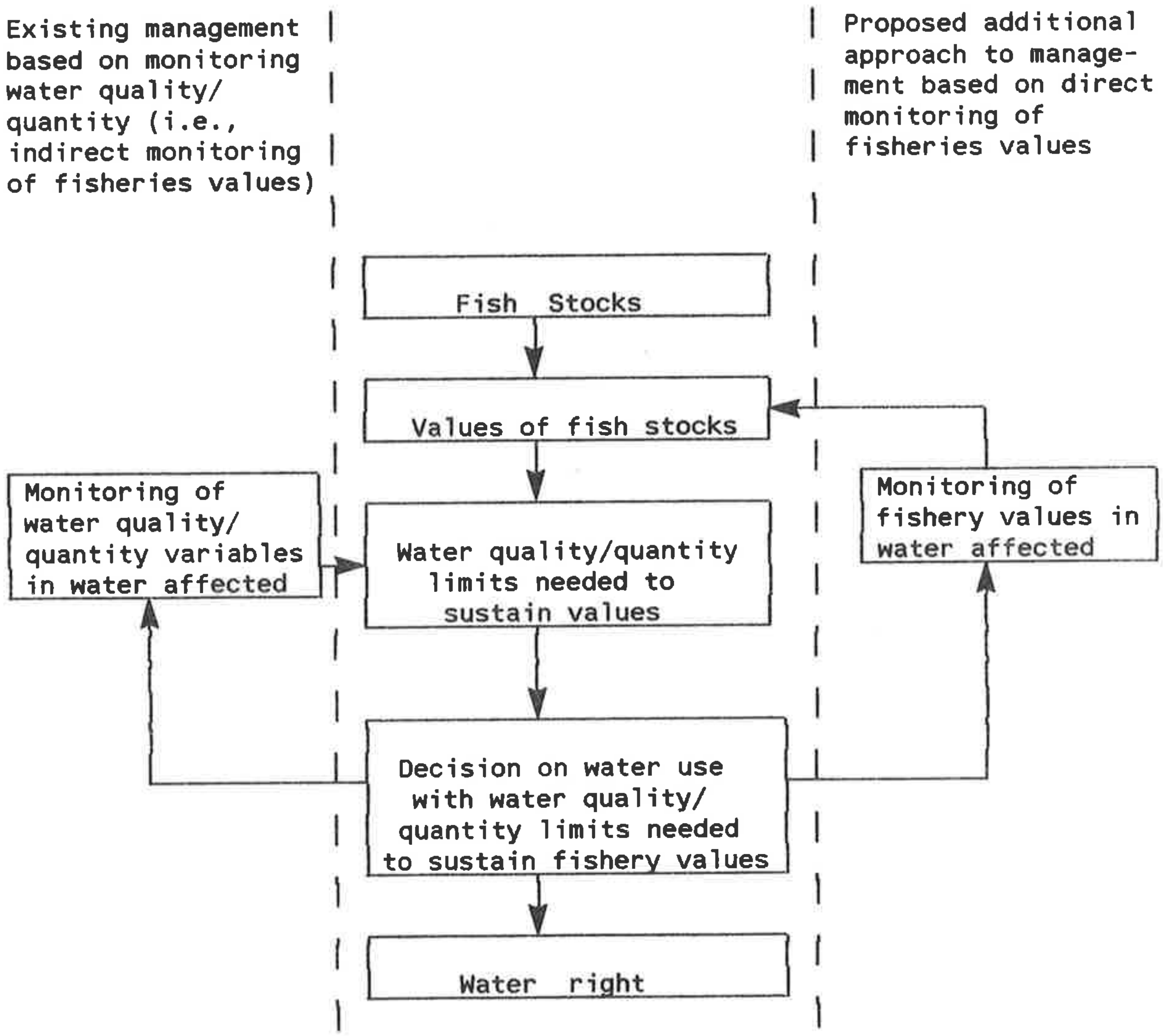


Fig. 1 : Existing and additional approaches to water management needed if fishery values are to be conserved.

VALUES OF FISH STOCKS IN NEW ZEALAND FRESHWATERS

Values of fish stocks in New Zealand need to be determined for management of both water and fisheries resources, but also need to be known before the water quality/ quantity limits required to sustain them can be identified. It is therefore important to define what the fishery values of fish stocks are.

At present, a fish stock is loosely defined as a population of fish forming a fishery resource in a discrete water body (e.g., a whitebait, eel, or trout fishery in a given river, lake, or wetland). The andromous or catadromous nature of some fish (e.g., eels and whitebait), the lumping of several species into one fishery (e.g., whitebait contains 3 main galaxiid species), and the probable presence of several discrete breeding populations of salmonid species within a river catchment means that this definition of fish stocks is biologically unsound. Future fishery management and measurement of values of fish stocks in a given water requires a more practical definition which is provided by reference to a <u>species of fish</u> in conjunction with a discrete water body. For example, a whitebait or eel fishery involving several species of fish would involve several stocks.

Values of fish stocks are defined here as any perceived use or benefit society can, or is gaining, from a fish stock. For example, some freshwater fish stocks provide sport and recreation, some are a commercial resource and others have an ecological significance or are scientifically important. Some fish stocks are valued for several of the above uses (e.g., recreational and commercial use) but overall these fish stocks are few in number and a general classification of values of fish stocks into Ecological, Biological/Scientific and Exploitable Fishery categories is useful (Table 1). Further classification based on specific types of use or benefit within a category is needed if values of fish stocks are to be both measured and compared. This classification is discussed by reference to examples within the 3 major categories of fishery values.

Ecological values of fish stocks

Fish stocks have an ecological value when they are essential for the maintenance of a freshwater ecosystem. If these key species are taken away from some ecosystems, the ecosystem changes significantly. The concept of a key species comes mainly from terrestial environments, where, for example, they may exert a significant controlling influence on some other species which, if left unchecked, would drastically change the environment. A lack of controlling predators in New Zealand probably allowed rabbits, deer and opossum to proliferate with serious consequences for the environment. Key ecological species are likely to be found in low diversity aquatic environments where fish are often the main predators. No examples are known in New Zealand as yet, although a possible key species is the grey mullet in the Waikato River. It is a detritovore, and Wells (1984) estimated that 50 tonnes of mullet would filter 456,250 tonnes of river sediment, incorporating 60 tonnes of carbon into tissue per annum. The 1982 catch in the Waikato River was 280 tonnes of mullet, suggesting that grey mullet could have a significant role in carbon cycling in this river.

TABLE 1: Categories of freshwater fish stocks in New Zealand based on their value to society (after Teirney et al., 1982; Rowe, 1983).

Category		Reason why important
Ecological	a	Species important in maintaining carbon and nutrient pathways in aquatic systems. Species essential to the integrity of lake, river and estuarine ecosystems.
	b	Species essential to fisheries in freshwater or marine ecosystems.
	c	Species indicative of river or lake health (natural bioassay).
Biological/Scientific	a	Rare native species.
	b	Rare acclimatised species.
	c	Rare communities of native species or rare Distributions of native species.
Fisheries	a	Recreational (actual and potential).
	b	Commercial (actual and potential).
	c	Traditional (existing Maori fisheries).

Fish stocks also have an ecological value when they are essential for the maintenance of another heavily utilised fish population. Take the fish stock away and the utilised fish population crashes. New Zealand examples of these key fish stocks could include :

a smelt, which are an essential food supply for the trout fisheries in some central North Island lakes;

b eels, which thin trout populations and ensure good sizes of trout in rivers;

c eels, which are likely to control rudd populations in the few small lakes still supporting trout fisheries (Note : In lakes where eels are absent, rudd proliferate and stunt, taking lures more readily than trout, so ruining trout fishing);

d eels in national parks which are not exploited, so providing a breeding reservoir that may be needed to replenish over-exploited stocks in waters outside national parks.

Finally, fish stocks have a general ecological value as early warning indicators of environmental degradation. In Great Britain trout fisheries in reservoirs are monitored to provide an early warning of water quality problems (pers. comm., Stott, B., MAFF, Great Britain). This use could apply to fish species, particularly those sensitive to changes in water quality. Such species provide a natural bioassay that is often more sensitive than aquatic plants or invertebrates. The ecological importance of trout as a general bioassay organism was demonstrated recently when trout deaths signalled an ammonia spill in a Taranaki stream. One might well ask whether the spill would have been identified if anglers were not concerned for the trout, monitoring them indirectly through their angling trips.

Taking the use of fish as indication of environmental change further, New Zealand's native fish species inhabit many of the small streams and creeks which are being increasingly abstracted for private irrigation. Regional water boards setting minimum flows may well ask, "what is the minimum flow required for such waters to preserve a given fish community and, therefore, preserve the ecology of the stream?" In this context, populations of one of the bully or galaxiid species may prove to be an appropriate benchmark or indicator species, but research will be needed to determine which species and what minimum flows are necessary to support a given density or distribution.

The identification of ecologically important fish species is only just beginning, and is dependent on research to identify the relationships between various fish species and their environment. As knowledge of fish and fish ecology grows so will the list of ecologically important fish species.

Biological/scientific values of fish stocks

Rare populations of animals are probably valued by society because of their scarcity per se, but they are also of value for their genetic complement and for the knowledge which may be gleaned from studies of their behaviour. Useful drugs have been obtained from many plant and animal species, and the study of animal adaptations has lead to many valuable technical and engineering advances.

Rare fish species exist in New Zealand, but do not have a high public profile probably because of their nondescript appearance. They do not compare with birds or tuataras. Nevertheless, they are scientifically and biologically important and those known to be present in lakes and wetlands are listed in Table 2. Apart from the rare native species (e.g., dwarf inanga, black mudfish), this table also contains rare acclimatised species (e.g., lake trout or mackinaw) which are common in North America. These are of biological importance because few stocks exist here and disease risk prevents further stocks being introduced into New Zealand. The existing populations provide the potential for creating new recreational or commercial fisheries in other waters.

Included in Table 2 are populations of fish species which are common in New Zealand, but which are found in unique environments. For example, the inanga (Galaxias maculatus) is the major component species of whitebait and has a marine larval and juvenile stage. However, a few landlocked populations exist in dune lakes and provide a rare opportunity for the study of feeding ecology and distribution, which would otherwise be next to impossible to examine at sea. Similarly, trout have been introduced to so many New Zealand lakes and rivers that pristine populations of native fish are rare. A number of lakes and rivers such as Lake Christabel (population of koaro, Galaxias brevipinnis), the Motu River and the Hauparapara River, Bay of Plenty have been identified as being of biological/scientific importance because they contain populations of native fish, relatively unaffected by introductions of exotic species.

TABLE 2 : Lakes and wetlands known to contain scientifically valuable fish stocks (after Teirney *et al*., 1982).

Lake or wetland	Fishery value
North Island	
Motutangi swamp (Northland)] Kaikino swamp (Northland)] Kaimaumau swamp (Northland)] Whangmarino swamp (Waikato)]	All are important habitats for the endangered black mudfish *Neochanna diversus*).
Lake Waiparera (Northland)	Only known location of probable land-locked inanga (*Galaxias maculatus*) population.
Dune lakes, North Head, Kaipara Harbour	Important habitat for dwarf galaxiid (*Galaxias gracilis*).
Lake Rotopounamu (Taupo)	Landlocked populations of kokopu (*Galaxias brevipinnis*) and smelt (*Retropinna retropinna*).
South Island	
Lake Chalice (Marlborough) and Lake Christabel (Grey River)	Dense and distinctive landlocked populations of *Galaxias brevipinnis*, unmodified by the introduction of exotic fish or catchment changes.
Lake Marion (Canterbury)	Faunistic reserve under Fisheries Act 1908.
Lake Pearson (Canterbury)	Only wild population of Mackinaw or lake char (*Salvelinus namaycush*) in New Zealand.
Lake Emily (Ashburton)	Only self supporting and substantial population of brook char (*Salvelinus fontinalis*) in New Zealand that is of value to anglers.
Hermitage swamp (South Westland)	Extensive lowland swamp of importance to whitebait rearing.
Lake Ohau (South Canterbury)	Supports the major population of sockeye salmon (*Oncorhynchus nerka*) in New Zealand.
Lakes Gunn and Fergus (Eglinton River Valley)	Only residual population of Atlantic salmon (*Salmo salar*) in New Zealand.
Lake George (Southland)	Series of lakes containing exceptional populations of giant kokopu (*Galaxias argenteus*) which is now rare in most parts of New Zealand, although it is still widespread.

The identification and listing of rare species of fish is a continuing process and reflects the extent to which our waters are still being explored by fishery biologists. When a rare population of fish is found and verified, some protection can be obtained by creating a faunistic reserve under the Fisheries Act 1983. Where such protection is not practicable the future of the rare species is dependent on the good will of land and water managers. However, in a planning tribunal decision on the draining of part of the Whangamarino wetland, which is habitat for the rare black mudfish, Judge R.M. Turner stated that there was no provision in the Water and Soil Conservation Act, or the 1981 Amendment (Wild and Scenic Rivers) to protect rare fish species. As a result the habitat of rare species can be modified, and this can be as damaging as direct harvesting or exploitation of the stocks.

Fishery values of fish stocks

The last and largest category of important and valued freshwater fish stocks includes the exploited and exploitable fish populations, or "fisheries". These are classified, mainly by use, into traditional, recreational, and commercial fisheries. There is overlap for some whitebait, eel and salmon fisheries (e.g., both recreational and commercial use occurs) but whereas this poses problems for management, it is not a serious obstacle for measuring fishery values.

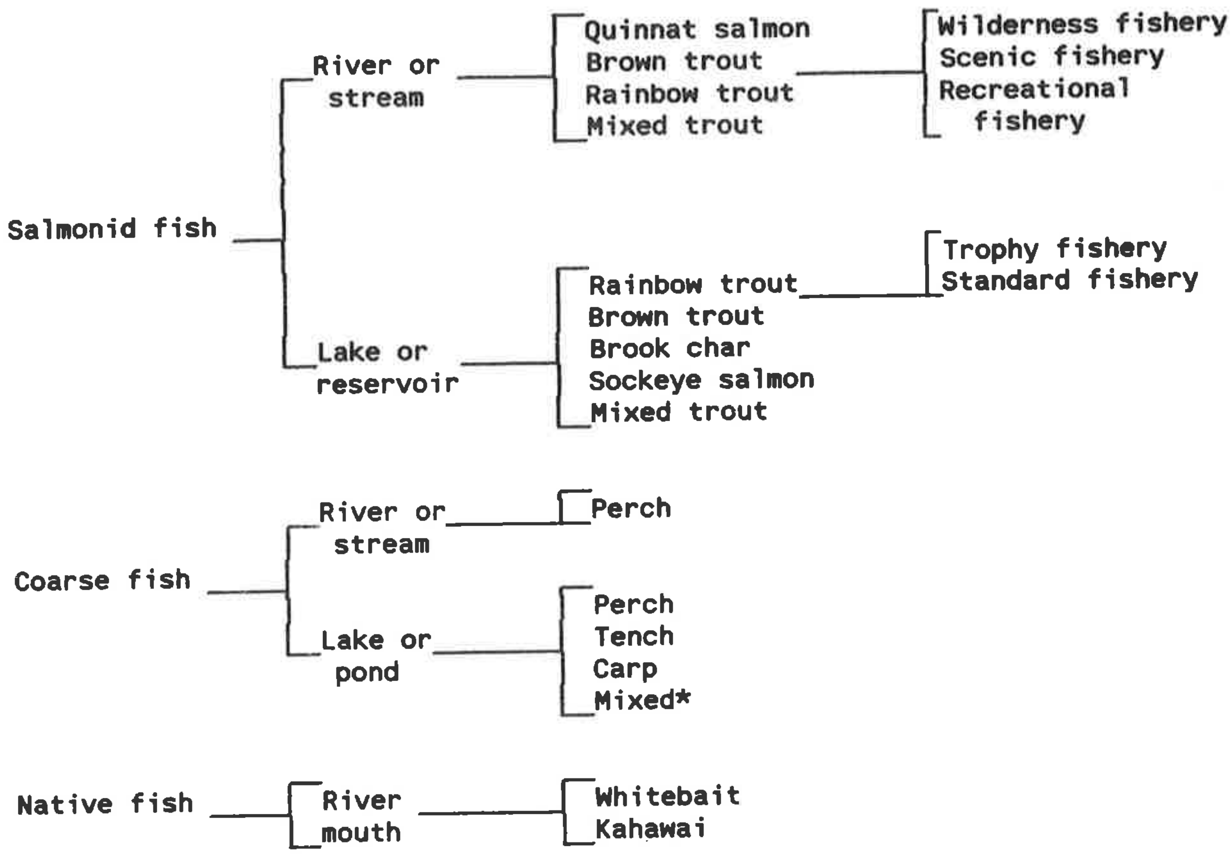
Traditional fisheries include stocks of fish which are exclusively exploited by Maori people, or by others with their consent. Examples are the lamprey fishery in the Wanganui River, the trout fishery in Lake Rotoaira, and the eel fishery in Lake Horowhenua. However, traditional fisheries also include non-exclusive fish stocks that are also exploited for Maori cultural purposes. The identification of such fisheries requires an understanding of the cultural background. Some fish species are delicacies, and are harvested for food to be served at Maraes when large numbers of guests attend rites of passage such as births, baptisms, weddings, conferring of honours, deaths and memorials. An integral part of these ceremonies is the <u>hakari</u>, or feasting, that dictates the dignity of the cultural event. The Marae and tribe acquire respect and status from the quality and quantity of traditional foods, particularly fish, presented at feasts. Consequently, failure to produce these foods leads to a loss of <u>mana</u> (status). This reflects badly on the <u>rangatiratanga</u> (chiefliness) of the chief of a tribe. One of the promises in the Treaty of Waitangi was to protect the rangatiratanga of the chiefs, so maintaining one of the bases for power and the integrity of Maori society. Traditional fisheries have not been listed yet and, although MAF is looking at ways of surveying these fisheries, it is unlikely that they could ever be compared or ranked in value.

By comparison, commercial freshwater fisheries are more readily defined and evaluated. Commercial fishermen are obliged to register with MAF and some are required to keep statistics on catches. As a result, we know what species are being commercially exploited and can often estimate the number of fishermen involved and the yield (Table 3). Difficulties can occur as, for example, with commercial eel fisheries in the Waikato River. Todd (1982) provided comparative catch data for the Waikato basin, but could not apportion yield or catch per unit effort, between the river including its tributaries, and the connecting lake fisheries.

TABLE 3: Sizes of some commercial fisheries in New Zealand freshwaters.

Species	Location	Yields per annum (tonnes)	Approximate value
Eels	Waikato River	400-500 (1980)	$1,000,000
Mullet	Waikato River	285 (1982)	$ 250,000
Catfish	Waikato River	40+ (1984)	-
Whitebait	Waikato River	10 (1980)	$ 200,000
Eels	Lake Ellesmere	300 (1980)	$ 700,000
Flounder	Lake Ellesmere	184 (1980)	-
Whitebait	Cascade River (SI)	10.5 (15yr avg)	-
Smelt	Canterbury rivers	15 (1982)	$ 7,500
Salmon[1]	South Island	200-400 (1985)	-

[1] from pond and cage rearing (projections for 1989 are 4000-5000 tonnes which includes returns from salmon ranching).



(* rudd are fished for but are classified as a noxious fish at present)

Fig. 2: Types of recreational fisheries present in New Zealand freshwaters.

Commercial fishermen are likely to range widely, depending upon their relative success, and so will exploit a number of stocks in different waters. Apart from Todd's (1982) analysis of the Waikato Basin eel fishery and an early analysis of the Lake Ellesmere eel fishery, surveys of commercial fisheries in freshwaters are rare. Wheeler (1983) reported on mullet fisheries, also in the Waikato River, but no inventory of New Zealand freshwaters providing major commercial fisheries has been produced, and comparative analyses of data on yields, the number of fishermen involved (full and part time) and catch per unit effort have not been undertaken. Provided the data are available, the ranking of commercial fisheries should be a relatively straightforward exercise and could be based on yield (kg of fish/annum), economic worth, the number of fishermen and/or boats involved (full and part time), or a combination of these values.

Compared with commercial fisheries, recreational fisheries values are complex and are based on a range of sporting, social, recreational and aesthetic attributes which are not readily quantified or comparable. Nevertheless, most progress in assessing fishery values has been achieved with recreational fisheries. This is because they involve large numbers of people (over 200,000 New Zealand licence holders) and are important in both the domestic and international tourist industry. Recreational fisheries in New Zealand's freshwaters are based on 3 types of fish; salmonid species, coarse fish species, and native fish species. The species involved, and therefore the types of fishery, are classified in Fig. 2 for river, lake, and pond environments.

Recently, MAF surveyed anglers to assess the fishery values of New Zealand rivers supporting salmonid fisheries. Results from this national river angling survey have been used to identify nationally important rivers (Teirney et al., 1982) and these rivers were classified into wilderness, scenic, and recreational fisheries. The basis for this classification and identification was an analysis of a combination of values, including usage (numbers of angler visits per annum), overall importance grade awarded by anglers (on a scale of 1 to 5), quality of the fishery (size and catch rate of fish) as well as scenic, wilderness and other recreational values provided in addition to angling values. Regionally and locally significant angling rivers have also been identified for some districts (South Canterbury, Waitaki, Taranaki, Marlborough, Auckland, Hawke's Bay, Nelson) and are being steadily identified for the rest of New Zealand. Salmonid fisheries in lakes were excluded from this survey and fishery values of lakes will need to be assessed separately. The large number of parameters that could be involved in the evaluation of salmonid fisheries in lakes is listed in Table 4.

Whereas trout and salmon fisheries dominate recreational fisheries in New Zealand, there is a growing interest in coarse fishing. Coarse fishing is very different to trout angling and consequently values will be different. Coarse fishing usually involves the capture, liveholding and eventual

TABLE 4 : Criteria and parameters for selecting lakes with important salamonid fisheries (after Teirney *et al*., 1982).

Criteria	Parameters
Angling effort and use	- number of person-hours spent fishing per annum - number of people fishing/annum - number of anglers spending more than one day on a fishing trip to a particular fishery - extent of tourism related to fishery values - numbers of overseas and/or non-local anglers fishing
Species diversity	- number of fish species present
Fish abundance	- catch rates - bag limit
Quality of fish	- average size - fighting reputation - incidence of parasites - condition factor
Angling methods and restrictions	- number of legally permitted angling methods - percentage of anglers using different methods - extent of season - restrictions on night fishing - restrictions on stream fishing - restrictions on methods related to time or location
Recreational features	- boat ramps - local accommodation - road access - access to fishing locations - variety of fishing locations - opportunity for other pursuits (swimming, picnicking, etc.)
Wild and scenic attributes	- catchment scenically attractive (forested or bushed hills, diversity of landscape features) - absence of visible land use (e.g. pastures, forestry operations) - water quality (secchi disc measurements) - absence of visible man-made structures (housing, jetties, etc.) - attractive natural shoreline features (e.g. sand beaches, riparian vegetation)

release of a variety of fish species distinguished from salmonids by their coarser scales. In New Zealand such fish include perch, tench, carp and goldfish (rudd are caught but are a noxious fish in New Zealand at present).

Coarse fishing competitions are popular and an anglers daily catch is scored depending on the difficulty of capture of fish (some species require more skill than others), and the number and size of fish caught. World record sized perch and possibly tench are present in some lakes and ponds in the Waikato region and the presence of such fish would considerably enhance the fishery values of these waters. McDowall (1984) has recently reviewed the distribution and status of coarse fish species in New Zealand but, apart from his report, very little is known about the extent or nature of coarse fishing in New Zealand, let alone the values of different coarse fishing waters.

The final group of recreational fisheries involves native fish species such as whitebait and kahawai. Whitebait fisheries occur at river mouths and, unlike most other fisheries, are based on a mix of species. The variability in fishing effort and catch for these fisheries make assessing the importance of a given river mouth for whitebaiting difficult. Mean numbers of people per river per day are not considered sufficient to accurately reflect the popularity of a river for whitebaiting, and peak head counts over several whitebaiting seasons are needed to adequately rank whitebait fisheries. In general the value of a whitebaiting river is related to its popularity, which depends on its yield and proximity to a large population centre. A further refinement of fishery values is perceived by people who distinguish smelt from the galaxiid species, and who prefer koaro whitebait (*Galaxias brevipinnis*) to inanga whitebait (*Galaxias maculatus*). Comparative values of some whitebait fisheries have been studied in detail for Taranaki rivers (Taranaki Catchment Commission, 1981) and Bay of Plenty rivers (Saxton & Rowe, 1984). Recreational kahawai fisheries have only recently received attention. A study of kahawai fishing in the Motu River suggested that the high daily catch rate (6.5 kahawai per person per day) was the major attraction and value for this popular fishery which attracted over 800 people in the January-April period 1982 (Ritchie *et al*., 1982). An inventory of kahawai fisheries in river mouths is being prepared (unpublished data, B. Penlington, MAF, Rotorua).

Other native fish (mullet, flounder, eels) are caught for food but don't form easily recognised fisheries such as kahawai and whitebait.

CONCLUSION

To summarise, fishery values fall into 3 distinct categories (ecological, biological/scientific and exploitable fisheries), and can be classified further within these categories depending on the use of the fish stocks involved. Such a classification forms the basis for surveying and comparing fishery values in New Zealand freshwaters. A start has been made, but a considerable amount of work remains. Both water and fishery managers need to take an increasingly larger role in assessing fishery values in their respective regions, but there is also a need for a national overview to standardise assessment methods. Fisheries Research Division, MAF will probably have a significant role in this, and at present is intending to explore ways and means of assessing salmonid fisheries in lakes and traditional fisheries throughout New Zealand.

Other priorities preclude any major expansion of work into the field of resource assessment, consequently the water quality/quantity limits needed to sustain fishery values are a long way from being known. Nevertheless, a framework for identifying and assessing fishery values now exists, and methodologies needed to monitor fisheries values are available and are discussed in other papers in this seminar.

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FRESHWATER FISHERIES VALUES AND IMPACT ASSESSMENT

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ABSTRACT

The fisheries value of a waterway is defined as the quality or combination of qualities of a fish stock that people regard as desirable or distinctive, such as recreational trout fishing, preservation of an endangered fish species, or the presence of a commercial eel fishery. Ultimately it is the gain or loss of such fisheries values which determine whether an impact is perceived as significant or not. Types of impact assessment vary according to whether the impact has occurred (post-impact assessment), has yet to occur (pre-impact), or occurs during the period of interest (before and after study). Emphasis is placed on identifying the important physical and biological components of the study as a pre-requisite to defining the study objectives. The proposed study must then be adequately designed to answer these objectives. The importance of providing a clear interpretation of study results is also stressed.

The use of fish in biological monitoring is reviewed. Fish have a number of advantages; for instance, as fish are the top predators in most aquatic food chains, fish abundance is an indicator of well-being at lower trophic levels. Disadvantages of using fish include their mobility and relatively long life which may result in several years delay between an impact occurring and the results being evident. Two recent New Zealand studies involving use of freshwater fish in impact assessment are outlined.

INTRODUCTION

Although New Zealand is well endowed with freshwater resources, there are increasing demands being made on these resources. The multiple use concept is implicit in the 1967 Water and Soil Conservation Act, although, historically the balance of sharing has been heavily in the favour of

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Biological monitoring in freshwaters: proceedings of a seminar. Eds. R.D. Pridmore and A.B. Cooper. Water & Soil Directorate, Ministry of Works and Development for the National Water & Soil Conservation Authority, Wellington, 1985. Water & Soil Miscellaneous Publication 83.

single purpose use of water for 'industry' (hydro-electric power, irrigation, food processing, etc). Forests have been felled, swamps drained, rivers straightened, and so on; although more 'productive' land may result and rivers perform more efficiently as conveyors of flood waters or wastes, the aquatic ecosystem is the loser. The extinction of one indigenous freshwater fish, the grayling (Prototroctes oxyrhychus) is thought to be due to loss of habitat and displacement by introduced fish species, while the national decline in whitebait catches is attributable to loss of habitat (McDowall, 1978).

Today with 80% of New Zealand's population urbanised, and with increasing affluence, mobility and leisure time, there is an associated increase in aquatic recreation. At the same time there are substantial economic incentives available from Government to promote further development of natural resources. Considerable potential exists for conflict between these 2 interests. Also with an increasing public awareness and concern over environmental impacts, there is an obvious need for proper planning and evaluation of significant development proposals.

Within government the responsibility for environmental planning is vested with the Commission for the Environment. The Commission has defined environmental impact assessment as "a process whereby a conscious and systematic effort is made to assess the environmental consequences of choosing between various options which may be open to the decision maker .... assessment must begin at the inception of a proposal, when there is a real choice between various courses of action including the alternative of doing nothing. It must be an integral part of the decision-making process proceeding through all the developmental stages of a proposal through to actual implementation" (Commission for the Environment, 1981).

Impact assessment in freshwater fisheries covers a broad spectrum. At a relatively simple level it may involve a laboratory bioassay or a field survey to study whether a physical obstacle in a stream impedes fish passage. More complex assessments are the impacts of proposed hydro-electric schemes, or lignite mining and processing, on fishery values. Not only may the scale of impact change, but the fisheries values affected may change.

For the purposes of this paper, the term 'fisheries value' is defined as being the particular quality or combination of qualities which people relate to or regard as distinctive. Thus freshwater fish and fisheries can be broadly classified as to whether their value is intrinsic (ecological, scientific, zoogeographical, etc.), recreational, commercial, or traditional (Maori fisheries). Within this classification it is possible to further define values (either actual or potential). For instance, the fisheries value of a river could include one or all of the following : a recreational trout fishery, a traditional Maori lamprey fishery, a habitat for an endangered species, a recreational whitebait fishery.

Identification of these values is critical at the planning stage to ensure that the way in which they may be affected is adequately studied. Having identified the fisheries values of the area concerned, the degree to which impacts may be allowed will be determined by the sensitivity of the value(s) to potential change (a scientific assessment), and the amount of change that people (fishery managers, politicians, general public) are prepared to accept (a value judgement - a reflection of how 'important' the value is regarded to be). For instance, the impact of a particular effluent discharge may be unacceptable in river A, where the main fisheries value is a high quality trout fishery (an 'important' value and a sensitive species), but acceptable in river B where the main value is that of a recreational eel fishery (presumably a 'less important' value, and less sensitive species). The actual amount of change is determined by carrying out the appropriate monitoring programme and measuring the appropriate variables.

Of course, in addition to fishery values, water has various other 'positive' values - agricultural, hydro-electric power, industrial, recreational, aesthetic enjoyment etc. - and 'negative' values - erosion and flooding. Impact assessment decisions are based on the gains and losses in these values and it is frequently necessary for decision makers to balance the diverse ecological, political, and moral values which would be effected by a certain course of action.

Given the breadth of topic, "Freshwater Fisheries Values and Impact Assessment", the present paper is designed to give an overview of the types and format of fisheries impact assessment studies and an introduction to the extensive literature available. It does not deal with the statistical features of survey design and data analysis. For information on these topics, readers are referred to Eberhardt (1976) and Green (1979).

TYPES OF IMPACT ASSESSMENT

Impacts

Impacts affect fishery values primarily through changes in the quality and quantity of fish habitats. Types of impacts which affect the quality of habitats include effluent addition, sediment loading, temperature variations, dissolved oxygen levels, etc. In addition to these physico-chemical impacts, there are associated aesthetic impacts; for example, an effluent discharge may reduce the appeal of a river for recreational fishing. Quantity effects on habitats include the impacts of water abstraction, drainage, variable discharges, and lake levels.

A generalised list of types of impacts and their effects on fishery values is given in Table 1.

Assessments

Types of assessments carried out are determined by a combination of the availability of time, money and resources, and practicality. The simplest type of assessment involves the intuitive judgement of an experienced fisheries biologist. Such expert opinion is expedient but highly subjective. Although widely used in public hearings, expert opinion carries with it the risk of rebuttal by a conflicting expert opinion. 'Desk top' assessments may be done based on comparable studies contained in scientific literature. In 'minor' impacts, this is frequently all that is called for; the most frequent concern in this approach is whether the results obtained in one situation can be applied to another. Other types of assessments involve the manipulation of field data and are

a pre-impact studies;

b post-impact studies;

c combination of pre- and post-impact studies.

The pre-impact study involves prediction of impacts based on an appropriate study programme. This is the approach adopted by the Commission for the Environment and allows the assessment of options prior to any impact

Table 1: Catchment modifications and consequent changes (generalised) which affect fisheries values.

Changes Affecting Fisheries Values	Catchment Modifications						
	Hydro/ reservoir	Abstraction	Mining	Channelisation and drainage	Agriculture/ horticulture	Forestry	Industrial domestic discharge
A Changes affecting water/habitat quality							
Turbidity	•	•	+	•	•	+	++
Sedimentation	+	•	+++	•	+	+++	+
Temperature	++	+	•	•	•	+	+
pH	+	•	+	•	+	+	++
Dissolved oxygen	+	•	+	•	+	+	++
Nutrients	•	•	•	•	+++	+	+++
Illumination	•	•	•	•	+	+++	•
Instream cover	+	++	+	+++	+	+	+
Riparian cover	•	•	+	++	++	+++	++
Substrate stability	++	•	•	•	•	•	•
Habitat diversity	++	++	+	+++	+	+	+
Physical obstructions	+++	++	•	•	•	+	+
B Changes affecting water/habitat quantity							
Catchment runoff	•	•	•	++	++	++	++
Seasonal flow pattern	+++	++	•	+	+	+	•
Flood frequency/magnitude	++	•	•	++	•	+	•
Low flow frequency/magnitude	++	+++	•	+	+	++	•
Groundwater	•	•	+	++	•	+	•
Fluctuating flows/levels	+++	+	•	•	•	•	•
Substantial/complete habitat loss	++	++	+	+++	++	•	•

+++ = major effect
++ = moderate effect
+ = minor effect
• = effects usually not significant

occurring. Recent examples are impact assessments and reports associated with hydro-electric development of the Upper Clutha (e.g., Ministry of Works and Development, 1982).

Post-impact studies involve the measurement of impacts after they have occurred. For example, studies on the Whakapapa River (Richardson & Teirney, 1982) and the interactions of native and introduced fishes (McDowall, 1968). Combined pre- and post-impact ("before and after") studies enable pre-impact predictions to be monitored by post-impact studies. New Zealand examples include the effects of logging on fisheries (Graynoth, 1979) and the effects of removing eels on a brown trout population (Burnet, 1968).

Refinements of the above types of assessments are extensive studies (single observations at many sites) or intensive studies (many observations at the same site), use of controls (non impacted areas), use of modelling, use of experimental simulation, and use of small scale experimental perturbation.

The relative merits of extensive and intensive studies for assessing catchment management practices were reviewed by Hall _et al_. (1978). They concluded that the extensive post-impact study using controls provided the most useful information - it gave the widest perspective both spatially and temporally, although the lack of pre-impact data were a disadvantage. Use of controls is stressed by Green (1979) who proposed 4 pre-requisites for optimal impact study design :

1 both pre- and post-impact data collected (temporal control);
2 type of impact and time and place of occurrence must be known;
3 able to measure all relevant biological and environmental variables associated with individual samples;
4 a control area is available (spatial control).

The place of simulation modelling in environmental assessment has been reviewed by Frenkiel & Goodall (1978) and Lauenroth _et al_. (1983). Computer simulation allows large quantities of data from interacting components of the ecosystem to be stored and processed. While modelling has considerable advantages in identifying key variables and relationships, comparing alternative management strategies and so on, it has limited predictive capabilities especially for organisms at high trophic levels like fish. Examples of the use of computer modelling in studies on freshwater fish stocks in New Zealand are Jellyman & Ryan (1983) (time-series modelling of elver migrations) and Glova & Duncan (in press), (modelling of changes of fish habitats with varying discharge).

Apart from bioassay experiments, experimental simulation and perturbation have not been widely used in New Zealand fisheries studies. However, 2 current fisheries programmes on the lower Waitaki River both involve experimental manipulation of flows in small (<0.5 $m^3\ s^{-1}$) and large (5-40 $m^3\ s^{-1}$) channels. An example of experimental perturbation is the recent simulated low flows in the Rakaia River brought about by the simultaneous shutdown of Lake Coleridge and Highbank Power Stations, resulting in a 39% reduction in flow. Backiel (1971) provides a review of the range of experimental work carried out on fish populations.

USE OF FISH IN BIOLOGICAL MONITORING

The value of monitoring aquatic organisms as environmental indicators has been reviewed by various authors (e.g., Burns, 1979; Williams, 1980). Invertebrates have been widely used as indicator species in biological monitoring (e.g., Winterbourn, 1981) although the use of fish provides several advantages. For instance, the taxonomy and ecology of fish has been widely studied, they are the top predators in most food chains and their abundance is generally an indication of well-being of lower trophic levels, and they have considerable recreational and commercial value. Disadvantages of monitoring fish include sampling difficulties, their mobility and hence avoidance of polluted waters, seasonal unavailability due to migrations, and their low density and long life histories relative to invertebrates.

There are various biological and social features of fish and fisheries which can be studied to assess the magnitude of proposed or actual impacts. These are given in outline form in Table 2 and discussed below as features related to fish or fisheries.

Fish community composition

The presence or absence of sensitive indicator species is widely used as a measure of well-being of aquatic environments. To be of use as a general indicator species, a fish should be widespread in geographic distribution, and non-specific in habitat preferences, but its physical tolerances should be well known. Among the New Zealand freshwater fish fauna (Table 3), brown trout (Salmo trutta) fit these criteria best. Unfortunately the physical tolerances of native fishes are not well known (Church *et al.*, 1979) and hence presence/absence data on these species is not used significantly.

TABLE 2: Features of fish populations and fisheries which can be studied as indicators of impact assessment.

A	FISH POPULATIONS		
	1 Community composition	-	presence/absence of indicator species
		-	species diversity
	2 Abundance	-	density, biomass
		-	catch per unit effort (sampling)
	3 Life history	-	feeding (diet, growth, condition)
		-	reproduction (fecundity, spawning success and recruitment)
		-	behaviour (habitat utilisation, induced movements, aberrant behaviour, mortality rate)
	4 Habitat availability	-	surveys of available/potential habitat
B	FISHERIES		
	1 Usage and catch	-	effort and use statistics
		-	catch per unit effort ('fishing', angling)
	2 Perceptions of users	-	response surveys

Although the presence of a species indicates that certain minimum conditions have been met, determining the significance of absence of a species is much more risky. Reasons for absence may be that there are unsuitable environmental conditions, that the species has not had the opportunity to colonise, or that another species has filled the niche (Cairns & Dickson, 1980). However, the absence of entire groups of species with similar environmental requirements is a much stronger indicator of exclusion.

Table 3 : Status, distribution, recreational (rec.) and commercial (comm.) importance, and habitat preference of 45 New Zealand freshwater fish.

Species	STATUS	DISTRIBUTION North Island	DISTRIBUTION South Island	ABUNDANCE	IMPORTANCE Rec.	IMPORTANCE Comm.	HABITAT PREFERENCES Estuary	Lowland lake	Swamp	Lower tributary	Mainstem	Middle tributary	Inland lake	Upper tributary
Kahawai	M	+++	+++	+++	+++	+++	+++							
Stargazer	M	+++	+++	++		+	+++							
Cockabully	M	+++	+++	+++			+++							
Stockell's smelt	N*		++	+++			+++							
Yelloweyed mullet	M	+++	+++	+++	++	++	+++			+				
Black flounder	N*	+++	+++	++	++	++	+++	+++		+	+			
Yellowbelly flounder	M	+++	+++	++	++	++	+++							
Giant bully	N*	+++	+++	+			+++							
Tench	I	++	+	++	+			+++		+		+		
Rudd	I	++	+	+++	+			+++						
Perch	I	++	++	+++	++			+++		++	+	+	+	
Dward inanga	N*	+		+				+++						
Goldfish	I	+++	++	++				+++	++	+		+	+	
Mosquitofish	I	++		++				++	++					
Catfish	I	++	+	++		+		+++	+++	++				
Black mudfish	N*	++		+				+	+++					
Brown mudfish	N*	++	++	+					+++	+				
Canterbury mudfish	N*		++	+					+++					
Sailfin molly	I	+		+					+++					
Common river galaxias	N*		+++	++						++	++			+++
Torrentfish	N*	+++	+++	++			M			+++	+++	++		
Bluegilled bully	N*	+++	+++	++			M			+++	+++	++		
Shortjawed kokopu	N*	+++	+++	+	+	+	M			+++	M	++		
Lamprey	N	+++	+++	+	+		M				M	++		+++
Koaro	N	+++	+++	++	++	++	M			++	++	+++	+++	++
Upland bully	N*	++	+++	++						++	++	+++	+++	+++
Cran's bully	N*	+++	+	++						++		++		++
Atlantic salmon	I		+	+	+								+++	
Mackinaw	I		+	+	+								+++	
Rainbow trout	I	+++	+++	++	+++						+	++	+++	+++
Quinnat salmon	I		++	++	+++	+++	M				M		+	+++
Brook char	I	++	++	+	+								+	+++
Alpine galaxias	N*		++	+										+++
Dwarf galaxias	N*	++	++	+								+++		+++
Longjawed galaxias	N*		+	+										+++
Inanga	N	+++	+++	+++	+++	+++	++	++	++	+++	+++			
Giant kokopu	N*	++	+++	+	+	+	+	++	+++					
Grey mullet	M	+++	++	++	+	++	+++	++			++			
Redfinned bully	N*	+++	+++	++			M			+++	++	+++		
Common smelt	N*	+++	+++	++	+	+	+++	++		+	++		+	
Shortfinned eel	N	+++	+++	+++	+	+++	++	+++	+++	++	+			
Common bully	N*	+++	+++	+++			M	+++	+++	++	++	++	+++	
Brown trout	I	+++	+++	+++	+++		+	++		+++	+++	++	+++	++
Banded kokopu	N*	+++	+++	+	+	+	M	+		+++	M	+++		
Longfinned eel	N*	+++	+++	+++	+	+++	+	+		+++	+++	+++	+++	++

Key:

Status	M	= marine straggler	Importance	+++ high
	N	= native species		++ moderate
	N*	= native species, found only in New Zealand		+ low
	I	= introduced species		
Distribution	+++	widespread populations	Habitat	+++ frequently occurs
	++	regional		++ often occurs
	+	localised		+ occasionally occurs
Abundance	+++	abundant		M migrates through
	++	common		
	+	uncommon/rare		

Although the composition of plant and invertebrate communities is widely used for calculating biotic indices, the composition of fish communities is not. At present the only index of community composition used in New Zealand fisheries studies is a species list. However, as the distribution and habitat requirements of species become better understood, there is potential for developing various biotic and diversity indices which could be of use for making comparisons between modified and non-modified catchments. (A useful review of fish community data is given by Hendricks et al., 1980). Table 3 shows the generalised composition of fish communities which may be found within different freshwater habitats in New Zealand.

Fish abundance

Various measures of fish abundance have been used to indicate environmental change. Graynoth (1979) and Glova et al. (in press) have used both density and biomass estimates to compare the effects of logging practices and floods on fish populations. Sampling catch/effort data are also useful measures of relative abundance, and units of expression can be devised to suit circumstances, e.g., number of fish per 100 metres of net per hour, and number of fish per volume of water filtered.

Fish life history data

A variety of parameters which affect the life history of a particular fish species has been studied as indicators of impact assessment. The composition and availability of food affects all aspects of fish life-histories - a lack of food is normally expressed as changes in growth rate and condition (e.g., Burnet, 1968) which in turn may result in failure to spawn, induced movements in search of food, increased susceptibility to disease and so on. Changes in water quality will bring about changes in invertebrate composition (e.g., Hirsch, 1958); although invertebrate biomass may not change significantly, a change in composition may mean that species which are relatively available to fish (e.g., mayflies) are replaced by less available species (e.g., chironomids).

The susceptibility of fish to changes in the environment varies with different life-history stages. Thus salmonid eggs and yoked fry living within the interstices of gravel are very susceptible to smothering by silt or scouring by floods, while larger fish are able to avoid silted areas and seek out areas of reduced flows during floods.

Habitat availability

Some impacts may result in complete habitat loss, as in the drainage of a wetland. As many freshwater fish exhibit marked habitat preferences (Table 3), loss of habitat means a corresponding marked reduction or even complete loss of certain species. Alternatively, habitat may be reduced in area or degraded in quality. Changes in water quality and benthos are frequently used to assess habitat changes, while a quantitative assessment of habitat is obtained by use of the "incremental method" (Co-operative Instream Flow Group, 1982; Glova, 1982). This latter technique allows quantitative determination of changes in potential fish habitats with alteration in stream flow. However, the method cannot be used to predict effects on fishery production as relationships between physical habitat, fish behaviour, and fish production are not yet well understood and are likely to be variable and site specific.

Fisheries

A downgrading of fishery values in commercial and recreational fisheries is almost invariably reflected in either catch-effort statistics, overall use, or the users perceptions. For instance, Richmond (1981) examined catch rates of trout when evaluating the effect of hydro-electric development on the Tongariro River fishery. Also, anglers have expressed concern at the detrimental effect of flow fluctuations on the Tekapo River fishery (Teirney et al., 1982).

Difficulties of data collection and interpretation

There are many problems associated with collection and interpretation of data on freshwater fish. For instance, sampling selectivity of equipment, trap avoidance by fish, and unequal spatial and temporal distributions of fish (the problem of randomness for survey design) pose difficulties for quantitative assessments. Biological problems include inadequate life history data, natural variability in population structure and size, and compensatory mechanisms exhibited when fish populations are subject to stress.

Compensatory mechanisms are widespread in fish, but not well understood. For instance, a fish population subjected to increased mortality (harvesting, predation, poisoning, etc.) might respond by decreasing the age at maturity, increasing fecundity, or increasing the rate of growth, which may in turn affect the onset of reproduction and vulnerability to predation, etc. In reviewing compensation in fish populations, Goodyear (1980) concludes that these processes are "often difficult or impossible to measure directly" especially as environmental impacts usually affect whole communities rather than just individual species.

The high natural variability inherent in many physical and biological phenomena is the major constraint in quantitative assessment (Beanlands & Duinker, 1983). In assessing the impact of an environmental change it must be possible to differentiate between change attributable to the impact and the dynamics of natural variation. In an extensive review of natural variation in abundance of salmonids, Hall & Knight (1981) concluded that temporal and spatial variation may be several orders of magnitude in extent, and may be sufficient to mask very significant perturbations caused by such practices as logging and agriculture. Several New Zealand authors have recorded large temporal variations in salmonid abundance. For example, Burnet (1959) recorded a 7-fold variation in total populations of brown trout in the Dolyeston Drain, Canterbury, during 5 years of observation, spawning brown trout in the Glenariffe Stream showed a similar degree of variation between 1965-1982 (Davis et al., 1983), and numbers of quinnat salmon trapped in the same stream showed an 8-fold variation between 1965-1978 (Flain, 1982).

The following steps are advocated as a general base for impact assessment studies. Outlines of the 3 types of study are given in Table 4.

Problem definition

At the onset of the study it is most important that an appreciation is gained of the main biological components - for instance, what are the fisheries values of significance, what are the main components of habitat, what are the main pathways involved? Compilation of a matrix (e.g., Church et al., 1979) may assist in identifying these components. A conceptual model should then be developed which should serve to identify the main structural and functional relationships of the various components. These components should then be considered as to their suitability for study by asking whether they are susceptible to change by the proposed impact (directly or indirectly) and whether it is realistic to study them. A "hunch" about a potential impact can be refined and stated either as a specific question for which a numerical answer is possible, or as a testable hypothesis (Beanlands & Duinker, 1983). A helpful question to ask when focusing on major components for study is "what would I do with the information if I had it?" A literature review should also be carried out to see how other workers have approached related problems.

TABLE 4 : Proposed sequence of events in impact assessment studies.

Pre-impact Study	Post-impact Study	Pre/post Impact Study
Problem definition	IMPACT OCCURS	Problem definition
↓	↓	↓
Define study objectives	Problem definition	Define study objectives
↓	↓	↓
Design study	Define study objectives	Design study
↓	↓	↓
Baseline study	Design study	Baseline study
↓	↓	↓
Impact prediction	Baseline study	Impact prediction
↓	↓	↓
Report	Impact evaluation	IMPACT OCCURS
↓	↓	↓
IMPACT OCCURS	Report	Monitoring
		↓
		Report

Define the study objective(s)

"Be able to state concisely to someone else what question you are asking" (Green, 1979). A testable hypothesis should then be formulated. Although the hypothesis may be left as a general statement it is better refined into a question or series of questions which require specific and preferably quantified answers. This type of question can normally be expressed as a null hypothesis which can be tested statistically. For example, the fisheries values of a section of river to be subjected to abstraction of water for irrigation, may be as a pathway for adult quinnat salmon moving upstream to spawn. A general hypothesis could be that the passage of salmon might be impeded by the abstration. This statement could be further developed to 'will the abstraction of $\underline{x}$ cumecs of water result in a minimum depth of less than 32 cm (the maximum body depth of salmon)?' The null hypothesis is then that an abstraction of $\underline{x}$ cumecs will not result in water depths of less than 32 cm.

Study design

If quantitative results are desirable then statistical methods must be considered at all phases of the study design, e.g., gear selection, sampling frequency, analysis of results. Critical features of design include ensuring that any random sampling required is truly random and provides the ability to distinguish between variance within samples and between samples. Some preliminary sampling to check on the efficiency of sampling equipment, the number of samples required, and the possibility of spatial differences in the distribution of the required organism, is extremely valuable at this point.

Given the existence of potentially large natural variations in fish abundance, much care must be taken in both the choice of variables to be measured and the experimental design. Again the use of controls is stressed for before and after type impact studies. "An effect can only be demonstrated by comparison with a control" (Green, 1979). In pre-impact studies, controls are appropriate if they form the basis of a post-impact comparison. For pre-impact studies, emphasis must be on establishing an adequate data base from which to predict the outcome of potential impacts. In practice, time for aquatic pre-impact studies is often short and the fact that many biological events occur on a seasonal basis compounds the problem. For obtaining quantitative data on fish populations it is suggested that 3 years data are the minimum required for meaningful predictions.

Time spent in refining study design is invariably time well spent. Upon starting a programme the dual temptations to immediately commence field studies and to measure anything and everything, must both be resisted.

Baseline study

For pre-impact and pre-post impact studies the objective is to establish a data base against which future changes in time and space can be predicted or measured. For post-impact studies the baseline study involves comparison of spatial distribution data and inferring the degree of change caused by the impact. (This would normally involve comparison of samples between the affected area and a non-affected area, but comparison of samples from within an affected area is sometimes used to infer change). Parameters to be studied will have been defined at the problem definition phase. Field studies may lead to some revision of the hypotheses. At the completion of the study, results are analysed and interpreted.

Impact prediction

Following analysis of baseline study data, a prediction is made on the likely effects of the proposed impact. If the study is a pre-impact one then the report phase follows.

Monitoring

After the impact has occurred further studies are done to determine the degree to which predicted impacts have occurred (hypothesis testing). In a pre-impact study this phase is not possible unless some pilot scale experimental perturbation is carried out, i.e., it may be possible to experimentally simulate the impact on a small scale or over a short time.

Report

The report must be more than a mere presentation of results. It must contain interpretation and any recommendation to reduce or eliminate undesirable effects. In a pre-impact study the implications of alternative strategies should be clearly spelled out and, if possible, a specific recommendation made. As with all scientific writing, results should be presented in such a way that they can be repeated by someone else and the results verified. Obviously, the effective communication of the results and recommendations are essential if environmental analysis is to influence decision making. As a rule-of-thumb, at least as much effort should go into communication as goes into analysis.

IMPACT ASSESSMENT : NEW ZEALAND EXAMPLES

Impact assessment in the New Zealand freshwater environment is in its infancy. Historically, there have been many post-impact statements and descriptions but few deliberate assessments which have had significant predictive value.

Although impact assessments in New Zealand can be classified according to the types of study (pre-/post-impact), they can also be classified by the type of modification involved, i.e., whether modifications are accidental or planned. Accidental modifications may, in turn, be natural (e.g., volcanic eruptions), or man-induced (e.g., toxic leaks, accidental release of exotic fishes, etc.). By definition, assessment studies of accidents are post-impact studies unless, fortuitously, a baseline study of the area already exists. Planned modifications are the more common type and include impacts of hydro-electric power generation, water abstraction, forestry, eutrophication, introduction of new fish species, etc.

Outlines of studies for 2 impact assessments are presented. Both are pre-impact studies and they serve to illustrate the diversity and type of field studies which can be involved in comprehensive and long-term programmes.

An assessment of the use of Chinese Grass Carp (*Ctenopharyngodon idella*) as a weed control mechanism

The Freshwater Fisheries Advisory Council, a council comprising mainly government and acclimatisation society personnel, has established a series of guidelines to control the selection and introduction of new species of fish into New Zealand. These guidelines include a suitable quarantine period; observations on behaviour, growth, food, parasites and diseases; and the release of disease-free fish into an isolated lake for studies of interactions with native and introduced faunas.

Accordingly study phases in the evaluation of grass carp have been as follows :

a initial introduction and disease and parasite control;
b feeding of young grass carp both in the laboratory and in outdoor ponds;
c artificial maturation and spawning techniques;
d outdoor trials with control areas in an agricultural drain, a small reservoir containing native fish, a larger drain system, and an isolated lake (the latter study includes impacts on both native and introduced fishes);
e likely impacts of grass carp on wildfowl populations.

An overall impact assessment and review of previous research and publication is contained in Rowe & Schipper (1985).

Assessment of reduced flows on fish habitats in the Rakaia River

FRD has been involved in a major studies programme on the Rakaia River for 5 years. The approach has been to carry out a series of integrated studies, some of which have been baseline studies while others have been evaluations of potential methods for studying the effects of reduced flows on instream values. Studies have been :

a fish inventory and life history studies (distributions, migrations, densities), e.g., Davis et al. (1983);

b habitat quantification (computer modelling using the instream flow incremental methodology), e.g., Glova & Duncan (in press);

c habitat quality (invertebrate densities and rates of colonisation, dynamics of invertebrate drift), e.g., Sagar (1983);

d fishery values (recreational and commercial fisheries, expert angler evaluation of flow requirements), e.g., Unwin & Davis (1983).

In addition to the above, complementary studies by the Ministry of Works and Development have looked at minimum depth in relation to the passage of fish and jet boats (Mosley, 1982), temperature modelling (Jowett & Mosley, 1983) and catchment hydrology. At present, results of these studies have been presented at a National Conservation Order hearing for the Rakaia River.

DECISION MAKING

The most important phase of impact assessment is decision making, i.e., "So what? How significant is the impact?" These types of decisions occur at 2 levels - by the biologist interpreting the data, and by the resource manager assessing the report.

In presenting an interpretation of results, it is essential that the biologist presents the facts clearly and that conclusions are both clear and logical. The scrutiny of colleagues is strongly recommended. "Good science can be defined as that which is acceptable to the scientific community as determined by peer review" (Beanlands & Duinker, 1983). There is no room for personal bias and the temptation to draw conclusions beyond those supported by the data must be resisted. Maintenance of scientific integrity is essential.

A common difficulty for biologists is the transition from objective conclusions to subjective judgements. For instance, because a relationship is statistically significant, it does not automatically follow that it is biologically meaningful. Statistics are an invaluable aid to decision making - they do not make decisions but define the likelihood of an event taking place. As a consequence of the inherent uncertainty of biological prediction it must be recognised by resource managers and the like that it is unrealistic to expect biological predictions to conform to the same degree of precision as engineering calculations. This is especially so for animals like fish which are high in the trophic levels and whose behaviour is consequently influenced by the well-being of animals (and plants) at lower levels.

In discussing how to evaluate whether impacts are important, Erickson (1976) considered that impact assessments frequently provide quantitative data on environmental parameters but do not address the fundamental question of whether values are gained or lost. Ultimately it is values that people relate to. Will this development downgrade the wilderness experience afforded by this river? Will this impact create a significant new fishery?

Normally impact assessments are reviewed by other agencies - for instance the board or a tribunal of a regional water board, the Commission for the Environment. Depending on the extent and sensitivity of the impact, a variety of other impact assessments, engineering options, public submissions and so on may also have to be considered by the agency concerned. Again subjective judgement is called for, with decision makers frequently having to weigh up and balance contrary viewpoints and mutually exclusive options. Ultimately, decisions may be based as much on subjective judgement involving values, feelings, and beliefs, as on the results of scientific studies. Again the overall assessment of whether values are gained or lost is relevant. Erikson (1976) considers that 2 very important criteria for evaluating and deciding between competing values are diversity and uniqueness of values - diversity recognises the desirability of maintaining many kinds of human values and environmental resources, while uniqueness recognises the relatively greater worth of resources and values that are uncommon within any particular area. Beanlands & Duinker (1983) state that impact assessment is concerned with the perceptions and values of society which may change with time. They suggest that significant impacts are those which result in the irretrievable loss of ecosystem components (e.g., gene pools) or functions (e.g., primary production) although the ultimate concern can almost always be traced to human values.

FUTURE DIRECTIONS

With increasing interest in the further development of New Zealand freshwater resources for 'industrial' usage, impact assessment studies are essential to present the environmental consequences of developments. Although water is a renewable resource, fish habitats are seldom renewable. Before decisions are made which would result in the downgrading or loss of fisheries habitats, the potential impacts of development need to be studied and the results presented to resource managers in a clear and objective fashion. To enable wise decisions to be made several matters need to be considered by both fishery biologists and resource managers.

Managers

a Presently there is a lack of a national policy for management of freshwater resources. As a result it is possible that a unique aquatic environment, a national asset, could be lost to benefit a purely local interest.

b Frequently inadequate time is available for appropriate biological studies. Fisheries studies usually need to take into account seasonal features. Also, as natural variations in population abundance are large, several years data may be required to provide an adequate data base from which change can be measured. The problem of time is compounded by changing priorities of developers with the consequence that forward planning for fisheries researchers and managers is extremely difficult.

c Given the complexity of biological systems and the high degree of uncertainty involved in making definitive predictions, managers will frequently have to accept the subjective judgement of an experienced and credible fisheries biologist.

Biologists

a Biological inventories are not impact assessments. Assesssments must include clear statements of the implication of a proposed development, as far as these can be ascertained.

b Increased emphasis must be placed on initial conceptual thinking and design of study programmes. Desk-top studies, literature surveys and review by colleagues are essential components in ensuring the right questions are being asked.

c Greater emphasis should also be placed on quantitative simulation modelling. Mathematical modelling can indentify cause and effect relationships that are not visually obvious while computers allow large quantities of data to be stored and accessed.

d Inter-disciplinary studies should be encouraged. Although freshwater fish species and communities are important as aquatic indicators, their abundance and well-being depend upon a series of physical, chemical and biological components. Ideally, fisheries studies should routinely involve hydrologists, water quality personnel, invertebrate biologists, and statisticians. Aquatic plant biologists, economists and other specialists should also be involved when appropriate.

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METHODS FOR ESTIMATING FRESHWATER FISHERIES RESOURCES

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ABSTRACT

This paper reviews the techniques available for estimating freshwater fisheries resources, particularly those commonly used in New Zealand. Electric fishing, drift diving, netting, and trapping are the most widely used techniques, but aerial observation, echo sounding, poisoning, and explosives have also been used. Anglers may be surveyed to derive fish density from catch rates and patterns of angling.

Computer data bases can provide useful information about fish distribution, and Fisheries Research Division, MAF, holds a comprehensive data base of freshwater fish distribution that should be consulted before any survey is carried out. Information from any surveys in New Zealand which collect additional fish distribution data may be entered into the data base by putting information onto appropriate forms available from Fisheries Research Division.

INTRODUCTION

The information necessary to manage freshwater fisheries includes fish abundance, identity, population dynamics, habitat, diet, and exploitation rates. Gathering this information requires the use of 1 or more of the following techniques: fish capture, fish observation, angler surveys, and a search of existing data bases. This paper briefly describes each of these methods and comments on their use in New Zealand. Further descriptions of these techniques are available in Bagenal (1978) and Nielson & Johnson (1983).

FISH CAPTURE AND OBSERVATION TECHNIQUES

Estimates of total fish abundance are made by catching or observing either all fish in a given area, or, more usually, a proportion of the population.

Biological monitoring in freshwaters: proceedings of a seminar. Eds. R.D. Pridmore and A.B. Cooper. Water & Soil Directorate, Ministry of Works and Development for the National Water & Soil Conservation Authority, Wellington, 1985. Water & Soil Miscellaneous Publication 83.

To estimate total abundance from a proportional catch, fish are caught, marked, and released. The relative rates of recapture of marked fish are used to estimate the total population (Youngs & Robson, 1978). Alternatively, total abundance may be estimated by the "removal method", in which diminishing returns from a systematically applied capture technique are used to estimate fish abundance at the start of sampling (DeLury, 1951; Zippin, 1956, 1958; Youngs & Robson, 1978). Capture or observation techniques can also be used to determine relative abundance of species and seasonal variations of single species. However, if the sampling method is not likely to equally observe or catch all fish of interest, or is not carried out seasonally, some species or size classes will be under-represented or appear absent.

Passive fish capture techniques

Passive capture techniques include nets, traps and pots. These methods have the advantage that equipment is generally simple and cheap, need only a boat or divers for setting and retrieving, and can also be used in deep, inaccessible water. However, passive capture may be selective for size, species, and sometimes sex of fish. Relative abundance can be determined from passive capture, assuming that catch per unit effort (CPUE) is proportional to fish density. However, season, water temperature, turbidity, and currents can influence CPUE.

Nets can be used to catch most fish species, and the type of net and mesh size to be used is dependent on the target species and its habitat. Gill and trammel nets (Fig. 1) may be used in lakes or rivers, but in rivers they are best set parallel to the current to avoid catching debris, which limits the nets' effectiveness. Hoop nets and fyke nets (Fig. 1) function like traps, and are less likely to injure fish than gill or trammel nets. Fyke nets usually have 1 or 2 wings to intercept fish movement, whereas hoop nets do not. Hoop and fyke nets may be baited or unbaited.

Solid-frame traps and pots, such as the traditional eel pots or hinakis, and minnow traps (Fig. 1) may also be used in rivers and lakes. Salmonid eggs or salmon food are a common bait for small fish, and experience in New Zealand with Gee minnow traps showed that they were very successful in all habitats except fast braided rivers (Davis _et al_., 1983; pers. comm., G.A. Eldon, Fisheries Research Division, MAF.).

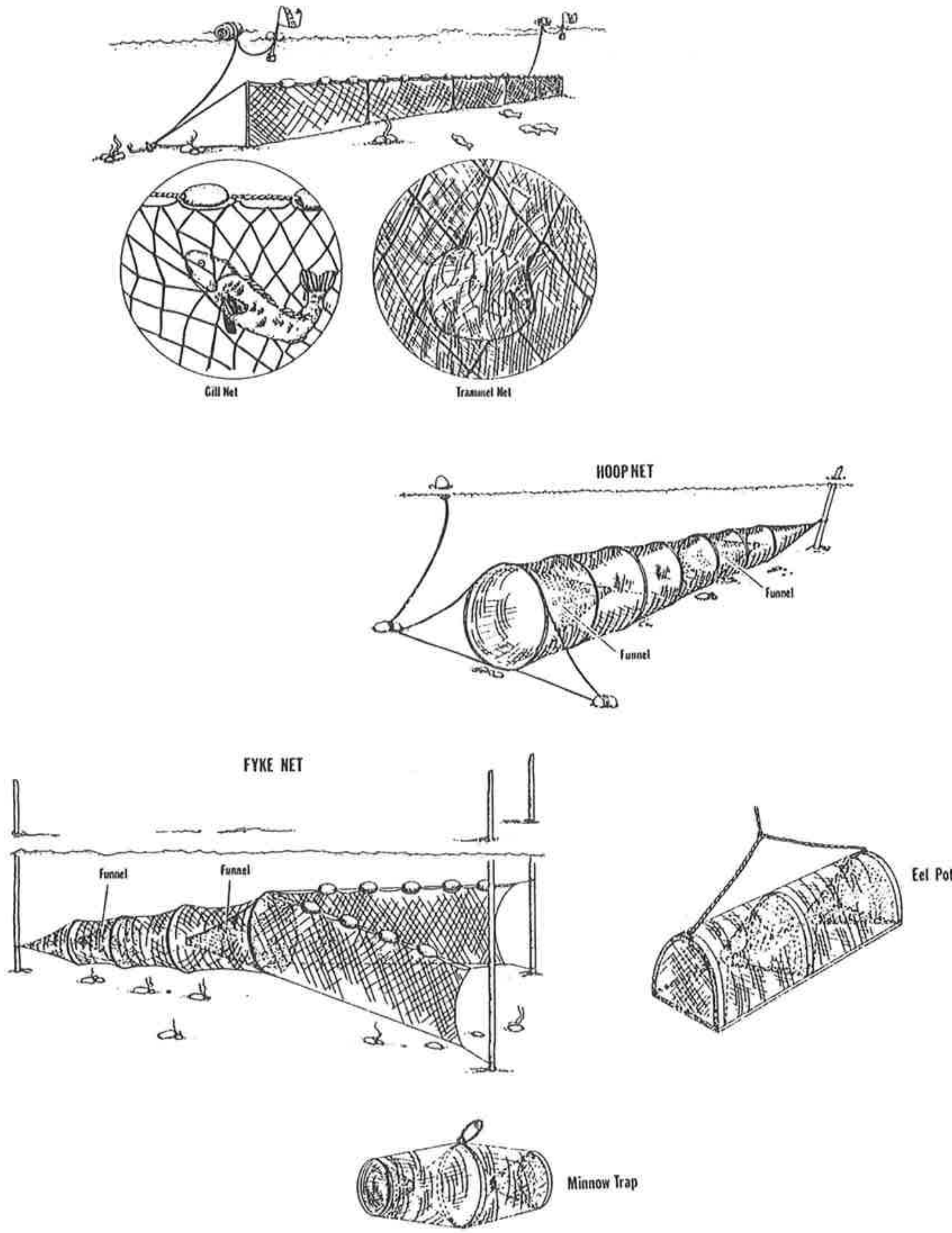

Fig. 1: Passive capture techniques - nets and traps (from Sundstrom, 1957; Dumont & Sundstrom, 1961; Nielsen & Johnson, 1983).

Migrating fish can be trapped readily in small to medium-sized rivers using a weir and trap arrangement. This was the Maori method of catching adult lampreys as they migrated upstream to spawn in spring, and is currently used for catching migrating salmonids. Migrating smelt and eels can also be caught using a weir and trap of an appropriate design.

Passive sampling methods can be used in studies which estimate growth rates, reproductive cycles, migratory patterns, distribution, and diurnal activity, as the fish are usually collected alive without damage. The exception may be gill or trammel nets, which can damage fish and lead to a high mortality.

Active fish capture techniques

Active capture techniques (i.e., those in which the sampler is used to actively seek out fish) include trawl nets, seine nets and push nets. Trawl nets (Fig. 2) are most appropriate for lakes and estuaries as they can be used to sample fish from any depth. The species caught will depend on the mesh size and depth of the net. Seine nets (Fig. 3) pulled by a boat or by people wading are useful in lakes, but are less useful in most rivers because of the current and snags. A gill net can be successfully drifted downstream by divers and then hauled ashore like a seine net in some rivers. Push nets are useful for sampling small fish in shallow water at lake and river margins.

Electric fishing

Electric fishing is now a highly sophisticated sampling method, and electric fishing machines have been designed especially for New Zealand conditions (Burnet, 1959, 1967). Electric fishing is generally limited by requirements of safety and efficiency to water 1 m or less in depth, but gear can be mounted on a boat for use in slow-flowing rivers. All electric fishing has an element of risk, especially when it is done from a boat, and it is covered by special legislation which requires that operators are specially trained (Part VII, Freshwater Fisheries Regulations, Fisheries Act 1983).

Two electric fishing systems are commonly used. The most powerful system, called a "mains set", consists of a 240 volt AC generator and a pulse unit (which stay on the bank) and a cable reel. The other system, a "backpack

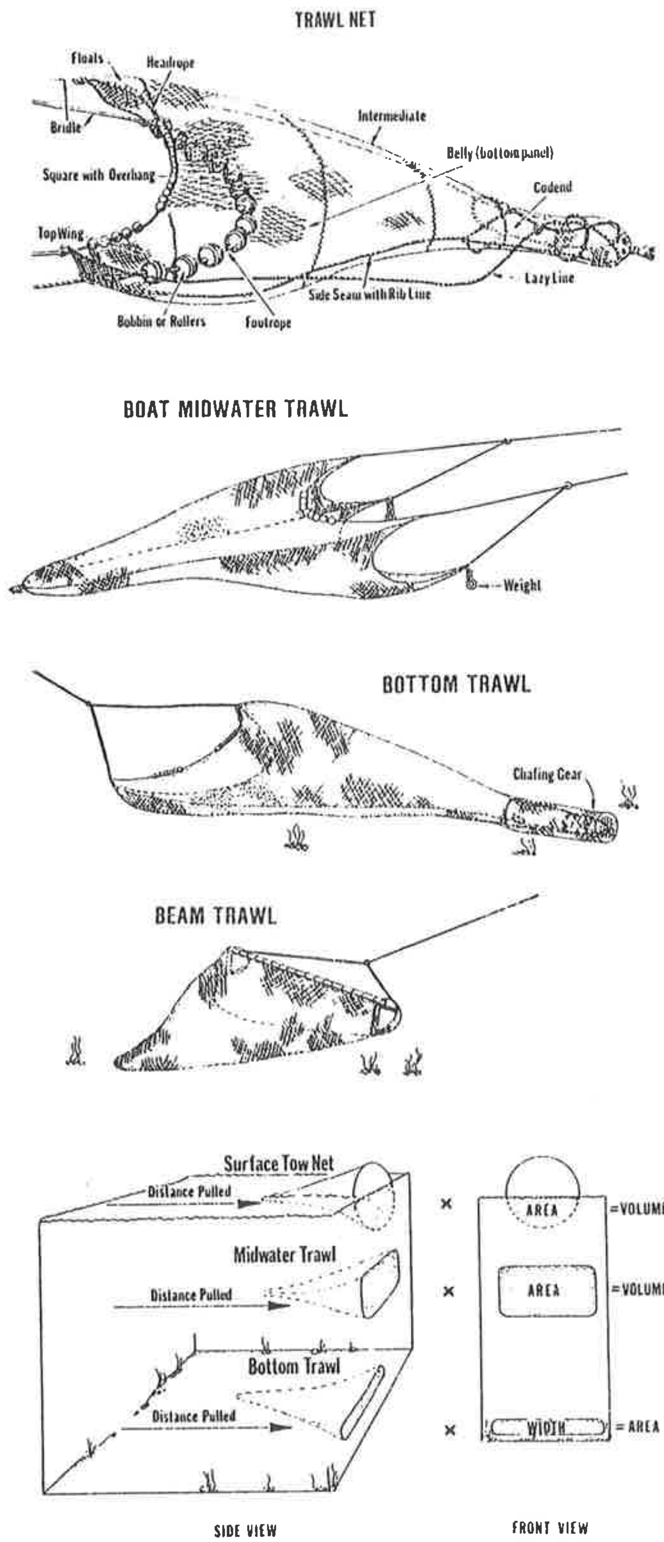

Fig. 2: Active capture techniques - trawl nets and their geometric sampling volume (from Nielsen & Johnson, 1983).

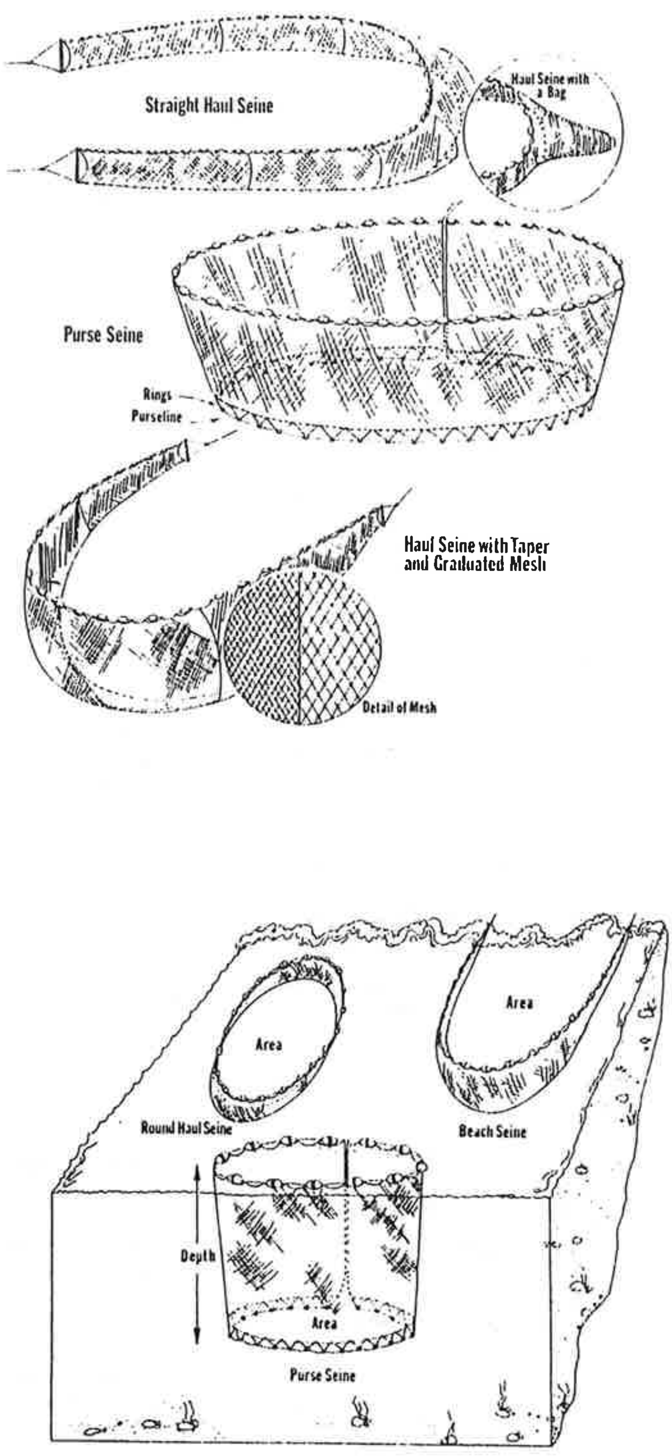

Fig. 3: Active capture techniques - seine nets and their geometric sampling volume (from Nielsen & Johnson, 1983).

set", consists of a lead acid battery and a small pulse unit carried on the operator's back. In both cases the fishing field is generated by a hand-held insulated anode, and the current normally returns along the stream bed and via an earth strap cathode to the machine. Electric fishing is very appropriate for catching cryptic fish, such as bullies, juvenile trout, galaxiids, and torrent fish, and is useful for mark-recapture methods because it rarely causes lasting damage to the fish.

Poisons

Poisons can be used as effective sampling tools in lakes. The method has the limitation that it usually kills the fish. Rotenone, antimycin, chlorine, and creosote are typically used as fish poisons. In a successful exercise in Lake Parkinson, near Hamilton, rotenone was used to eliminate rudd and tench, and to capture grass carp in the process. Rotenone is particularly useful because instead of killing fish outright it blocks the oxygen uptake mechanism in the gills, causing the fish to swim to the surface in an attempt to find more oxygen. If they are removed from the rotenone quickly enough and transferred to clean water, their survival chance is high.

A disadvantage of poison as a sampling technique, apart from its destructiveness, is that the toxicity of a poison to a range of species is variable, and will change with temperature, water quality, concentration, and exposure time. Species such as catfish may need as much as 7 days continuous exposure to be affected. In addition it requires a water right, and can cause public outcry because of its destructive nature.

Explosives

Fish may be sampled using gelignite or explosive fuse cord which stuns or kills. Platts (1974) used 6.4 km of explosive fuse cord to sample fish in a 4.4 km stream reach. This sampling method is not, however, straightforward. Stunned or dead fish do not necessarily float, and therefore divers may have to inspect the sampling site underwater to recover all fish. This technique was used on the Rakaia River recently, but it was not very effective and caused adverse reaction from anglers. The most appropriate application for explosives is for rapid sampling of fish in small streams, in conjunction with nets. Explosives have rarely been used to sample fish populations in New Zealand, and their use requires specialist training and sometimes permission from local authorities.

Fish observation techniques

Observation methods can be indirect, such as echo sounding, or direct, such as observation from river banks, platforms, the air, or underwater. To be effective, all techniques except echo sounding require clear water, and should only be used for larger fish and non-cryptic species. Observation from banks and platforms can be a useful technique for studying fish behaviour (Shirvell & Dungey, 1983) but is less useful for population estimates. Aerial observation can be useful for counting adult salmonids and their redds but is expensive and is not generally as reliable as underwater observation.

Underwater observation by drift diving is simple and cheap, and is being used increasingly in New Zealand rivers. A list of suggested equipment is given in Helfman (1983). The technique requires trained divers wearing snorkel equipment, and can be used to assess relative abundance, size classes, distribution, habitat preferences, and behaviour of salmonids (Northcote & Wilkie, 1963; Griffith, 1981; Hicks & Watson, in press). Eels and bullies may also be seen, but, because of their cryptic habits, those seen will be a small proportion of the total population. For optimum counting efficiency, water clarity should exceed water depth, and in no instance should underwater visibility be less than 2 m. A convenient way to assess visibility is to measure the distance at which a Secchi disc viewed horizontally from underwater just disappears from view. The minimum number of divers required can be calculated as the river width divided by twice the underwater visibility, e.g., 3 divers for an 18 m wide river with visibility of 3 m.

The basic method for systematic trout counting is to choose representative sections of river about 1 to 2 km long (which would take 1-2 h to drift dive), and to map them, marking the pools and runs at easily identifiable points, such as rapids and riffles, both on the map and at the stream edge. At the start of a drift dive the divers spread themselves equally across the river in a straight line, and float downstream with the current. Divers scan through the area in front of them, identifying and counting all the fish they see. At the end of each pool or run the team stops and the counts are recorded; the team then regroups and begins the procedure again. Divers need to avoid touching the bottom to prevent dislodging sediment and algae, both of which reduce visibility and, therefore, reduce counting efficiency. Behavioural differences between different species and sizes of fish, shadows cast by bright sun, and the nature of the substratum also affect counting efficiency. Cold limits the time divers can spend in the water, and though most divers in 7 mm wetsuits can cope with up to 1 h in water above 5 or 6°C, below this temperature, drysuits are necessary. The length of the station to be drift dived should take the temperature into account. A rough guide for travel speed, including counting stops, is 1 km per hour. During drift dives of over 1 to 2 hours, divers tend to lose concentration and accuracy.

Even under optimum conditions it is difficult to estimate the proportion of the fish present that have been observed. Thus drift diving usually only gives relative abundance, i.e., of fish seen in 1 season compared to another, or of 1 size class or species compared to another. Techniques based on multiple dives have been proposed which may overcome this to some extent, e.g., the bounded counts method (Hicks & Watson, in press), but variations in conditions in each river, and possibly even at each site, will lead to differences in counting efficiency. The bounded counts method assumes that no fish are seen twice, and that all fish are equally likely to be seen. The abundance is given as twice the maximum number of fish seen minus the next highest count from a series of 2 or more dives.

In lakes, SCUBA can be used, with divers being towed by a boat to survey large areas using either a foot stirrup or a manta board. Transect dives may also be useful.

ANGLER SURVEYS

Surveys of anglers yield information about the catch, fishing habits, and anglers. Interviews at the river bank, lake shore, in boats on a lake, or at a launching ramp are the most reliable method, but are also time-consuming. Mail and telephone surveys can be used, and though these may be statistically more valid, because a larger number of anglers can be randomly sampled more easily than with on-site surveys, the response rate can be poor. The first survey may need to be followed by a further effort to elicit responses or to find the reason for non-response.

With a measure of the catch and the effort expended by the angler the CPUE (e.g., fish per angler-hour) can be calculated and used as an index of stock density. However, surveys need careful statistical design for this information to be reliable. Angler surveys must also have clearly formulated objectives in terms of their target population of anglers, the kind of information to be collected, and the temporal and spatial limits of the survey.

DATA BASES

Two data bases are held by Fisheries Research Division, MAF, Wellington, and the information contained in them is available to any interested person. The first is a freshwater fish data base. It provides a method of storing and retrieving information about New Zealand freshwater fish distribution and habitat (McDowall & Richardson, 1983). Reliable records extend back to the sixties, and electric fishing has led to a massive growth in the stored information. The data base now has over 6000 locality records for both native and introduced fish. Books of data cards are available to anyone regularly engaged in stream surveys and should be used by everyone who carries out electric fishing. Date, locality and species found are the minimum information required, but there is provision on the cards to record measurements of pH, dissolved oxygen, temperature, substratum, bottom fauna, and other relevant information. Provision for a carbon copy of each card gives a useful record of sampling for those who complete cards. Retrieval usually follows a written request to Fisheries Research Division, Box 297, Wellington, stating the stream name, catchment number (Anon., 1956), species, and reason for request. The data can be printed as a summary sheet, or can be plotted onto small-scale maps with up to 3 species together. Processing of requests rarely takes more than 1 week.

Data on registered freshwater commercial fishermen and their catches is also held by Fisheries Research Division. Furnishing a fishing return to the Fisheries Statistics Unit of Fisheries Research Division is a condition of licencing for fishermen, and 221 fishermen currently report their catches of eels and catfish. The data base is divided into acclimatisation society districts and is updated weekly. The catch and number of pots or nets used can be retrieved for the whole country, for individual districts, or for individual fishermen. Data cannot be retrieved on a catchment basis.

CONCLUSIONS

In attempting to cover the broad and complex field of fisheries resource assessment, techniques have inevitably been skimmed over or omitted. I have, however, tried to concentrate on those methods that are most useful in New Zealand at the moment. These are electric fishing, drift diving, and netting of various types. A summary list of the methods and the species and environments for which they are appropriate is given in the Appendix.

No single textbook describes all fisheries methods in enough detail to be sufficient by itself, although Nielsen & Johnson (1983) is a good starting point. Anyone who wants to estimate fisheries resources should compare several textbook descriptions and look at some papers describing application of the technique. Finally, there is no substitute for field trial of a method to assess its applicability for sampling fish in a given situation.

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APPENDIX : Freshwater fisheries resource assessment techniques, their target species, and the appropriate environments for their use.

Technique	Species												Environment			
													Lake		River	
	Adult salmonids	Juvenile salmonids	Perch	Catfish	Eels	Adult galaxiids	Juvenile galaxiids	Adult lamprey	Juvenile lamprey	Smelt	Torrentfish	Bullies	Deep	Shallow	Deep	Shallow
Gill net	+	+	+										+		+	
Trammel net	+	+	+										+		+	
Hoop net	+	+	+	+	+	+							+	+	+	+
Fyke net	+	+	+	+	+	+							+	+	+	+
Trap net	+		+										+			
Pots, traps and hinakis	+		+	+	+	+						+	+	+	+	+
Weir and trap	+	+			+		+	+								+
Trawl net	+	+	+							+		+	+	+		
Seine net	+	+	+							+		+	+	+	+	
Push net		+	+							+		+		+		+
Electric fishing		+	+	+	+	+			+	+	+	+		+		+
Poisons	+	+	+	+	+		+			+		+	+	+		
Explosives	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Echo sounding	+	+	+							+		+	+			
Bank observation	+															+
Aerial observation	+													+		+
Drift diving	+	+													+	+
Towed diver	+	+								+		+	+			
Angler survey	+		+										+	+	+	+

BIOASSAYS USING FRESHWATER FISH FOR WATER QUALITY MONITORING AND ASSESSMENT

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ABSTRACT

A questionnaire on the usage of fish bioassays in New Zealand revealed a relatively low level of application of these techniques. Most bioassays have been conducted by regional water boards and central government agencies. The results have generally been confined to unpublished reports, although a variety of papers have been published.

Bioassays can be used to establish water quality criteria, monitor habitat quality, monitor effluent toxicity, screen potentially toxic samples, and to provide legal proof of toxicity. The measurable responses used in fish bioassays include indicators of performance, biochemistry, physiology, respiration, behaviour, and mortality.

Standard methods exist for acute and chronic toxicity tests, but behavioural, performance and clinical bioassay techniques are still at a stage of development where standardisation appears impractical. Remote sensor and automated bioassay alarm systems have reached a level of commercial viability overseas.

However, the development of bioassay techniques and skills in New Zealand appears to be limited by a lack of knowledge and a lack of commitment to provide the resources necessary. Recent and impending changes to the Fisheries and to the Water and Soil Conservation legislation may catalyse an increased use of fish bioassays for toxicity assessment.

INTRODUCTION

The primary intention of this paper is to provide some basic information on the existing and potential uses of fish bioassay techniques, together with comment on their applications and interpretations. It does not provide

Biological monitoring in freshwaters: proceedings of a seminar. Eds. R.D. Pridmore and A.B. Cooper. Water & Soil Directorate, Ministry of Works and Development for the Natonal Water & Soil Conservation Authority, Wellington, 1985. Water & Soil Miscellaneous Publication 83.

detailed methods on fish bioassays, although the sources of such information and guidance will be discussed.

In preparing this paper, I circulated a questionnaire on fish bioassay usage to all regional water boards and acclimatisation societies in New Zealand. The questionnaire was also sent to those universities, government agencies, and private enterprises in New Zealand which may have had the need to apply bioassay techniques. The responses indicated that, although a few organisations used bioassays regularly, the majority were either unaware of the potential applications of this powerful biological tool, or daunted by the technical literature associated with the methods. This suggested that a review should start with a few definitions and a little philosophy about the reasons for using bioassays.

BIOASSAYS - WHAT ARE THEY?

The first problem is that "bioassay" means one thing in America, something else in Britain and Europe, and could mean either in New Zealand.

For example, in the Water and Soil Technical Publication 18 (1979), which reviews several biological methods for assessing water quality in New Zealand, 2 papers refer to bioassays; Jones uses the medical and traditional British definition while Burns adopts the ecological and expanded American usage. Unfortunately, these are very different and some standard definition should be adopted for use in this country.

The traditional British definition of a "bioassay" is a test which uses the observed degree of response of an organism or live tissue to estimate the unknown concentration of a chemical in water. The British coined the term "bioevaluation" (Alabaster & Lloyd, 1980) to mean the application of techniques which use the observed responses of a tissue, organism, or ecosystem to determine the effects of a known concentration of a substance in water. "Bioassay" is a search for information while "bioevaluation" is in the context of a search for meaning. Toxicity testing is viewed as a type of bioevaluation in Britain. However, the usual American (American Public Health Association, 1971) and frequent New Zealand usage has tended to apply the term bioassay as a general umbrella for all 3 categories. I propose to shelter under that umbrella in this paper.

A fish bioassay involves the experimental or routine measurement of the response(s) of fish to known or unknown environmental parameters which require evaluation under controlled circumstances. The intention is to provide information for either the prediction or interpretation of the effects of water quality on fish.

Although a variety of organisms have been used in bioassays, fish appear to be used most frequently for freshwater applications. This is probably because most concerns about the biological consequences of water quality relate to fisheries and it is desirable to conduct bioassays on the most sensitive species and life stage likely to be present. Where this is impractical or when whole ecosystem responses are of interest, then a range of other organisms may be used.

FUNCTIONS OF BIOASSAYS

There are a variety of predictive or interpretative functions which bioassays can serve. To a certain extent these can overlap or be complementary, but there are some useful distinctions which require that different purposes are discussed separately (Alabaster & Lloyd, 1980).

Establishing water quality criteria

Bioassay experiments can be conducted to determine "maximum acceptable concentrations of toxic substances" (M.A.T.C.) in water, or even unacceptable rates of change of habitat parameters (e.g., temperature, pH, flow). These findings may be appropriate for only 1 specific water-body and/or species of fish, or the studies may be designed to provide criteria for more general application. Such criteria may then be reflected in water quality classification standards or in water right conditions designed to provide a specified degree of protection to fisheries.

Screening tests

Simple bioassays can be useful to determine whether a particular discharge or chemical is likely to be hazardous for fisheries or could have been the cause of a previous undesirable event such as a fish kill. These short-term experiments are generally used as a first step in deciding whether further detailed investigation may be necessary.

Legal tests

Standardised and simple bioassay tests can be used to provide legal proof that discharge conditions have been breached, or that a particular event was or could have been caused by a specific pollutant or discharge. In some countries (e.g., Italy) the conditions set on the discharge of complex effluents include a requirement that at all times a designated dilution of the effluent should not be acutely toxic to fish under standard bioassay conditions.

Hence a simple bioassay test can provide legally acceptable proof of whether a discharge meets a fish toxicity standard. Another form of legal bioassay test involves the simulation of a pollution event to determine if a particular discharge was the most probable cause of a fish kill, the results being used as evidence in a prosecution.

Effluent monitoring tests

Simple or sophisticated bioassay techniques for monitoring wastewater discharges are becoming increasingly popular. These can include non-standard and innovative methods of exposing fish to raw or diluted effluents and observing specific responses. Techniques can range from monitoring fish in a continuous flow chamber using photoreceptors and computer analysis of behaviour, to placing fish into batch samples of effluent at periodic intervals to test for acute toxicity.

Freshwater habitat monitoring

Fish can be a practical and sensitive indicator of water quality in freshwater habitats. Where water quality is highly variable in location or fluctuates with time, fish bioassays can be used to measure habitat impacts directly. As with effluent monitoring either continuous or periodic bioassays may be most practical. Generally the whole habitat is not monitored but the bioassay is conducted at a representative location in a lake or river, or in a diverted sample or flow. The critical question with habitat bioassays is the choice of the response(s) to be measured.

MEASURABLE RESPONSES OF FISH

It is important to choose a response for measurement in fish bioassays that provides unambiguous and usable information for the researchers or managers of water quality, effluents or habitats. The response should be obvious enough to be detected yet sensitive enough to give warning of risk, should remedial action be required. In general the response chosen will depend on the purpose or function of the bioassay.

Mortality

The most observable and irreversible response is death, and either acute mortality (minutes to days) or chronic mortality (weeks to years) can be measured in bioassays. Such tests for toxic substances have been used for screening samples, legal purposes, setting water quality criteria, and monitoring of effluents and habitats. Despite the desirability of using less drastic responses the ultimate lethal response does have limited applications in legal, toxic substance screening, and water quality criteria investigations where new information is sought about the conditions likely to cause mortality.

For the purposes of monitoring habitats and effluents, a manager generally aims to protect the health of fish or an ecosystem rather than just preventing fish kills. Hence it is necessary to monitor sublethal responses. However, some of these sublethal responses (e.g., overturn, flashing, swimming failure) are indicators of impending mortality from which the fish may not recover, regardless of remedial action. Such responses are not necessarily variable with concentration or exposure time, and hence are of limited value in monitoring.

Behavioural responses

Changes in activity rate and orientation or attempts at avoidance or escapement are not always in response to toxic or life threatening conditions. There are difficulties in the measurement of these responses using sonar, infra-red perturbations, and voltage field fluctuations, as well as problems with acclimation causing a decreasing response. These aspects combined with the debate over interpretation mean that behavioural responses are of limited value for bioassay monitoring at present.

However, it appears that respiratory aberrations constitute one type of behavioural and physiological response which is highly sensitive, measurable, and useful as an indicator of stress or poor health.

Respiratory responses

Two respiratory responses are analogous to heavy breathing and coughing. The variables measured are ventilation frequency (the gill cover movement rate) and coughing spasm (gill purging) intensity. A variety of measurement techniques have been developed, including measurement of opercular nerve impulses and bioelectric potentials. The use of microcomputer analysis has permitted the development of commercially available remote sensors with hazard alarms. Such bioassay techniques are highly sensitive and have applications for establishing meaningful water quality criteria and for continuous monitoring of effluents and habitat (Solbe, 1979; Cairns et al., 1982; Thompson et al., 1982).

Biochemical/histological/physiological responses

Studies of biochemical changes (especially enzyme inhibition or induction), histological changes (especially tissue damage), and internal physiological responses (ranging from osmoregulatory ability to blood cell counts) are forms of bioassay commonly undertaken on fish. They are valuable for monitoring fish health, investigating the causes and mechanisms of fish kills, and may become useful in the establishment of water quality criteria.

Performance responses

Two indicators of fish "performance" are often measured for monitoring or investigation purposes. A reduction of maximum swimming speed in sublethal but stressful conditions can be useful in establishing water quality criteria. Measurement of growth rate can provide an integrated assessment of the health responses of fish to variable habitat conditions.

If growth rate monitoring should be undertaken, then as many environmental parameters as possible should be controlled. For this reason growth rate measurement is usually carried out under experimental conditions with standardised feeding of captive fish which are subjected to changes in only 1 habitat variable. Such chronic responses can be useful for justifying water quality criteria on economic grounds.

However, it is also possible to undertake growth rate monitoring of wild fish in natural habitats provided certain conditions are met. In particular, a large enough sample of individually marked fish must be available for recapture at regular intervals, as in repeated spawning migrations. Secondly, the population density and structure must be kept constant or below the level at which competition significantly influences growth.

Under these circumstances a change in growth rate can then be a response to modifications of important determinants of habitat quality. Such determinants include the abundance, distribution and dimensions of prey items and the physical/chemical characteristics of the waterbody. This form of bioassay has potential for development as a powerful technique for monitoring ecosystem functioning and productivity.

Life cycle responses

Some effects of poor water quality or sublethal concentrations of toxicants may not be detected during short-term bioassays or with tests on only one developmental stage of a fish. Usually there is a particular stage in the life cycle of each fish species which is most sensitive or vulnerable. Developmental impacts like failure to sexually mature, impairment of fertilisation success, and deformity and mortality at the egg, embryo or larval stages can be investigated through long-term chronic bioassays or by detailed acute bioassays during the critical stage. The results of such investigations are useful in establishing water quality criteria, and have also been used to determine the causes of reproductive failure. Long-term chronic bioassays are essential to investigate the effects of sublethal concentrations of toxicants on the disease resistance of fish.

Bioaccumulation responses

Fish may respond to environmental conditions in a way that has little effect on the fish itself, but which reduces the value of the fish to a human consumer or other predator. In New Zealand the bioaccumulation of mercury is a response which has required bioassay to determine both the sources of mercury in the food chain and the recommended maximum rates of consumption of different fish populations (Brooks et al., 1976). A second form of bioaccumulation is the tainting of fish flesh with non-toxic concentrations of chemicals ranging from industrial phenols to the geosmins derived from blue-green algae. Determination of the nature and sources of unpalatable flavours often requires bioassay studies.

The phenomenon of bioaccumulation through a food chain can make it more practical (for analytical and sampling reasons) to monitor the concentrations of certain pollutants through fish bioassays rather than by analysis of low and variable levels in water.

STANDARD METHODS AND NEW TECHNIQUES

With such a diversity of reasons for conducting bioassays and such a choice of responses to measure, it is reassuring to know that some standard methods exist. Unfortunately, every country and nearly every relevant agency has different standard methods. Until New Zealand adopts some standard terminology, techniques, and codes of practice, I can only guide you to the overseas literature.

The American text, Standard Methods for the Examination of Water and Wastewater (American Public Health Association, 1971) contains detailed descriptions of acute toxicity tests for static and continuous flow bioassays. The United States Environmental Protection Agency (1973) describes procedures for chronic bioassays. Wedermeyer & Yasutake (1977) outline recommended clinical methods for investigating the biochemical and physiological responses to fish to environmental stress. Alabaster & Lloyd (1980) provide a review chapter on the full scope of fish toxicity testing procedures in Britain and also summarise the standard Japanese, Polish and Swiss testing techniques. Jones (1979) reviews methodologies for acute and chronic toxicity tests and outlines recommended standard methods for laboratory tests in New Zealand. Solbe (1979) reviews bioassay studies and interpretation in Europe and describes the development of continuous bioassay pollution alarm systems based on remote monitoring of the heart beat, ventilation and coughing of captive fish.

Perhaps the best introduction to new techniques is the annual review issue of the Journal Water Pollution Control Federation; a comprehensive chapter on bioassay procedures and results is based on an extensive review for each year (e.g., Vol. 52, No. 6, June 1980). Reports on recent research show increasing interest in monitoring fish ventilation frequency, coughing, aberrant movement and swimming ability using remote sensors connected to computer analysers and alarms.

USE IN NEW ZEALAND

In order to determine the extent of fish bioassay usage in New Zealand, I carried out a literature survey and circulated a questionnaire to every regional water board, acclimatisation society and relevant government agency, as well as to those universities and private companies which may have had the need to apply bioassay techniques.

Of the regional water boards, 12 out of 20 responded, with 4 users reporting 20 bioassay tests. Only 7 out of 21 acclimatisation societies responded with 2 societies reporting 6 tests, of which 4 were in collaboration with the regional water board. Four central government agencies reported 12 bioassays and 1 university researcher has published on bioassay investigations conducted in collaboration with a government agency. One biological consultancy has conducted an undisclosed number of confidential bioassays for clients and I know of 1 major industry that has carried out toxicity testing of its effluents.

Despite the conclusion by Jones (1979) that there have been "no published studies on the effects of pollutants on aquatic animals in New Zealand", there were some publications prior to that time although not as numerous as the reported uses. Pike (1971) has published on the toxic effects of chlorine on brown trout in New Zealand waters and this research has subsequently been used by the United States Environmental Protection Agency (1976) to establish water quality criteria. Bioassay responses of indigenous and exotic fish to elevated temperatures have been investigated by Simons (1983, 1984) for the Waikato Valley Authority. The effects of DDT on trout reproduction were reported by Hopkins et al. (1969), and a Massey University investigation of the bioaccumulation of mercury and other heavy metals in trout was reported by Brooks et al. (1976). Several dozen other bioassays are recorded only in unpublished file reports, statements of evidence to courts and tribunals, and confidential reports to clients.

A preliminary analysis of the purposes for which the 35 reported bioassays were conducted reveals that 9 were screening tests, 10 were to establish water quality criteria or water right conditions, 9 attempted habitat monitoring, 2 tested effluents for toxicity, and 5 were used in fish kill investigations or prosecutions.

The species of fish tested included 17 bioassays using trout, 14 with native species and 2 each with mosquitofish and goldfish. The responses investigated included acute toxicity on 24 occasions, thermal tolerance in 7 instances, chronic impairment of function in 4 studies, 1 assessment of bioaccumulation response and 1 assessment of performance through growth. A variety of pollutants and habitat variables were investigated. In 21 cases the study involved a single known toxicant but 7 explored temperature effects, 6 tested complex discharges, 3 evaluated habitat quality in general and 2 attempted to track down mystery contaminants including a suspected fish virus of substantial concern.

FUTURE USES OF BIOASSAYS

A wide variety of potentially useful bioassay techniques have been developed and some of these have been applied to waters and fish in New Zealand on a limited scale to date. However, an increasing concern about water quality and an apparent change in emphasis towards biological monitoring suggest that bioassay usage is likely to become more regular, widespread, and sophisticated. Even local body politicians have suggested using fish to monitor the quality of treated sewage effluent waste streams (e.g., Dr J. Chick in the Taumarunui Gazette on 11 September 1984).

Two legislative developments may also stimulate managers of fish and water to use bioassay techniques on a regular basis. The pollution provisions of the 1951 Freshwater Fisheries Regulations prohibited any discharge of a variety of specified substances ranging from oil to chlorinated hydrocarbon pesticides, and also prohibited the discharge of any other substance to such an extent as to be injurious to fish. Judicial rulings made it clear that proof of previous or potential injury was not necessary for any of the list of specified substances (Lowe, 1976). However, the pollution provisions have been transferred to the 1983 Fisheries Act which prohibits the discharge of any specified or unspecified pollutant to the extent that fish are detrimentally affected. This suggests that Acclimatisation Societies may find it necessary to use bioassays more regularly to enforce the provisions of this legislation and demonstrate detrimental impacts.

Secondly, the 1967 Water and Soil Conservation Act contains water quality standards which prohibit the "destruction of natural aquatic life by reason of a concentration of toxic substances". This criterion has tended to result in effluent discharge conditions which only prevent acute mortality. The proposed new classification standards of the Water Quality Criteria Working Party (1981) are likely to supercede the old and inadequate standards in the near future. The new standards all contain criteria which prohibit any "substantial adverse effect on the aquatic community by reason of toxic substances", and any excessive bioaccumulation of toxic substances. When these criteria are adopted, it will be necessary to have access to bioassay results based on chronic or behavioural responses to toxic substances. For most species of fish and other fauna in New Zealand these data are not available. Consideration should be given to the type of bioassay facilities and philosophies that would be necessary to obtain this information.

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EPILOGUE

A.B. Cooper and R.D. Pridmore
Water Quality Centre, MWD, Hamilton

In the discussion at the end of the conference Dr Taylor presented 3 objectives of NWASCA. These were :

1 to maintain and enhance water quality;
2 to promote "best use" of water;
3 to protect waterways which have special attributes.

Many at the conference were convinced _a priori_ that the meeting of such objectives requires biological monitoring. The statutory requirement of the Water and Soil Act that "there shall be no destruction of aquatic life ..." (at least, for classified waters) would seem to make it incumbent on those charged with managing our waterways to undertake some kind of biological monitoring. The need for biological information in making management decisions was further argued in the papers of McBride and Roper. These authors emphasised the value of biological monitoring in checking the adequacy of receiving water standards and in detecting possible water right violations. They also noted that water managers should face little difficulty in justifying to their boards and the public the need for biological monitoring when the biota to be studied are of direct interest to man, e.g., recreational and commercial fisheries, nuisance slimes. Despite a consensus at the conference on the need for biological monitoring in managing freshwaters, the paper by Carter points out that few New Zealand water boards have a current commitment to biological monitoring. He notes, however, that there is a bright future for biological monitoring in water quality surveys and resource management provided that:

a some standardisation or guideline of methods be given;
b the data can be analysed quickly and results given in a concise form.

In recent years, New Zealand biologists have put considerable effort into critically reviewing existing biological methods for their applicability to New Zealand waters (Standing Biological Working Party, 1979, 1981) and in publishing recommended methods (Biggs et al., 1983). These efforts have been further extended at this conference, particularly by the papers of Etheredge, Biggs, Quinn & McFarlane, Winterbourn (sampling paper), Davenport, and Hicks. Water managers in New Zealand are also becoming well-served with manuals for the identification of freshwater flora and fauna (e.g., Chapman & Lewis, 1976; McDowall, 1978; Taylor, 1980; Winterbourn & Gregson, 1981; Pridmore & Hewitt, 1982; Upritchard, 1985). Although these papers make useful contributions, in general, biological methods are not yet sufficiently standardised and rapid to meet the criteria identified by Carter; for this reason water managers require some assistance in identifying those methods most appropriate for meeting specific management objectives. At present, biologists seem reluctant to provide such a "cookbook" for fear of it being indiscriminately used (e.g., see paper by Harper). But the absence of a "cookbook" should not provide reason for abandoning freshwater biological monitoring. Many of the water boards (by our reckoning, at least 13 of the 20) already have staff whose formal training has been in some field of biology. It is our view that these people should be capable of perusing the conference proceedings and the available literature to decide on those procedures which may assist them in particular studies. There is no denying, however, that their decisions would be much easier to defend if the methods they chose were recommended for use by an appropriate national body (e.g., New Zealand Limnological Society, NWASCA, Catchment Authorities Association).

An area that has lacked attention by many New Zealand freshwater biologists is data presentation. The opinion was expressed during the conference (see papers by McBride, Carter, Pridmore, Penny, and Stark) and in the conference discussion that too often biologists present their data solely as species lists. Although ecosystems are complex (see papers by Winterbourn, Towns, and Penny), attempts must be made to simplify and codify the information gathered. That this is possible has been shown by several of the conference papers, with Pridmore describing the use of Cochran's Q-test for presence-absence data, Quinn & McFarlane and Hickey demonstrating the use of phototroph-heterotroph ratios for describing microbial communities, Biggs evaluating the usefulness of algal indicators of pollution, and Stark describing a variety of indices for use with macroinvertebrate data. The value of these techniques lies in their ability to reveal differences in biological communities. The challenge posed by management needs is to relate such differences to particular causes and to predict the direction and extent of change in biological communities that will occur upon certain proposed management options being exercised. The immense value to management of meeting this challenge is clearly demonstrated by the usefulness of the quantitative relationship established between total phosphorus and algal biomass in lakes (e.g., see Pridmore et al., 1985) as this makes it possible to predict the algal concentrations which will occur under different phosphorus loading regimes. It is incumbent upon research biologists to demonstrate further relationships (preferably quantitative) between biological response and phenomena causing this response. This is necessary if reasoned management decisions are to be made upon the results of biological monitoring.

Future directions

This conference has served to gather together state-of-the-art information on biological monitoring techniques and the proceedings should serve as a useful reference document. The conference also highlighted those areas where further effort is required before biological monitoring becomes an integral part of water management in New Zealand. The need for a step-by-step handbook on biological monitoring for management was identified but the specialist biologists thought that this was not currently feasible given the extent of present knowledge. It therefore appears that production of such a handbook must remain a long-term objective. In the short-term, there is a need for research biologists to demonstrate the management value of their science; viz., more case studies are required which show how the results of a biological monitoring programme can be used to supply answers to specific management questions. Research biologists at the conference held the view that to meet this need water managers must become more involved so that the questions to be answered are clearly defined.

It is the organisers hope that this conference has gone some way to making the research biologist and the water manager more aware of each other's problems and perspectives and therefore served to focus future efforts in biological monitoring in freshwaters.

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