BIOMONITORING, AND THE MACROINVERTEBRATE FAUNAS OF CANTERBURY STREAMS

A thesis
submitted in partial fulfilment
of the requirements for the Degree
of
Masters of Science in Zoology
in the
University of Canterbury

by
Aslan E. Wright-Stow

University of Canterbury
2001

TABLE OF CONTENTS

ABSTRACT

A wide-ranging macroinvertebrate and physico-chemical survey of 230 3rd and 4th order streams throughout the Canterbury region was conducted between November 1999 and March 2000. Kick-net sampling, spot water sampling and habitat surveys were used. Invertebrate community composition appeared to be influenced by two overriding factors; the physical condition of the stream, and the amount of anthropogenic development within the catchment. Faunas dominated by Ephemeroptera and Trichoptera and with some Plecoptera present tended to occur in pristine high altitude streams with low conductivity, well vegetated riparian zones, heterogeneous streambed substrates and periphyton consisting primarily of diatom biofilms. Faunas dominated by Crustacea, Oligochaeta and Chironomidae occurred commonly at degraded lowland sites with high conductivity, little or no riparian vegetation, more homogeneous fine substrates and periphyton dominated by thick mats and filaments. Between these two extremes, gradual change in faunas was found, with Trichoptera dominating intermediately disturbed sites. A striking decrease in the relative abundance of Ephemeroptera along an ecological gradient appeared to be associated with increasing intensity of landuse.

A comparative investigation of three biotic indices widely used in New Zealand for assessing stream health, indicated that the MCI, QMCI and SQMCI may not assess the health of all sites, consistently. The inconsistencies were probably brought about by two factors. Firstly, presence-absence data used in calculating the MCI may not detect subtle differences in community structure, whereas the quantitative data used by the QMCI and SQMCI may pick up small differences and therefore group sites into different degradation bands. Secondly, published degradation bands for the MCI, QMCI and SQMCI do not appear to be directly comparable in Canterbury. The utility of a quantitative MCI with low-level (order, class, phylum) identification was also investigated, and found to be a potentially viable alternative to the MCI and its derivatives when a low-cost, rapid assessment technique is needed, but expertise in identification is lacking.

The health of streams in the Canterbury region as assessed by the MCI, was investigated. The MCI indicated that streams were generally more healthy if they

were further inland, at higher altitudes, and were in forested or unmodified catchments. Stream health was poorest in lowland sites with pastoral and urban/city developed catchments, although 42 pastoral sites with MCI values >100 and taxonomic richness >25 indicated that healthy streams were attainable in agriculturally developed land.

Finally, a multimetric approach for assessing the health of Banks Peninsula streams using macroinvertebrates was developed. Five biological metrics (QMCI, % EPT, % Chironomidae, % Mollusca, No. Ephemeroptera) that best discriminated selected reference sites from sites impaired by habitat disturbance and organic pollution were combined into an index of biological integrity; the Banks Peninsula Macroinvertebrate Index (BPMI). Strong relationships between the BPMI and MCI and QMCI suggested that the extra effort required to produce a multimetric index did not result in improved assessment of stream condition. However, a multimetric index can provide additional information on the source of degradation to a stream and indicate where restoration or mitigation should be focussed.

CHAPTER ONE

GENERAL INTRODUCTION

1.1 Project origins

The introduction of the Resource Management Act 1991 (RMA) in New Zealand has required that a holistic approach to the sustainable management of resources is taken. Thus, the Act requires that ecosystems are sustainably managed for the use, development and protection of water resources. Responsibility for this was given to regional councils through the development of policy statements, regional plans and resource consents. Regional councils are also required to report every 3 years on the condition of rivers and streams for State of the Environment Monitoring (SEM) with the aim of gathering information for decision making about the environment. In order for a regional council to implement the Act and report on the state of the environment, its water managers require information on the condition of water bodies and factors affecting their condition. This thesis stems from Environment Canterbury's 1999-2000 SEM and RMA biomonitoring survey of the Canterbury region with which I was actively involved. The information collected was also used for testing the value of a River Environment Classification (REC) system developed by the National Institute of Water and Atmospheric Research (NIWA) (Snelder et al. 1998) as part of the Ministry for the Environment's (MfE) invertebrate indicator programme. The REC is based on a hierarchical model of physical factors that are known to affect instream biological communities. Its usefulness to constrain variance in macroinvertebrate communities has been tested by Suren et al. (2000).

1.2 Background

The composition of a macroinvertebrate community is determined by a variety of factors including physico-chemistry (e.g. Cobb et al. 1992, Collier 1995, Jacobsen et al. 1997, Death 2000, Winterbourn 2000) dispersal (Williams and Hynes 1976, Edwards and Sugg 1993, Boothroyd and Stark 2000) and biogeography (Harding 1994, Boothroyd and Stark 2000). Of these factors, catchment landuse is often paramount because of its influence on physico-chemical conditions and resource availability (Thompson and Townsend 2000). Previous studies have shown that the structure of a stream community changes along an 'ecological gradient' associated with changes in landuse (Harding and Winterbourn 1995, Harding et al. 1999). Generally, sensitive taxa such as Ephemeroptera, Plecoptera and Trichoptera (EPT) are replaced by enrichment-tolerant taxa, such as those found within the non-insect groups Crustacea and Oligochaeta, and the insect family Chrionomidae (Diptera) as the amount and intensity of catchment development increases (Quinn and Hickey 1990a, Harding and Winterbourn 1995, Quinn et al. 1997, Harding et al. 1999).

Following initial European settlement of Canterbury around 1850, parts of the plains were drained and cleared for agriculture (McDowall 1998). Tussock grassland that covered much of the plains and high country was burnt, and the beech forest retreated under the impact of agriculture (Johnston 1969). Presently, about three quarters of the land in the Canterbury region is used for farming (Cook 1999).

Typically, anthropogenic and particularly pastoral development alters key aspects of stream habitat that influence invertebrate communities, including the quantity and quality of food available, the physical shape of the stream, flow regime and water quality (Winterbourn 1986). Removal or modification of riparian vegetation may reduce habitat availability for adult insects (Collier and Smith 1998) and can result in higher water temperature through reduction of shade especially in slow flowing streams (Davies-Colley and Quinn 1998, Biggs et al. 1998). Losses of riparian cover may also be associated with invasions of macrophytes (Bunn et al. 1998) and increase periphyton growth (Towns 1981, Davies-Colley and Quinn 1998). Riparian root systems can also reduce erosion (Waters 1995) and allochthonous inputs (leaves, twigs) from bank vegetation provides a supply of food to some invertebrates (Cummins 1974, Cottam 1999). In combination these factors often mean that a reduction in bank vegetation results in a change in invertebrate community composition and abundance (Clenaghan et al. 1998) and changes in stream ecosystem function (Stone and Wallace 1998). Anthropogenically modified streams are often enriched with nutrients and increased sediment inputs (Quinn et al. 1997), which can also result in changes in invertebrate community composition (Quinn and Hickey 1990a, Collier et al. 1998).

Both sediment supply and enrichment levels influence periphyton communities (Kutka and Richards 1996), and excessive sediment deposits change the character of surface and hyporheic habitats, increase embeddedness of stones and reduce the interstitial spaces (Ryan 1991, Davies-Colley 1997) where many insects live (Waters 1995).

Because of their central role in stream food-webs, macroinvertebrates can be viewed as integrators of information on stream ecosystem structure and function and water quality (Winterbourn 1999). They provide information on the energy base of the ecosystem, water quality, habitat diversity, and the availability of appropriate kinds of food to support native fish and trout populations. For this reason, invertebrates make ideal monitoring agents (Winterbourn 1999).

Macroinvertebrates are often used, at least in part, to assess stream health. Twelve of the 16 regional councils in New Zealand, as well as the Nelson City Council, and Marlborough District Council (equivalent to regional councils) presently use macroinvertebrates for biomonitoring or State of the Environment Monitoring (Winterbourn 1999, Jon Harding pers. comm.). As a biomonitoring tool, macroinvertebrates have several advantages over other groups of biota. They are easy to sample, are relatively easy to identify, and they provide a variety of responses to changing environmental conditions because of the diversity of species, feeding habitats and life-histories (Boothroyd and Stark 2000). These and other advantages and disadvantages of using macroinvertebrates for biomonitoring are given in Table 1.1.

Table 1.1 Advantages and disadvantages of using macroinvertebrates for freshwater lotic biomonitoring (from Boothroyd 1999).

Advantages	Disadvantages
relatively long life spans makes then good integrators of environmental conditions over time	discrimination between effects of pollution and other factors often difficult
sedentary nature and thus representation of local conditions	complex life histories may result in misinterpretation and seasonal responses
varying tolerance to changes in environmental conditions	high spatial heterogeneity resulting in high replication for statistical rigour
diversity of forms, feeding habits, refuge and habitat requirements, and life cycles	specific behaviour patterns (e.g. drift) may confuse interpretation
ease of identification	poor taxonomic knowledge of some major groups
ease of sampling	
increasing knowledge of ecology	

A number of approaches to biomonitoring have been developed in the form of diversity, comparative and biotic indices, and multimetric and multivariate assessments. Diversity indices combine information on taxonomic richness, dominance and/or evenness (Winterbourn 1999), components that describe the response of a community to the quality of its environment (Boothroyd and Stark 2000). Comparative (or similarity) indices in contrast to diversity indices make comparisons between two or more populations or communities, often control and reference sites, and are most frequently used to assess the degree of change brought about by a specific impact such as logging of a catchment (Winterbourn 1999).

Biotic indices are numerical expressions coded according to the known pollution tolerances of individual taxa, which are combined into a single score. They have found wide spread use in New Zealand, particularly the Macroinvertebrate Community Index (MCI) and its derivatives. Stark (1985) developed the MCI and its quantitative variant (QMCI) for the detection of organic enrichment in stony-bottomed streams. Later, a semi-quantitative variant (SQMCI) was developed to reduce sampling and processing effort (Stark 1998).

The multimetric approach involves using a variety of indices (metrics) at the same time to evaluate site conditions. Multimetric methods were developed from the U.S. Rapid Bioassessment Protocols (Plafkin et al. 1989), and were originally designed to determine whether a stream contained an expected faunal community. Commonly, multimetrics use combinations of composition (e.g. % Ephemeroptera, % Trichoptera, % Diptera), richness (e.g. number of taxa), tolerance (e.g. MCI, QMCI) and feeding group (e.g. % predators, % filters) metrics. This technique requires careful selection of reference sites, a consideration that is often overlooked until too late (Winterbourn 1999).

Finally, multivariate predictive techniques are becoming increasingly used for biomonitoring. Based on methods developed in the United Kingdom and modified for use in Australia, RIVPACS and AusRivAs are multivariate software techniques that predict the aquatic macroinvertebrate families expected to occur at a particular site. They use combinations of environmental variables as predictors and compare site assemblages with those of reference sites (Boothroyd and Stark

2000). AusRivAs differs from RIVPACS in several ways, including the ability to incorporate the habitat from which the sample was taken (riffle, edge habitat), the geographic region from which the sample was taken, seasonal factors, and that it reduces emphasis on rare taxa (Boothroyd and Stark 2000).

As discussed above, the structure of stream invertebrate communities often changes along environmental gradients that are associated with human activities and landuse, and because of this they make useful biomonitoring tools (Fig. 1.1). Much of the information that has been used to derive the relationships, between invertebrate community composition and environmental gradients shown in the figure, has come from studies at the catchment and reach scale. For example, intensive studies at Whatawhata in the Waikato (Quinn and Cooper 1997) and on the Taieri River in Otago (Townsend et al. 1997) have aimed to relate community structure to impacts of landuse. Few studies however, have established how invertebrate community composition is associated with environmental factors at a regional scale in New Zealand (Collier 1995).

The research reported on in this thesis is based in a large-scale (230 sites) macroinvertebrate and habitat survey carried out in the Canterbury region. Specifically, I have investigated the roles of catchment, riparian and physico-chemical factors, which might affect the composition of invertebrate communities in a range of 3rd and 4th order streams (Chapter 3). Chapter 4 compares results obtained by the MCI, the QMCI and the SQMCI for the assessment of Canterbury stream health, and explores reasons for differences in the way they rank and group the same sites. In addition the utility of a quantitative MCI with low-level (order, class, phylum) identification (referred to as Ordinal QMCI; OQMCI) is considered in this chapter. Chapter 5 focuses on Canterbury stream health as assessed by the MCI and relates it to several environmental variables; altitude, catchment landuse, stream water conductivity, stream clarity, water temperature, stream-bank vegetation, stream substrate composition and algal composition. Finally, a multimetric index is developed in Chapter 6 for assessing the health of Banks Peninsula streams.

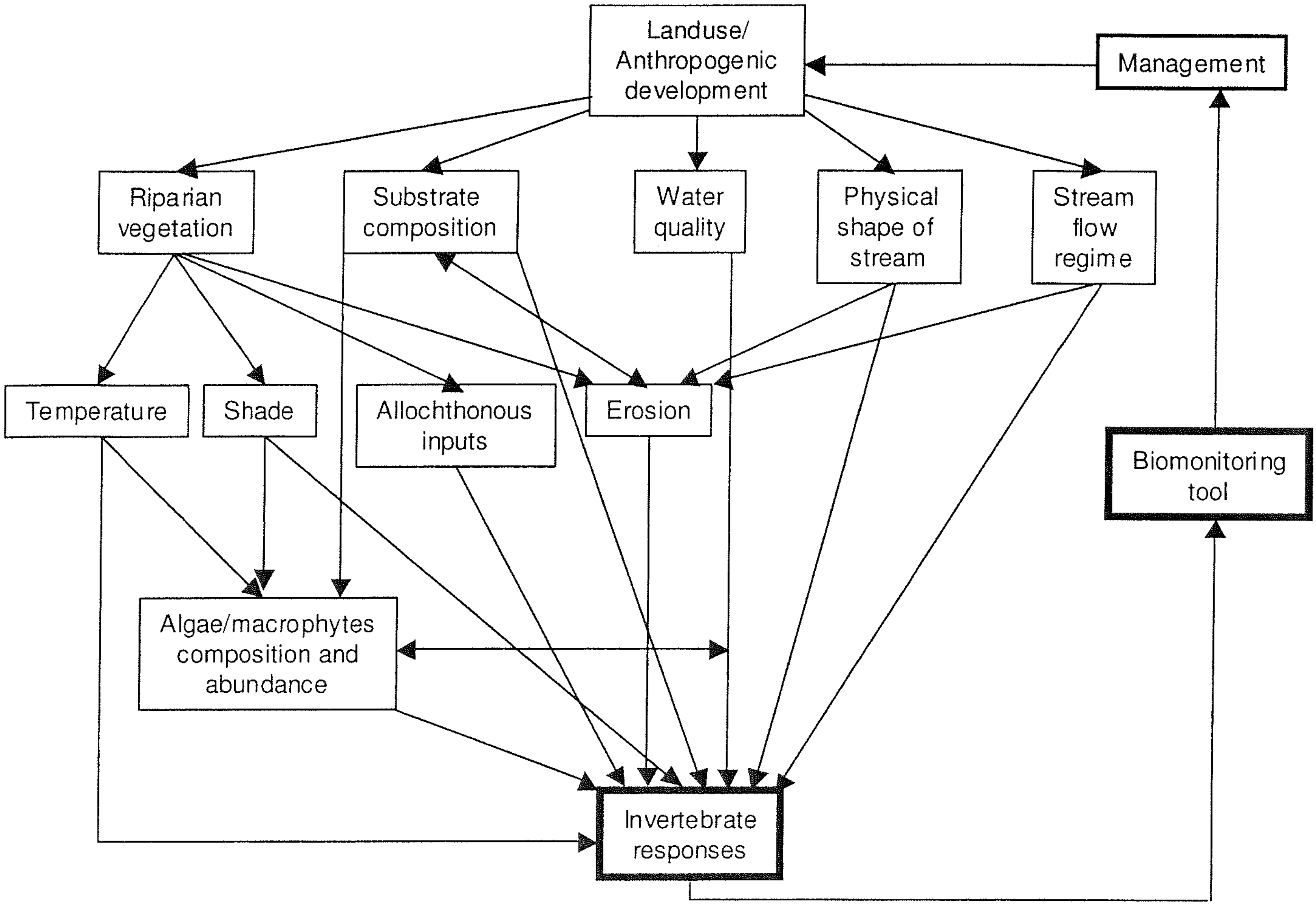


Fig. 1.1 Relationships between components of river ecosystems examined in this thesis. Note how invertebrates can be seen as integrators of information on stream ecosystem structure, function and water quality, and thus make ideal monitoring agents. Bold boxes are the primary areas of focus reported on in this thesis.

CHAPTER TWO
GENERAL METHODS

2.1 STUDY AREA

2.1.1 Canterbury landscape

Canterbury is the largest geographic region in New Zealand with an area of 45 346 square kilometres (Cook 1999). The landscape that we see today was formed by processes of mountain building, glaciations and post glacial erosion (Gage 1969). Thick accumulations of glacial deposits provide evidence of successive ice advances, while post glacial erosion and deposition is indicated by fans formed since the disappearance of the ice (Fitzharris et al. 1992). Four distinct landscape area are recognised in the Canterbury region; the greywacke stone of the Southern Alps, rolling foothills of much younger rock, Banks Peninsula, which is a tertiary volcanic complex, and the huge alluvial fans of the plains (Cook 1999).

The greywacke mountains shape the soil types and weather patterns of Canterbury (Vucetich 1969). Rising to elevations of >1500m a.s.l. they act as barriers to westerly winds, influencing the climate, and supply material which forms the plains. The greywacke sandstone is easily shattered and provides a continuous supply to the major rivers via shingle fans. Harder stones and boulders are deposited on the plains by the larger rivers such as the Waimakariri, Rakaia and Rangitata, and finer material is carried out beyond the river mouth and deposited on the continental shelf. The volcanic landform of Banks Peninsula consists of underlying basaltic rock covered in loess soil types (Vucetich 1969). The plains consist of a very thick layer of gravel covered in variable thickness loess deposits which are fertile and easily eroded (Vucetich 1969, Cook 1999).

2.1.2 Climate and weather

Canterbury experiences some of New Zealand's greatest climatic extremes. It lies in a latitude of prevailing easterly winds and when an east-moving anti-cyclone moves onto the Western Tasman sea, wind blows over Canterbury from the north-west (de Lisle 1969). The nor' wester is a strong turbulent wind often bringing rain to the West Coast, and high temperatures and low humidity east of the Southern

Alps. It is the predominant wind in the foothills and high country, whereas in coastal districts nor' easterly or easterly winds are most common. Strongest winds come from the west, with gusts of >100kph recorded at Christchurch on average 2-3 days a year (de Lisle 1969).

Rainfall in coastal parts of Canterbury is not high compared with many other parts of New Zealand (about 625 mm/year, Fitzharris et al. 1992). However, rainfall in the foothills and high mountains averages >1000 mm/year and 2000-4000 mm/year, respectively (Cook 1999). Rainfall is spread evenly throughout the year, except on Bank's Peninsula, which has a pronounced winter maximum (de Lisle 1969). High temperatures and wind on the plains mean that droughts occur frequently during spring and summer. January and February are the warmest months, July is the coldest, and most of Canterbury receives about 1900-2000 sunshine hours a year (de Lisle 1969). Canterbury shares the highest temperature ever recorded in New Zealand with Marlborough at 42^{0}C. Midwinter minimum daily average temperature for coastal Canterbury is 1.6^{0}C, inland is distinctly cooler with minimum midwinter daily average of –2.5^{0}C. Maximum daily average temperature in summer is between 20-23^{0}C, warmer than most other South Island regions except Central Otago and Blenheim (Cook 1999).

2.1.3 Modification of natural environment by man

Following their arrival in about the ninth century, Maori destroyed much of the forest and scrub on the Canterbury plains in a succession of fires (McDowall 1998). European settlement occurred around 1850 and at that time the plains probably consisted of interspersed swamps and native flax, grass, and shrubs, which were drained and cleared by colonial farmers for agriculture (McDowall 1998). The tussock grassland that covered much of the plains and high country was burnt, and the beech forest retreated, not through milling, but under the impact of agriculture. In contrast the podocarp forest on Banks Peninsula, the plains and foothills was deliberately milled and cleared (Johnston 1969). Three quarters of the land in the Canterbury region is now used for farming (Cook 1999). Agriculture dominates the rural economy and approximately 90% of the farmland in the region is either grazing, arable, fodder or fallow land. Only a small proportion (0.3%) is used for horticulture (Cook 1999).

2.2 Study sites

A total of 230 3rd and 4th order streams were sampled from throughout the Canterbury region between November 1999 and March 2000 (Fig. 2.1). The sites were selected from NZMS 260 series 1: 50 000 maps and incorporated a range of conditions (i.e. catchment landuse, topography, source of flow, geology, vegetation types) commonly found within the region (Table 2.1). The sampling sites ranged from streams draining mountain catchments with relatively pristine native vegetation, to lowland catchments with upstream areas in pasture or urban development (Figs. 2.2-2.5). Most sites were relatively close to, but upstream of, roads or farm tracks, although five remote mountain sites had to be reached by helicopter. The sampling sites were all below 1100 m a.s.l.

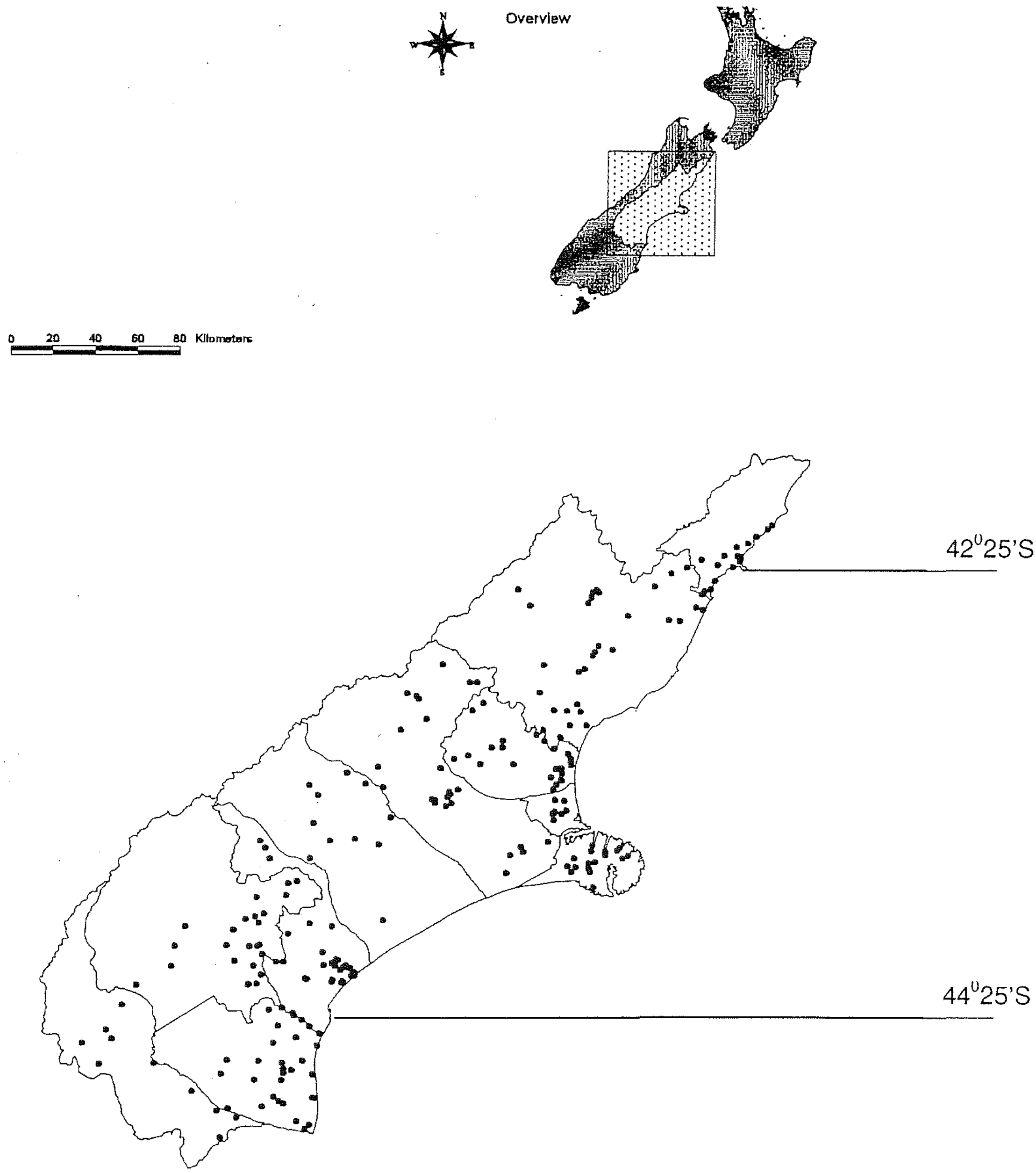


Fig. 2.1 Location of sampling sites within the Canterbury region.

Table 2.1 Summary of site and catchment variables. B shows the proportion of sites sampled within each landuse and topographic category.

A.

Variable	Median	25 percentile	75 percentile	Range
Altitude (m a.s.l)	160	20	350	0-1100
Average stream width (m)	5.55	3.74	9.38	1.30-53.3
Average stream depth (m)	0.33	0.25	0.45	0.10-0.91
Average steam velocity (mS^{-1})	0.63	0.36	0.9	0-2.5
Temperature (^{o}C)	14	12.5	15.5	7.5-19.5

B.

	% of total sites
Landuse:	
Exotic forest	3
Indigenous forest	11.7
Intensive pastoral farming[1]	34.9
Extensive pastoral farming[2]	42.6
Tussock	5.2
Urban/city	2.6
Catchment topography:	
$>20^{0}$	33
$10\text{-}20^{0}$	27.4
$5\text{-}10^{0}$	12.6
$<5^{0}$	27

[1], Dairy, deer, horticulture
[2], Sheep

2.3 METHODS

2.3.1 Invertebrate sampling

The sampling strategy used in this research was intended to minimise inter-site effects of season and river size on invertebrate abundance and composition. Invertebrates were sampled twice (17 November – 20 December 1999, and 31 January – 23 February 2000) at base-flow levels by six field workers. All field workers were trained prior to sampling to ensure that a consistent level of information was collected. Invertebrate samples were collected semi-quantitatively using a 500 μm mesh "kick-net" sampling technique, from three runs in three reaches at each site. All habitat types within each run were sampled. The three replicates were combined in the field to give one sample per site. This was preserved in 90% ethanol. Invertebrates were collected by moving across the stream disturbing the substrate to a depth of 10 cm within a 50 cm wide strip in front of the net. Hands and feet were used to disturb the substrate and remove

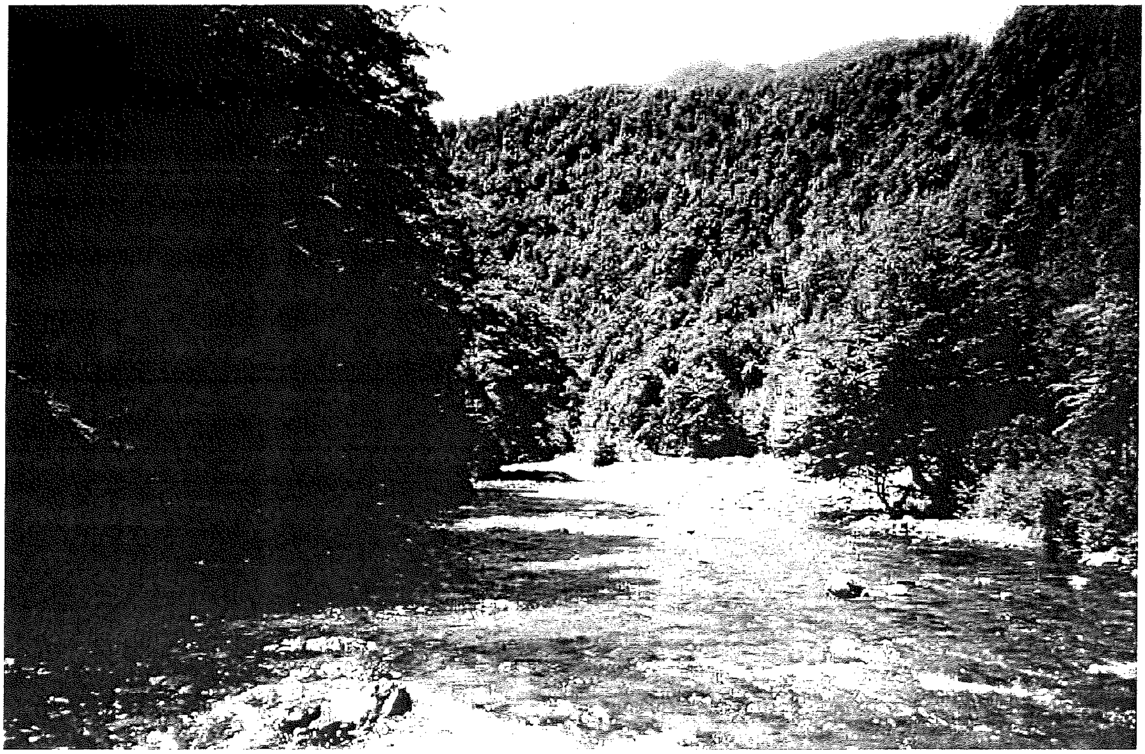


Fig. 2.2 Andrews Stream, Arthurs Pass National Park; example of a high altitude 'pristine' stream.

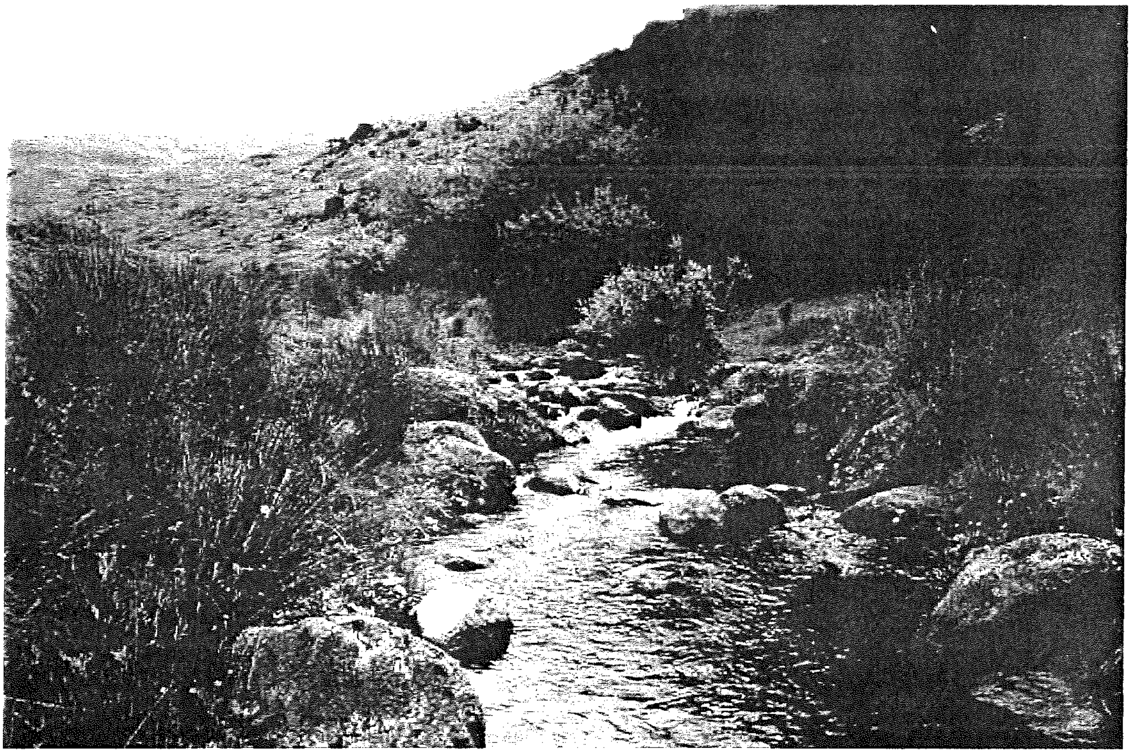


Fig. 2.3 Opara Stream, Okains Bay; example of a typical Banks Peninsula stream.

Fig. 2.4 North Opihi Stream, near Timaru; example of an agriculturally impacted stream.

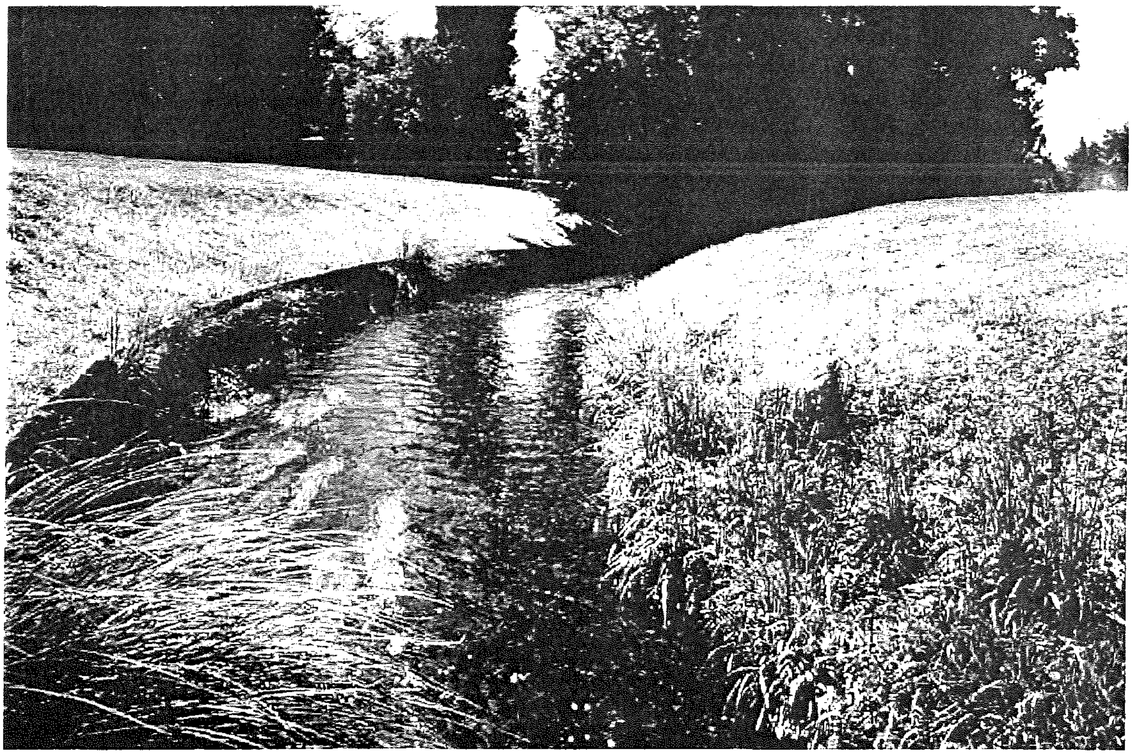


Fig. 2.5 Waimari Stream, a tributary of the Avon River in Christchurch; example of a stream impacted by urban development.

attached invertebrates. Streams that had large beds of macrophytes were sampled by brushing and washing plants into the net, or by placing them in a bucket for washing. Samples were elutriated in the field to remove large stones.

In the laboratory, samples were washed through a 500 µm sieve and sub-sampled to a manageable level (up to 1/32). The sample or sub-sample was transferred to a perspex Bogorov tray (Winterbourn and Gregson 1989) and sorted at up to 35X magnification under a dissecting microscope. Taxa were identified to genera where possible, or to alternate levels used by Stark (1993). The first 100 invertebrates in each sample were counted, and the remainder of the sample was scanned for taxa not found previously. QA/QC controls showed that operator variability between samples was much less than the natural variability observed between rivers, and that 100 counts gave very similar results to 200 and 300 counts (Suren et al. 2000). See Appendix 1 for validating the modified 100 fixed count method and assessing between-operator variability. Taxa were identified using Chapman & Lewis (1976), Winterbourn & Gregson (1989) and Moore (1997).

2.3.2 Environmental measurements

Environmental measurements were made once, between 17 November and 20 December 1999. Site elevations, source of flow and river order were determined from NZMS 260 1: 50 000 maps. Catchment, instream and riparian conditions were assessed visually, either categorically or as percentages. Environmental measurements recorded are shown in Table 2.2.

Table 2.2 Environmental measurements recorded at each site and assessment methods.

Environmental variable	Assessment method
Dominant catchment landuse	C
Percent canopy cover over stream bed	Q
Percent bank vegetation cover over low flow banks	Q
Bank stability	C
Bank substrate heterogeneity	C
Macrophyte abundance	C
Bottom substrate providing cover for fish and invertebrates	C
Siltation/embeddedness	C
Packing of substrate material	C
Percentage stable material	C
Shape and angularity of substrate	C
Substrate heterogeneity	C
Periphyton cover	C
Composition of streambed	Q
Percent algal types	Q

C, Categorical assessment
Q, Qualitative assessment

Maximum stream width (wetted zone) and depth in each of the three runs sampled was measured, and spot water temperature was recorded with a mercury thermometer. Average water velocity in each run was determined by timing a float over a given distance (generally 10 m), and a water sample was taken at each site from which conductivity and pH were measured in the laboratory on the day of collection. Clarity was determined with a clarity tube (Biggs et al. 1998). Field data sheets are given in Appendix 2.

2.3.3 Environmental assessment scores

Assessment scores were developed to help condense complex environmental information. Periphyton, streambed composition and bank vegetation 'scores' were calculated following the Stream Health Monitoring and Assessment Kit (Biggs et al. 1998). Tables 2.3, 2.4 and 2.5 show how the scores were calculated.

Table 2.3 Method used to calculate periphyton scores. Modified from Biggs et al. (1998).

Periphyton (on exposed surfaces)		Periphyton score	Run 1					Run 2					Run 3					*Enter average*
			Stone No.					Stone No.					Stone No.					
			1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	
Thin mat/film:	Green	**3.5**	*(a)*	*Enter proportion * peri score*														*(b)*
	Light brown	**5**																
	Black/dark brown	**5**																
Medium mat:	Green	**2.5**																
	Light brown	**3.5**																
	Black/dark brown	**4.5**																
Thick mat:	Green	**2**																
	Light brown	**2**																
	Black/dark brown	**3.5**																
Filaments, short:	Green	**2.5**																
	Brown/reddish	**2.5**																
Filaments, long:	Green	**0.5**																
	Brown/reddish	**2**																
No Periphyton:		**0**																

Site score *(c)*

(a) Enter proportion (0.0-1.0) of each stone covered by each periphyton type, multiplied by periphyton score.
(b) Enter average score for each periphyton type, for the three combined runs
(c) Calculate average score for each of totals in (b) = **site score**

Table 2.4 Method used to calculate substrate composition scores. Modified from Biggs et al. (1998).

	Bedrock	Boulders (> 25cm)	Large cobbles (12-25cm)	Small cobbles (6-12cm)	Gravels (0.2-6cm)	Sand	Mud or silt	Man made (e.g. concrete)	Woody debris	Water plants (rooted in stream bed)
Score	1.25	3.75	5	3.75	2.5	1.25	0	0	2.5	2.5
enter %										
enter score * %										

enter total of (score * %) [ ]

overall score = total (score * %) / 100 [ ]

Table 2.5 Method used to calculate riparian vegetation scores. Modified from Biggs et al. (1998).

	Native trees	Wetland vegetation	Tall tussock grassland, not improved	Deciduous trees (willow, poplar)	Evergreen trees (conifers)	Scrub	Rocks, gravels	Short tussock grassland, improved	Pasture grasses and weeds	Bare ground, roads, buildings
Score	4	4	3.6	3.6	3	3	3	2.6	0	0
%, true left										
%, true right										
Total % (L + R)										
Total % * score										

Total of all (score * %) [ ]

overall score = total (score * %) / 100 (i.e., average L and R bank scores added) [ ]

An overall habitat assessment score was also calculated. Individual environmental measurements were assigned scores based on their perceived importance to stream health in a manner similar to that in the Waikato Region AUSRIVAS/RIVPACS trial (Coysh and Norris 1999). Initially, 15 variables were included; landuse, canopy cover, bank vegetation, bank stability, bank heterogeneity, bank and channel roughness elements, channel heterogeneity, macrophyte type, substrate availability, siltation, packing, % stable substrate, rock angularity, substrate heterogeneity, and periphyton types. Multiple regressions were used to determine which variables were most strongly correlated with stream health indices, and subsequently the habitat assessment was reduced to seven variables; landuse, bank vegetation, bank heterogeneity, macrophyte type, siltation, packing, and substrate heterogeneity (Table 2.6). In summary, streams were given high scores if landuse was of low intensity, banks were heterogeneous and well vegetated, macrophytes, if present, were not obstructing flow patterns, and substrate was tightly packed, of assorted sizes and did not include high proportions of fine sediments.

2.3.4 Data analyses

Invertebrate abundance data from the November-December 1999 sampling period were added to data from January-February 2000 to produce a combined data set. Where physico-chemical measurements were recorded during both sampling periods (periphyton scores, conductivity, water clarity, water temperature, flow velocity, stream width and depth) results were averaged. A number of biological indices were calculated and formulae and justification for using these are given, where appropriate, in subsequent chapters. They included taxonomic richness, numbers of EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa, the Macroinvertebrate Community Index (MCI) and its quantitative (QMCI) and semi-quantitative (SQMCI) variants (Stark 1985, Stark 1998), and relative proportions of each major taxonomic group.

Multivariate (TWINSPAN, DECORANA, PCA) and descriptive (correlations, ANOVA, Tukey test, frequency distributions, scatter plots) statistics were calculated with PC-ORD (McCune and Mefford 1999), STATISTICA and EXCEL software, and are discussed in the following chapters where appropriate.

Table 2.6 Variables measured and scores given in creating the habitat assessment score. Overall score is the sum of each categorical score.

	Score
Dominant catchment landuse	
1. Indigenous forest	40
2. Exotic forest	30
3. Tussock (not/lightly grazed)	20
4. Tussock (modified and grazed)	20
5. Pasture - sheep farming	10
6. Pasture - dairy farming	5
7. Pasture - deer farming	5
8. Horticulture - crops	1
9. Scrub land	1
10. Urban	1
11. City	1
Bank vegetation cover (low flow banks only)	
1. Over 80% of stream bank surfaces covered by vegetation.	20
2. 50-79% of the streambank surfaces covered by vegetation.	10
3. 25-49% of the stream bank surfaces covered by vegetation.	5
4. Less than 25% of low flow stream banks covered by vegetation.	1
Bank heterogeneity	
1. Natural stream meander pattern, irregular sided banks, high bank heterogeneity from changes in shape, substrate or vegetation: natural banks.	20
2. Natural stream meander pattern irregular sided banks, low heterogeneity from bank substrate of vegetation, banks natural to semi-natural.	10
3. Channelled stream, meander patter greatly altered, few bends or sinuosity. High heterogeneity of substrate or vegetation.	5
4. Channelled stream, meander pattern greatly altered, few bends or sinuosity. Low heterogeneity of bank substrate or vegetation. Vertical or uniformly sloped banks, often with reinforcing.	1
Macrophytes	
1. Clumps of submerged macrophytes not covering the majority of the streambed (i.e., <50%), and not greatly obstructing flow patterns in the stream, OR an absence of plants in stony streams.	20
2. Clumps of emergent macrophytes, not choking the streambed or causing stagnation, OR submerged macrophytes covering the majority of the streambed (i.e., >50%) but not obstructing the water flow.	10
3. Submerged macrophytes covering the majority of the streambed, and obstructing the flow in the channel, reducing water velocities in places.	5
4. Emergent macrophytes completely or largely choking the channel, causing areas of stagnation and excessive silt entrapment.	1
Siltation/embeddedness	
1. Larger substrates (gravel, cobble and boulders) associated with few (0-25%) fine sediments (i.e. clay, silt, sand).	20
2. Gravel, cobble and boulder particles associated with 25 - 50% fine sediments.	10
3. Gravel, cobble and boulder particles associated with 50 - 75% fine sediments.	5
4. Gravel, cobble and boulder particles associated with >75%% fine sediments.	1
Packing of substrate elements	
1. Assorted sizes tightly packed.	20
2. Moderately packed.	10
3. Mostly a loose arrangement.	5
4. Loose arrangement. No packing evident, all material easily moved.	1

Table 2.6 Continued.

Substrate heterogeneity	
1. Heterogeneous substrate. Bed surface often rough, usually >4 size classes (see table 2.4); no class >50% cover.	20
2. Slightly heterogeneous, bed surface with only a few large rough elements. 3-4 size classes but no size class is >50%.	10
3. Mostly homogeneous, bed surface only slightly rough/undulating, no large rough elements, 2-3 size classes, but usually dominated by > 50% of 1 class.	5
4. Homogeneous, bed surface uniform, few changes evident, only 1-2 classes	1

CHAPTER THREE

ENVIRONMENTAL FACTORS AFFECTING THE TAXONOMIC COMPOSITION OF MACROINVERTEBRATE COMMUNITIES IN CANTERBURY STREAMS

3.1 INTRODUCTION

Results from broad-scale surveys such as the '100 Rivers' project have indicated that factors related to the degree of catchment development have marked effects on macroinvertebrate community composition (Biggs et al. 1990, Harding et al. 1997). Other large-scale factors thought to influence lotic community structure include climate and geographic location (Friberg et al. 1997), geology (Charvet et al. 2000), and altitude (Jacobsen et al. 1997, Charvet et al. 2000). Important catchment to reach scale factors can include disturbance (Cobb et al. 1992), stability (Death 1991), temperature, shade, substrate size, water quality and riffle depth (e.g. Gray and Ward 1982, Ryan 1991, Collier 1995, Jacobsen et al. 1997). Refining our understanding of environmental influences on invertebrate taxonomic composition is an important component of both community ecology and impact assessment.

Previous studies have shown the structure of stream communities change along an 'ecological gradient' associated with changes in landuse (Harding and Winterbourn 1995). Generally, enrichment and pollution sensitive taxa such as Ephemeroptera, Plecoptera and Trichoptera are replaced by tolerant taxa such as those found within the non-insect groups Crustacea and Oligochaeta, and the insect family chironomidae (Diptera), as amount and intensity of catchment development increases (Quinn and Hickey 1990a, Harding and Winterbourn 1995, Quinn et al. 1997).

Broad-scale surveys such as the '100 Rivers' are useful in determining relationships at a national scale but can obscure regional patterns (Collier 1995) brought about by variations in latitude, topography and biogeography (Harding et al. 1997). A reasonable amount of literature is available on environmental factors driving invertebrate patterns at catchment and reach scales within the Canterbury region (e.g. Death 1991, Harding and Winterbourn 1995, Armstrong 1996), although few studies have looked at the region as a whole (Suren et al. 2000). The aim of the present study was to investigate the roles of catchment, riparian, and physico-chemical factors potentially affecting the taxonomic composition of invertebrate communities in a range of 3rd and 4th order streams in the Canterbury region.

3.2 METHODS

Descriptions of the study area, study sites, invertebrate sampling techniques and environmental measurement techniques are given in Chapter 2.

3.2.1 Environmental measurement scores

A number of environmental scores were developed to aid in interpreting complex environmental information. Full description of how these scores were developed are given in Chapter 2, therefore only a brief description of what they represent follows.

Habitat assessment score

These scores were designed to give an indication of overall stream and catchment condition. A site score was calculated from eight variables; landuse (which has a strong influence on the overall score), percentage of stream bank covered in vegetation, heterogeneity of banks, amount of macrophytes in the stream channel and their effect on stream flow pattern, amount of silt in the stream bed, packing of substrate materials, and substrate heterogeneity (Table 2.3). Scores ranged from 30 to 170 with high scores indicating good condition.

Algal score

This score was calculated as described in the Stream Health Monitoring Assessment Kit (SHMAK, Biggs et al. 1998). Scores ranged from 0-5 with high scores being representative of streams containing perphyton associated with 'clean' water (thin brown and black/dark brown biofilms). Low scores represented streams containing periphyton associated with 'enriched' waters (thick mats and filamentous algae).

Bank vegetation score

This score was again calculated as in SHMAK, and represent the condition of the riparian vegetation at the sampling site. Scores ranged from 0-5 with high scores indicating riparian vegetation consisting largely of native vegetation and low scores representing riparian vegetation associated with intense landuse (pasture grasses and weeds or bare ground).

Substrate score

Substrate score was calculated as in SHMAK. Scores ranged from 0-5 with higher scores being given to streambeds consisting of larger sized substrata (small cobbles, large cobbles, boulders), and sites with low scores consisting of finer substrates (sand, silt, mud).

3.2.2 Invertebrate community scores

The Macroinvertebrate Community Index (MCI) was calculated from presence-absence data following Stark (1998) as follows:

$$MCI = 20\sum a_i/S$$

where S = the total number of taxa in the sample, and a_i is the MCI score for the i^{th} taxon (Table 4.1 gives MCI scores used in this study). Scores >120 generally represent clean water while scores <80 represent probable severe degradation.

Taxonomic richness was calculated as the maximum number of taxa (family, genus, species) that were taken at each site, and number of EPT taxa is the number of Ephemeroptera, Plecoptera and Trichoptera taken at a site.

3.2.3 Statistical analyses

Sites were classified and ordinated by TWINSPAN and DECORANA on relative abundance data with the computer program PC-ORD (McCune and Mefford 1999).

TWINSPAN is based on dividing reciprocal averaging ordination space. Taxa are classified according to their abundance in site groups, and sites are arranged on the basis of differences in densities of taxa between sites. Indicator species are identified as those having the greatest differences in abundances between the groups. TWINSPAN does not analyse abundance data directly. It is based on presence/absence data; however, it approximates quantitative data by creating a variable number of 'pseudospecies' that represent abundance classes. The 'pseudospecies cut levels' are used to define the ranges of the abundance classes (McCune and Mefford 1999). The classification of sites was stopped at TWINSPAN level three and pseudospecies cut level were set at 0, 2, 5, 10 and 20. Early editions of TWINSPAN had problems with outputs being dependent on the order in which data were entered. The problem was an error in the formula, which has subsequently been corrected (McCune and Mefford 1999). DECORANA (Detrended Correspondence Analysis) arranges sites in four-dimensional space based on species similarity. Rare taxa were down-weighted using the programme's down-weighting function to reduce their influence in the ordinations.

Relationships between environmental variables, some biological characteristics and DECORANA axis scores were examined using bar graphs and Spearman rank correlation coefficients. The percentage variation explained by each DECORANA axis when displayed in two dimensions was calculated using Relative Euclidean Distance measures. A Tukey HSD test on DECORANA axis 1 scores was used to statistically test differences between the TWINSPAN river groups, and Spearman correlation coefficients were used to investigate the inter-relationships between measured environmental variables.

3.3 RESULTS

3.3.1 Invertebrate community characteristics

A total of 125 taxa (family, genus, species) were identified from the 230 sites. Aquatic insects formed the majority (99), with most belonging to the orders Diptera (25), Trichoptera (24), Ephemeroptera (11), and Plecoptera (10). Average number of taxa per site was 22.4 and the median was 22. Individual sites had from 10 to 40 taxa. Oligocheates were the most frequently occurring taxa, followed by the leptophelebid mayfly *Deleatidium*, being present at 208 and 207 of the 230 sites, respectively (Table 3.1).

Table 3.1. The 20 most frequently occurring taxa and the number of sites at which they were found (total = 230).

Taxon	Number of sites with each taxon present
Oligochaeta	208
Deleatidium	207
Orthocladiinae	202
Pycnocentrodes	197
Hydrobiosis	196
Potamopyrgus	192
Psilochorema	184
Paracalliope	176
Austrosimulium	173
Aoteapsyche	170
Olinga	159
Oxyethira	150
Ostracoda	150
Tanypodinae	148
Chironominae	143
Archichauliodes	123
Diamesinae	111
Hudsonema	108
Eriopterini	106
Pycnocentria	98

3.3.2 Classification of sites

Sites were classified into river groups by TWINSPAN using relative abundance data of the 125 invertebrate taxa. Locations of sites within the TWINSPAN groups are shown in Figure 3.1. TWINSPAN level 1 separated the sites into two main clusters; the 153 sites in river groups 1-4 and the 77 sites in river groups 5-8 (Fig. 3.2). At TWINSPAN level 2, river classes 1-4 were separated into classes 1-2 and 3-4. Indicator species diagnostic of this separation were the mayfly *Deleatidium* (1-2) and the snail *Potamopyrgus* (3-4). The indicator species identified as separating river groups 3 and 4 at TWINSPAN level 3 were the amphipod *Paracalliope* and the caddisfly *Helicopsyche* (Fig. 3.2). At TWINSPAN level 2 river groups 5-8 were separated into 5-6 and 7-8. *Paracalliope* and the caddisfly *Pycnocentrodes* were identified as indicator species separating river groups 5 and

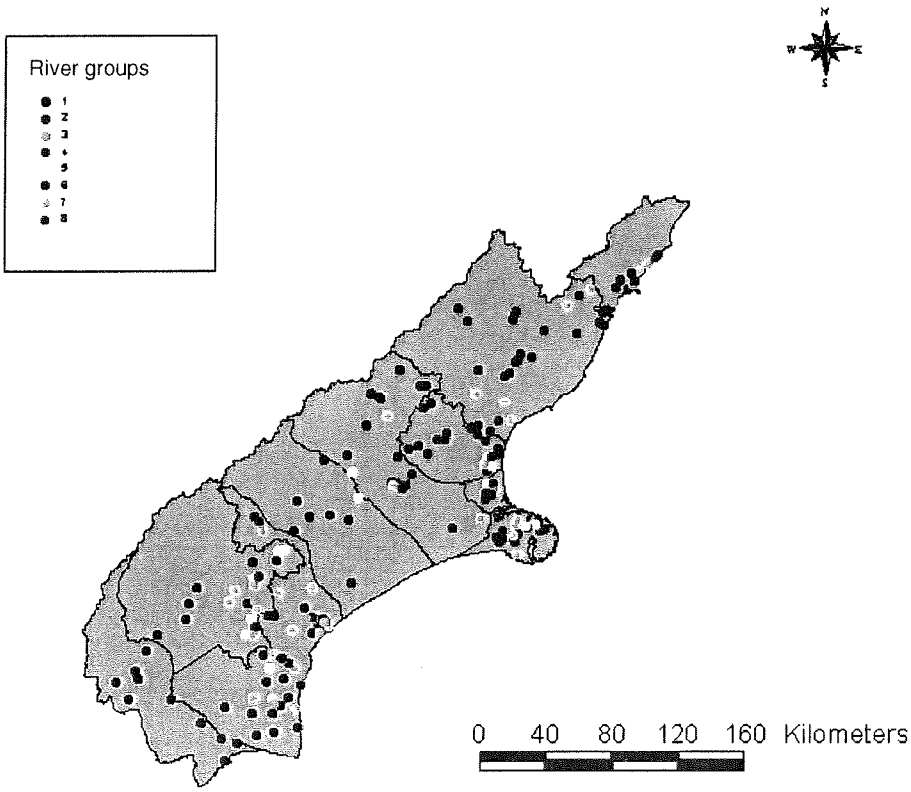


Fig. 3.1 Location of the TWINSPAN river groups in the Canterbury region.

6 and the caddisfly *Oxyethira* was identified as separating classes 7 and 8. At TWINSPAN level 3, river groups 5 and 6 were separated with caddisfly genera *Pycnocentrodes* and *Oecetis* being the diagnostic indicator species. *Potamopyrgus* and Ostracoda were indicator species diagnostic of the separation of river classes 7 and 8.

The dichotomy at TWINSPAN level 1 appears to separate 'cleaner water' faunas (river groups 1-4) from 'moderately degraded water' faunas (river groups 5-8). Groups 1-4 had high relative abundances of ephemeropterans and trichopterans, with dipterans, chironomids, molluscs and oligochaetes also usually present (Fig. 3.3). Groups 5-8 differed from 1-4 in that they had low relative abundances of Ephemeroptera and generally higher abundances of oligochaetes, molluscs, crustaceans and chironomids.

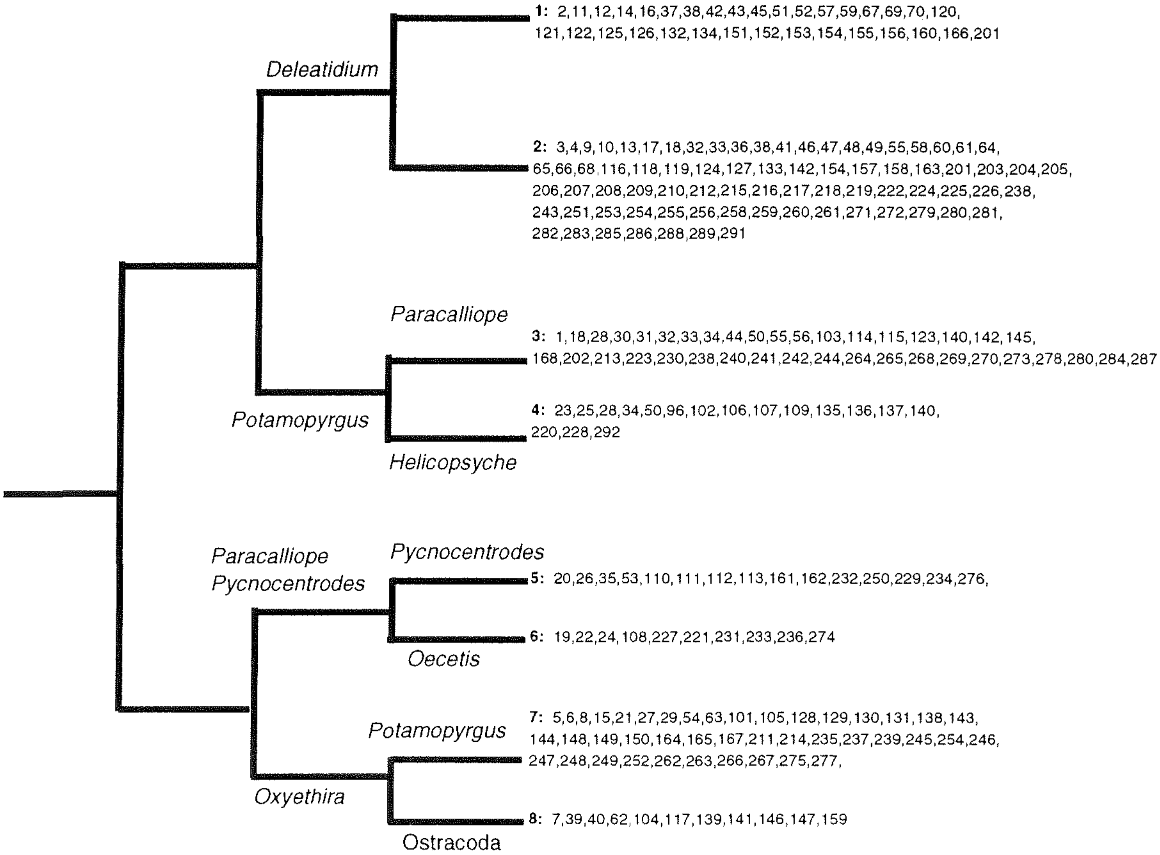


Fig. 3.2 TWINSPAN dendrogram showing sites within each river group and taxa diagnostic of the separations.

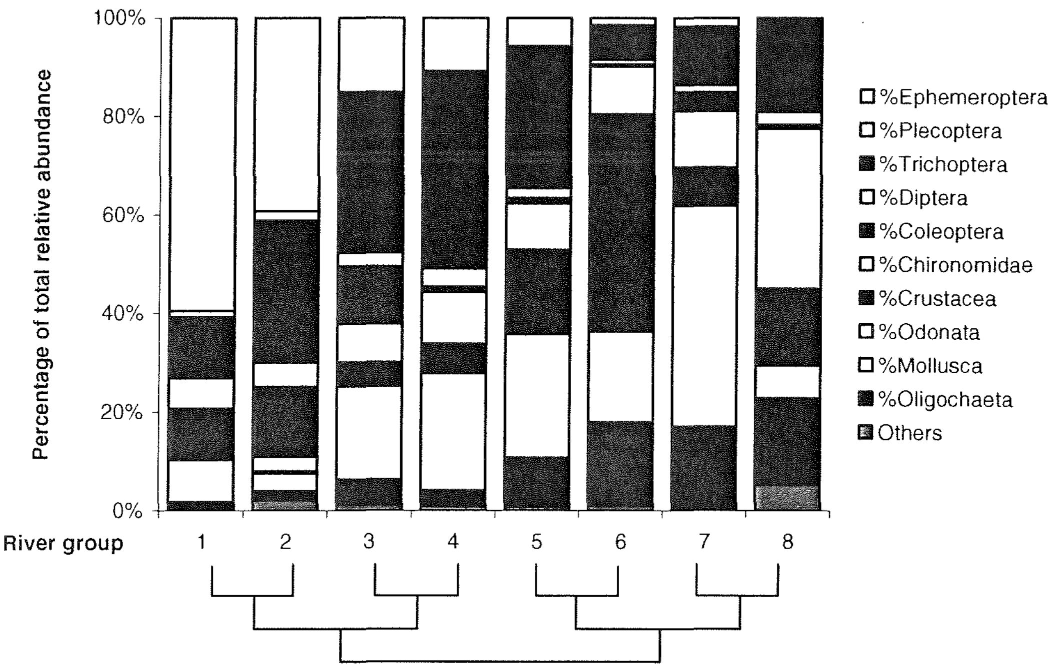


Fig. 3.3 Mean percent contribution of major invertebrate groups to the eight TWINSPAN river groups.

At TWINSPAN level 2 there was a further separation (groups 1-2 from 3-4), again into 'clean water' faunas dominated by Ephemeroptera and with Plecoptera present (groups 1-2) and 'mildly degraded water' fauna. This latter group still had high relative abundance of Ephemeroptera and Trichoptera, but also relatively high abundances of molluscs and oligochaetes, and chironomids and crustaceans present (groups 3-4; Fig. 3.3).

The moderately degraded river groups 5-8 were separated into 'cleaner water' faunas with high relative abundances of crustaceans and molluscs (groups 5-6), and 'degraded water' faunas with low relative abundance of Ephemeroptera and high proportions of chironomids, crustaceans, molluscs and oligochaetes (groups 7-8; Fig. 3.3).

3.3.3 TWINSPAN level 3 divisions

River group 1

Ephemeroptera (mainly *Deleatidium*) strongly dominated this group of 32 sites, with Plecoptera also present (Fig. 3.3). These sites were found at relatively high altitudes, had relatively low conductivity, high bank vegetation scores and habitat assessment scores, high MCI values and relatively low taxonomic richness (Fig. 3.4, Table 3.2).

River group 2

This was the largest river group, consisting of 65 sites. The group was dominated by Ephemeroptera and Trichoptera, with Plecoptera also present. Non-insects were common, but each one contributed to only a small proportion of total relative abundance (Fig. 3.3). Streams were generally found at relatively high altitudes, were fairly wide, and had high velocities. Conductivity was low and clarity was high. Sites within this group had high bank vegetation scores and habitat assessment scores, high MCI scores, high numbers of EPT taxa and high taxonomic richness (Fig. 3.4).

River group 3

These 40 sites had moderate relative abundances of Ephemeroptera and Trichoptera, Coleoptera and Mollusca. This group also had crustaceans present and higher relative abundances of oligochaetes than groups 1 and 2 (Fig. 3.3).

Table 3.2 Mean values for catchment, riparian, physico-chemical and invertebrate community descriptors for the main TWINSPAN groups. The number of sites in each group is indicated in parentheses. Standard errors are given in Figure 3.4.

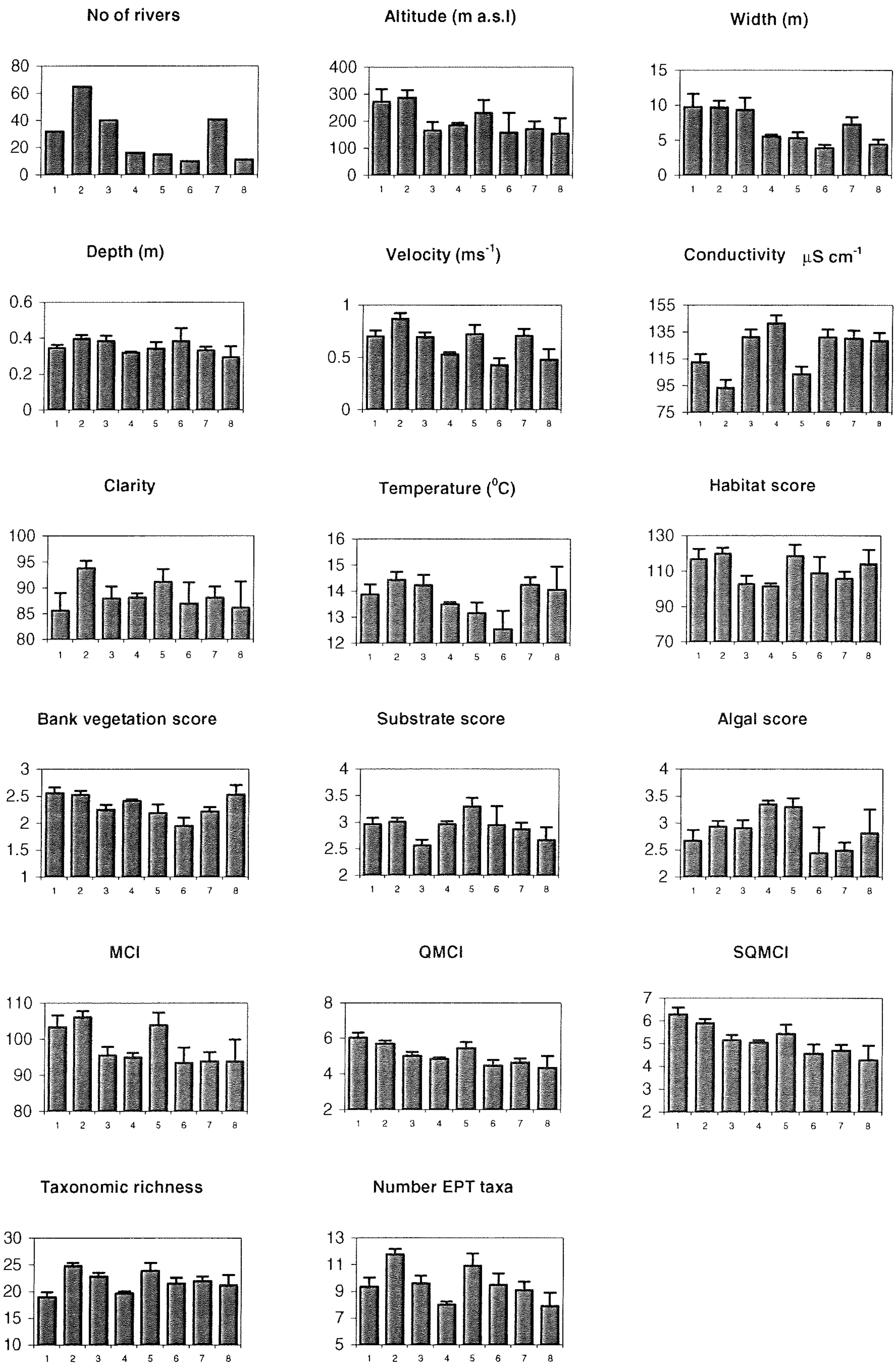
	River group							
	1 (32)	2 (65)	3 (40)	4 (16)	5 (15)	6 (10)	7 (41)	8 (11)
Catchment/riparian descriptors								
Habitat score	116.8	119.9	102.8	101.6	118.7	108.9	105.8	114.0
Bank vegetation score	2.6	2.5	2.2	2.4	2.2	1.9	2.2	2.5
Physico-chemical descriptors								
Altitude (m a.s.l)	271.5	287.4	165.0	184.7	230.2	157.5	170.7	153.9
Width (m)	9.7	9.6	9.3	5.6	5.4	3.9	7.2	4.4
Depth (m)	0.3	0.4	0.4	0.3	0.3	0.4	0.3	0.3
Velocity (mS^{-1})	0.7	0.9	0.7	0.5	0.7	0.4	0.7	0.5
Substrate score	3.0	3.0	2.6	3.0	3.3	2.9	2.9	2.7
Algal score	2.7	2.9	2.9	3.3	3.3	2.4	2.5	2.8
Conductivity ($\mu S\ cm^{-1}$)	112.8	93.6	131.2	141.7	103.4	131.0	130.2	128.7
Clarity (cm^{-1})	85.6	93.8	87.9	88.1	91.2	86.9	88.0	86.2
Spot water temperature (^0C)	13.9	14.4	14.2	13.5	13.2	12.5	14.2	14.1
Invertebrate community indices								
MCI	103.4	106.1	95.5	95.0	103.9	93.5	93.9	93.8
QMCI	6.0	5.7	5.0	4.8	5.5	4.5	4.6	4.4
SQMCI	6.3	5.9	5.2	5.1	5.4	4.6	4.7	4.3
Taxonomic richness	19.0	24.8	22.8	19.6	23.9	21.5	21.9	21.2
Number of EPT taxa	9.3	11.8	9.6	8.0	10.9	9.5	9.1	7.9

Sites in group 3 generally occurred at lower altitudes than those in groups 1 and 2, but had comparable widths. Conductivity was reasonably high, substrate scores were low, and MCI scores were moderately low (Fig. 3.4).

River group 4

These 16 sites had a similar overall taxonomic composition to group 3, but with high contributions by Mollusca, Chironomidae and Trichoptera, and low contributions by Ephemeroptera and Coleoptera (Fig. 3.3). Group 4 streams were found at intermediate altitudes, and had slower velocities than streams in groups 1-3. Conductivities in this group were relatively high and clarities low. Algal scores were moderately high and habitat assessment scores were low. MCI scores, taxonomic richness and number of EPT taxa were all quite low (Fig. 3.4).

Fig. 3.4 Mean values (+1 SE) of catchment, riparian, physico-chemical and invertebrate community descriptors for the 8 TWINSPAN river groups.



River group 5

The 15 sites in this group had similar overall taxonomic composition at the order level to groups 3 and 4, but with higher relative abundances of Crustacea and Oligochaeta, and lower relative abundances of Ephemeroptera (Fig. 3.3). Conductivity and water temperatures in this group were relatively low, whereas substrate, algal and habitat assessment scores were all quite high. MCI, taxonomic richness, and number of EPT taxa were also high (Fig. 3.4).

River group 6

This was the smallest river group, consisting of 10 sites. The group was dominated by Crustacea and had few Ephemeroptera and Trichoptera (Fig. 3.3). Group 6 streams were at low altitudes and generally were narrow widths and slow flowing. Conductivities were relatively high, and temperatures were low. Bank vegetation and algal scores were low, as were MCI scores (Fig. 3.4).

River group 7

The faunas of these 41 sites were dominated by Mollusca, but had moderate relative abundances of oligochaetes, chrionomids and trichopterans (Fig. 3.3). These sites had relatively high velocities, conductivities, and water temperatures. Algal and habitat assessment scores were low at this group of sites (Fig. 3.4).

River group 8

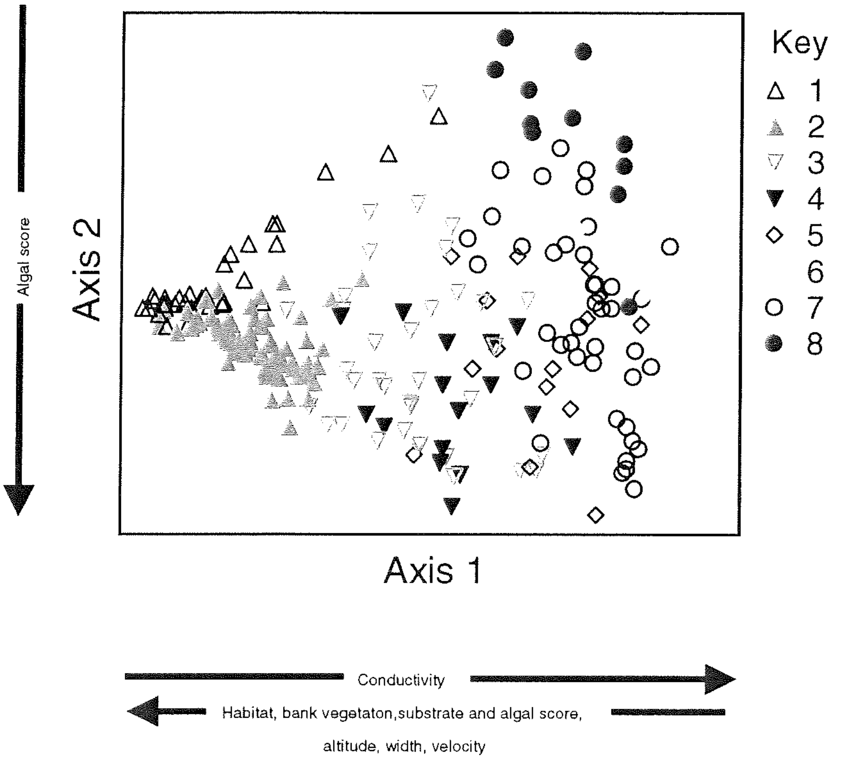
These 11 sites, like those in groups 6 and 7, had low abundances of Ephemeroptera. Chironomidae made up the largest taxonomic group, while Trichoptera, Crustacea and Oligochaeta were present in similar but lower proportions (Fig. 3.3). Few EPT taxa were present. These sites were at low altitudes, were relatively narrow, had fairly high conductivities and low clarity. Substrate scores were low, whereas bank vegetation and habitat scores were high (Fig. 3.4).

3.3.4 *Associations with environmental variables*

Ordination of sites by DECORANA using relative abundances of invertebrate taxa allowed the importance of measured environmental factors as drivers of invertebrate patterns to be investigated (Fig. 3.5). Axis 1 scores explained 48.3% of the variation in the data set, axis 2, 14.1% and axis 3, 10.2%. DECORANA axis 1 was positively correlated with conductivity, and negatively correlated with habitat assessment score, bank vegetation score, altitude, width, velocity, substrate and algal score (Table 3.3). Axis 2 was correlated most strongly with algal score. Axis 3 was correlated most strongly with habitat assessment score and algal score (positively) and with conductivity and water temperature (negatively; Table 3.3).

Figure 3.5 shows there was extensive overlap between river groups, particularly those in river groups 3, 4, 5 and 7. However, a post-hoc comparison of means (Tukey HSD test) based on DECORANA axis 1 scores showed that river group 1 was separated significantly further to the left of axis 1 than groups 3, 4, 6, 7 and 8. Similarly, river group 2 was significantly further to the left than groups 3, 6 and 7 (Table 3.4). Environmental variables, which are often associated with agricultural development, were identified as being important in driving invertebrate patterns (Fig. 3.5).

Fig. 3.5 DECORANA plots of sites based on relative abundances of invertebrate taxa, with TWINSPAN river groups and significant relationships with environmental variables (See Table 3.3) shown.



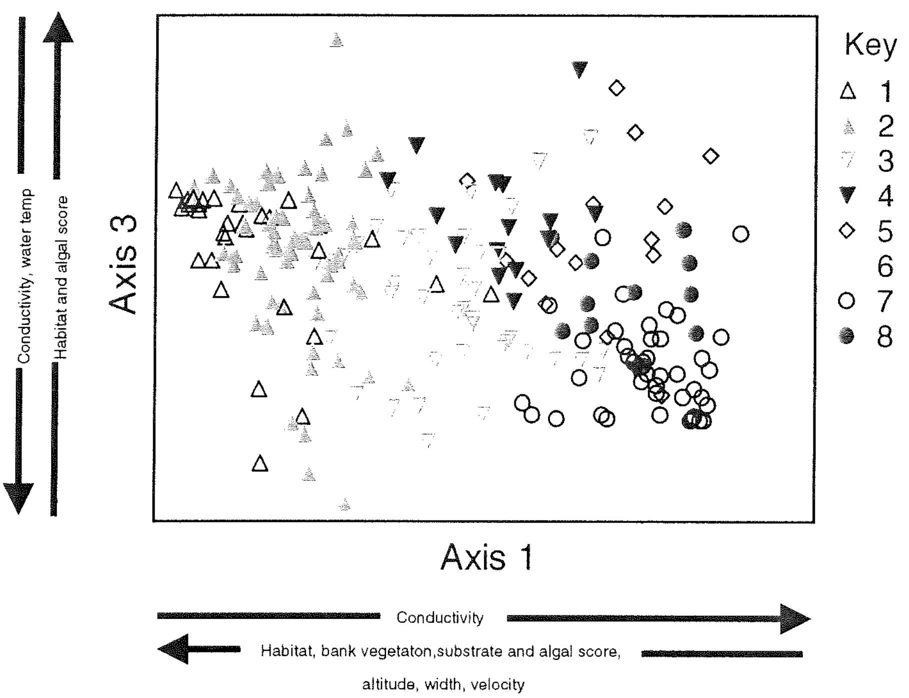


Table 3.3 Spearman rank correlations between measured environmental factors and DECORANA axis scores. n.s., not significant; *, P<0.05; **, P<0.01; ***, P<0.001.

	Axis 1	Axis 2	Axis 3
Catchment/riparian descriptors			
Habitat score	-0.54***	n.s.	0.24***
Bank vegetation score	-0.61***	n.s.	0.16*
Physico-chemical descriptors			
Altitude (m a.s.l)	-0.58***	n.s.	0.17*
Width (m)	-0.33***	n.s.	n.s.
Depth (m)	n.s.	-0.15*	n.s.
Velocity (mS^{-1})	-0.51***	n.s.	n.s.
Substrate score	-0.41***	n.s.	0.21**
Algal score	-0.35***	-0.29***	0.23***
Conductivity (μScm^{-1})	0.51***	0.17*	-0.24***
Clarity	n.s.	n.s.	0.14*
Spot water temperature (^{0}C)	n.s.	n.s.	-0.40***

Table 3.4 Post-hoc comparison of means based on DECORANA axis 1 scores (Tukey HSD test) between the eight TWINSPAN river groups. n.s., not significant; *, P<0.05; **, P<0.01; ***, P<0.001.

	1	2	3	4	5	6	7	8
1								
2	n.s.							
3	**	*						
4	*	n.s.	n.s.					
5	n.s.	n.s.	n.s.	n.s.				
6	**	*	n.s.	n.s.	n.s.			
7	***	**	n.s.	n.s.	n.s.	n.s.		
8	*	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	

3.3.5 Inter-relationships among environmental variables

Inter-relationships among measured environmental variables were investigated using rank correlation analysis (Table 3.5). High conductivity was associated with low clarity, low substrate, bank vegetation and habitat assessment scores, low altitude, narrow channel width and low water velocity. High clarity streams tended to have high substrate, bank vegetation and habitat assessment scores, were found at high altitudes and were generally wide. pH was significantly correlated with temperature and substrate scores. Higher temperatures were associated with low substrate scores, low habitat assessment scores and large wide streams. Substrate scores were positively associated with bank vegetation scores, algal scores, and especially habitat assessment scores. High substrate scores generally occurred at higher altitudes and in shallower streams (Table 3.5). Bank vegetation scores were correlated positively with habitat assessment scores, altitude, width and velocity. Both algal scores and habitat assessment scores were correlated significantly with altitude, while depth and width were associated with increasing velocity (Table 3.5).

Table 3.5 Spearman rank correlation coefficients between measured environmental factors and scores. n.s., not significant; *, $P<0.05$; **, $P<0.01$; ***, $P<0.001$.

	Conductivity (μScm^{-1})	Clarity	pH	Temp (^0C)	Substrate score	Bank vegetation score	Algal score	Habitat assessment score	Altitude (m a.s.l)	Average width (m)	Average depth (m)
Clarity	-0.33***										
pH	n.s.	n.s.									
Temp (^0C)	0.17*	n.s.	0.28***								
Substrate score	-0.38***	0.23***	0.23***	-0.20**							
Bank vegetation score	-0.31***	0.18**	n.s.	n.s.	0.28***						
Algal score	-0.27***	n.s.	n.s.	-0.15*	0.23**	n.s.					
Habitat assessment score	-0.43***	0.31***	0.20**	-0.28***	0.73***	0.48***	0.18**				
Altitude (m a.s.l)	-0.68***	0.23***	n.s.	-0.18*	0.46***	0.47***	0.22**	0.51***			
Average width (m)	-0.25***	0.21**	0.18**	0.22**	n.s.	0.31***	n.s.	n.s.	0.14*		
Average depth (m)	n.s.	n.s.	n.s.	n.s.	-0.18**	n.s.	n.s.	-0.20**	n.s.	0.41***	
Average velocity (mS^{-1})	-0.48***	n.s.	n.s.	n.s.	n.s.	0.25***	0.16*	n.s.	0.31***	0.41***	0.25***

3.4 DISCUSSION

The primary objective of this chapter was to investigate which environmental factors were affecting the taxonomic composition of aquatic macroinvertebrate communities in the Canterbury region. The TWINSPAN classification of sites separated sites initially into 'clean water' and 'degraded water' groups based on their macroinvertebrate faunas. Ordinations and correlations with environmental variables revealed that conductivity, altitude, width, velocity, temperature and habitat assessment, bank vegetation, substrate and algal scores were important factors associated with the taxonomic composition of invertebrate communities, and that many of measured environmental variables were inter-related.

3.4.1 Invertebrate community characteristics

The taxonomic diversity of the 230 Canterbury streams (125) was higher than that recorded at 29 Northland sites (84,Collier 1995), at 26 North Westland sites (83,Collier 1989), at 45 acid brown water Westland streams (90,Winterbourn and Collier 1987) and 43 North and South Island forested streams (61,Rounick and Winterbourn 1982), sampled using comparable procedures. The stream invertebrate community in Canterbury was dominated by insect taxa (79.2%), consistent with other New Zealand surveys that have employed similar levels of taxonomic identification. For example, Harding (1990) found that 85% of the fauna at 3 Westland sites consisted of insects, and Brown (1998) found that 89.6% of the fauna at the Cobb River, Takaka was insects. Diptera and Trichoptera were the most taxonomically diverse orders (25 and 24, respectively). While many other surveys have found Trichoptera to dominate (e.g. Harding 1990, Collier 1995, Brown 1998) fewer have found Diptera to be the most taxonomically diverse order (Friberg et al. 1997). As in many streams around the country *Deleatidium* was the most frequently occurring insect genus (Winterbourn and Collier 1987, Harding 1994, Dewdney 2000).

DECORANA ordinations showed that while some of the river groups identified were relatively tight, in general the sites fell on a continuum of environmental gradients (Fig. 3.5). Such a continuum was also apparent for relative abundance of Ephemeroptera (Fig. 3.3). This change in relative abundance of Ephemeroptera is striking and appeared to be associated with increasing intensity of landuse. The reduction in relative abundance of mayflies was complemented by increases in non-insect taxa, notably the molluscs and oligochaetes. The occurrence of declining ephemeropteran relative abundance with degree of development/enrichment is consistent with the results of several other studies (Harding and Winterbourn 1995, Quinn et al. 1997, Harding et al. 1999) and supports the suggestion of Suren et al. (2000) that mayflies may be a useful group for separating streams into management units. Furthermore, the absence of plecopterans from river groups 3-8 is generally consistent with these sites coming from developed catchments with warmer temperatures and reduced riparian cover (Quinn and Hickey 1990a, Harding and Winterbourn 1995). Trichopterans were poorly represented in river groups 6-8, although as found by Harding and Winterbourn (1995), *Oxyethira* was the TWINSPAN indicator species responsible for separating degraded streams. This genus however, is commonly associated with enriched streams (Quinn and Hickey 1990b, Quinn et al. 1997). These findings support the view that the structure of stream communities changes along an 'ecological gradient' associated with changes in landuse (Harding and Winterbourn 1995). Important factors associated with these changes in community composition include pollution and eutrophication (Suckling 1982), channelisation (Quinn et al. 1992, Quinn et al. 1997), riparian vegetation (Quinn et al. 1997, Maddock 1999), and increased sedimentation (Lenat et al. 1981, Quinn et al. 1992, Collier and Smith 1998).

The locations of TWINSPAN river groups within the Canterbury region (Fig. 3.1) suggests that invertebrate faunas of 3rd and 4th order streams, based on relative abundances, cannot easily be grouped into the 4 Canterbury 'Ecoregions' designated by Harding (1994). The groups appear to fit more closely into the River Environment Classification (REC) classes proposed by Suren et al. (2000), which is based on source of flow, geology and landcover.

3.4.2 Environmental factors affecting taxonomic composition of invertebrate communities

Both bank vegetation and habitat assessment scores were identified as important catchment/riparian descriptors affecting species composition (Table 3.3). Habitat assessment scores were designed to indicate overall habitat condition, and as such, catchment landuse was a major contributing factor. Land development can increase water temperature and degree of enrichment (Quinn and Hickey 1990a), change channel morphology (Davies-Colley 1997), increase suspended solids and fine sediments and reduce amounts of coarse woody debris in streams (Quinn and Hickey 1990a), as well as change light levels and increase flow variability (Quinn and Hickey 1990b). These factors associated with landuse have been found by others to affect invertebrate community structure (e.g. Scott et al. 1994, Suren 1994, Tate and Heiny 1995, Carter et al. 1996, Allan et al. 1997, Townsend et al. 1997, Bis et al. 2000).

The significant correlation between axis 1 scores and bank vegetation scores (r = -0.61) indicate that factors associated with riparian vegetation have important influences on invertebrate communities in Canterbury streams. This finding is generally consistent with previous observations that streams with forested riparian zones have higher taxonomic richness and abundance of 'clean water' faunas than comparable sized streams with riparian vegetation consisting of agricultural grasses (Harding and Winterbourn 1995, Bis et al. 2000). It suggests further that the management of riparian vegetation should be an effective tool for improving stream conditions. Healthy functional riparian vegetation provides three important benefits to streams and therefore the invertebrates within them. Firstly, they reduce or buffer the impact of land-based processes on waterways by slowing down and reducing inputs from surface and subsurface flows. For example, they can filter nutrients, soil, microbes and pesticides, denitrify groundwater, and use nutrients for plant growth before they enter the stream. Secondly, riparian vegetation can reduce or buffer the impact of water-borne processes on adjacent land by protecting banks from erosion and floods. Thirdly, they promote and sustain instream plants and animals by reducing fine sediment levels, maintaining water clarity, providing instream food supplies (plant litter) and habitat, preventing nuisance plant growths, maintaining lower summer water temperatures, reducing light levels and, maintaining natural food webs (MacGibbon 2000).

Of the physico-chemical descriptors, altitude, conductivity and velocity appeared to have the greatest influence on invertebrate community structure (Table 3.3). Generally, higher relative abundances of Ephemeroptera and Trichoptera and the presence of Plecoptera occurred at higher altitudes, and a greater relative abundance of taxa such as molluscs, oligochaetes and chironomids at lower altitudes. The relationship between altitude and taxonomic composition is consistent with that found in several other studies (Jacobsen et al. 1997, Charvet et al. 2000) and is likely to reflect, to a certain extent, the relative ease with which lower gradient, lower altitude land has been developed for forestry and agriculture (Carter et al. 1996)..

Conductivity has been considered a surrogate for nutrient loading in New Zealand (Biggs et al. 1990) and was found by Harding et al. (1999) to be significantly correlated with agricultural intensity. Sites with higher conductivities had lower relative abundances of EPT fauna and higher abundances of 'degraded water' faunas (Fig. 3.3, 3.4) concurring with this pattern.

Other physico-chemical descriptors found to be associated with invertebrate community composition included substrate score, water temperature, algal score and width. The substrate type from which samples were collected appeared to have influence on the taxonomic composition, with greater relative abundances of the Ephemeroptera and Trichoptera generally occurring on larger substrates (higher substrate scores; Fig. 3.3, 3.4). Sediment additions and deposition can change the character of the substrate, block interstices, and reduce interstitial volume, which may have a variety of effects on invertebrates ranging from minor interference with feeding through to death by smothering (Ryan 1991). Additionally, as larger stones are usually more stable than smaller ones (Death and Winterbourn 1994), relationships between taxonomic composition and substrate size are likely to have a stability component (Death 2000). The changes in invertebrate community structure from one dominated by Ephemeroptera and Trichoptera to one primarily comprised of Dipterans and non-insect taxa, with changes in substrate size are consistent with those of other studies (Chutter 1969, Lenat et al. 1981, Gray and Ward 1982, Quinn and Hickey 1990b, Ryan 1991, Cobb et al. 1992, Wohl et al. 1995, Angradi 1999).

Temperature is also known to be an important mechanism affecting growth and distribution of stream insects (Quinn et al. 1994, Hawkins et al. 1997) and was implicated as an important variable affecting taxonomic composition at a site in the present study. Hawkins et al. (1997) suggested that temperature can influence assemblage structure by affecting developmental rates of individual taxa and overall assemblage phenology, and also by excluding taxa directly. In the present study Plecoptera and Ephemeroptera to a lesser extent, were only found in groups that were associated with cooler streams suggesting either or both of the aforementioned factors may affect these groups. Plecoptera and Ephemeroptera appear to be largely restricted to rivers with summer temperatures typically below 19^{0}C and 21.5^{0}C respectively (Quinn and Hickey 1990a, Quinn et al. 1994). In contrast, mollusca and coleoptera were often associated with warmer streams and previous studies suggest that species within this group have relatively high upper thermal tolerances (Quinn et al. 1994).

The nature of the algal community as indicated by algal score was identified as an environmental factor associated with invertebrate communities in Canterbury streams. High scores are indicative of algae associated with enrichment such as thick mats and long filaments (Biggs et al. 1998). Two processes could account for algae influencing invertebrate communities, either individually or in combination (Quinn and Hickey 1990a). Firstly, if higher nutrient concentrations, temperatures and light levels are responsible for the build up of algae, the amount of relatively clean substrate preferred by some taxa will be reduced. Secondly, the presence of algae associated with enrichment may be determined by other factors (e.g. flow variability, higher temperature and higher sediment inputs) that reduce the abundance of intolerant grazing invertebrates in rivers with more developed catchments, resulting in a reduction in grazing pressure and a build up of periphyton (Quinn and Hickey 1990a). Algal scores were significantly correlated (negatively, Table 3.5) with conductivity suggesting that enrichment in Canterbury streams affects algal composition. Typically 'clean water' invertebrate taxa that are adapted to browsing thin biofilms on relatively clean surfaces (Ephemeroptera) were absent from river groups with high algal scores. In contrast, invertebrates that can burrow or creep over prolific algal growth such as *Potamopyrgus* and chironomids were abundant, thus supporting the first process above. Additionally, factors such as temperature and substrate composition influenced intolerant grazers (Ephemeroptera in particular), which would reduce grazing pressure and result in a build up of periphyton. Both these factors in combination are likely to influence invertebrate and algal composition.

Finally, stream width was found to be associated with taxonomic composition (Fig. 3.5). Studies elsewhere have shown where land and riparian vegetation have been cleared, stream channels are often narrower (Davies-Colley 1997, Bunn et al. 1998). Loss of benthic habitat will therefore affect stream communities with a change to a community dominated by non-insect orders and chironomids in narrow streams with high sediment loads (Figs. 3.3, 3.5). The association found is probably linked with agricultural development, the clearing of land and vast increases in sediment deposition that occur with intense landuse (Waters 1995).

3.4.3 Conclusions

This study has provided a broad perspective on the invertebrate communities in the Canterbury region, and some of the more important environmental factors affecting their distribution patterns. Results indicate that altitude is an important large scale 'driving force', while habitat, bank vegetation, substrate, algae conductivity, width, velocity and temperature are important catchment to reach scale 'forces' associated with macroinvertebrate community composition. Although these variables all have complex inter-relationships (Table 3.5) it appears that overall patterns are influenced by two overriding factors; physical condition of the stream and amount of anthropogenic development. Altitude, velocity and width were identified as important variables affecting the invertebrate community, indicating that the physical structure of Canterbury streams influences the taxa that inhabit them. Additionally, factors associated with intensive landuse are likely to drive invertebrate community composition. For example, anthropogenic development is greater at lower altitudes where land is more easily converted for agriculture or forestry. This modification of land reduces bank vegetation, increases sedimentation and therefore alters streambed conditions. Algal proliferations can occur through nutrient additions and reduced grazing pressure, and channel alterations affect width, velocity and temperature. Finally, the nature and extent of development are reflected in the fauna of streams, depending where on the ecological gradient they fall. Thus, in Canterbury streams, fauna dominated by Ephemeroptera and Trichoptera and with some Plecoptera present occur at pristine high altitude sites but are replaced by faunas dominated by non-insect taxa (crustaceans, oligochaetes) and Chironomidae at degraded lowland sites (Fig. 3.6). Between these two extremes fauna change in a continuum where Trichoptera dominate intermediately disturbed streams although the other groups are present in reasonable high abundances. The most important factor identified for these changes in community composition in Canterbury was the type and extent of riparian vegetation at a site.

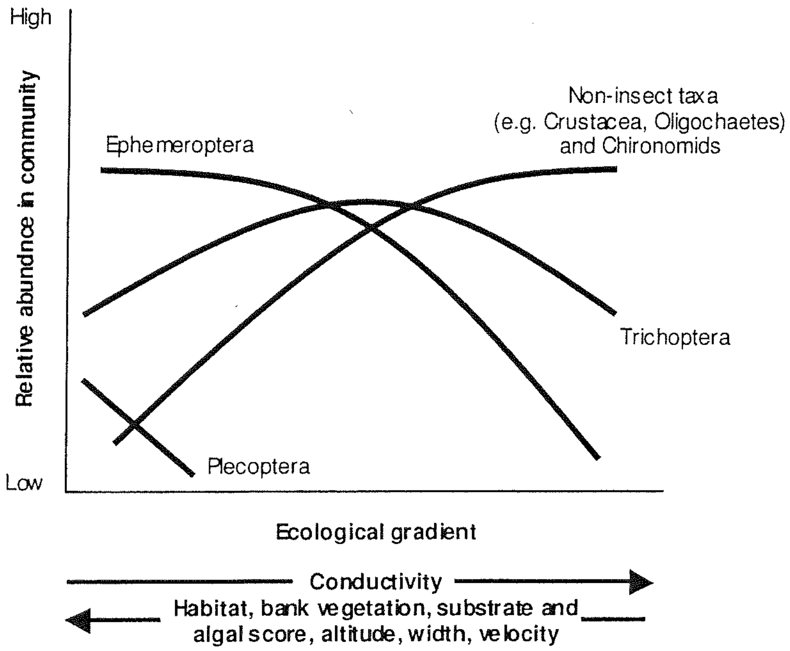


Fig. 3.6 Conceptual diagram of ways in which macroinvertebrate groups respond to 'ecological gradients'.

The extent of damage to stream health by anthropogenic development is often assessed with biotic indices, and in particular in New Zealand the Macroinvertebrate Community Index (MCI), the quantitative MCI (QMCI) and the semi-quantitative MCI (SQMCI). In the following chapter, I compare results obtained from Canterbury streams using these three indices.

CHAPTER FOUR

A COMPARISON OF MCI, QMCI AND SQMCI FOR THE ASSESSMENT OF CANTERBURY STREAM HEALTH

4.1 INTRODUCTION

The Macroinvertebrate Community Index (MCI) and its derivatives (QMCI, SQMCI) are biotic indices (single numbers that summarise complex biological data) used to interpret the health and condition of streams and rivers. They are used widely in New Zealand by regional councils, consultants, and researchers.

Stark (1985) developed the Macroinvertebrate Community Index and its quantitative version (QMCI) for the detection of organic enrichment in stony-bottomed streams. Later, a semi-quantitative variant (SQMCI) was developed to reduce sampling and processing effort (Stark 1998). These biotic indices are based on the British Biological Monitoring Working Party Score System (BMWP) and rely on allocation of scores (1-10) to taxa (usually genera) of freshwater macroinvertebrates based on their pollution tolerances (Stark 1985). High scores are characteristic of pristine conditions, whereas low scores are characteristic of degraded conditions. MCI values are given in Table 4.1. A preliminary version of the MCI was developed for Taranaki ringplain streams and has been modified to incorporate new species where necessary, by professional judgment to suit other regions throughout New Zealand.

MCI values are calculated from macroinvertebrate presence-absence data, whereas the QMCI uses quantitative macroinvertebrate data and SQMCI uses coded abundances of taxa based on a Rare/Common/Abundant/Very Abundant/Very Very Abundant scale (Table 4.2). In theory, MCI values can range from 0-200, whereas QMCI and SQMCI values can range from 0-10.

Table 4.1 Taxon scores used in calculating MCI, QMCI and SQMCI scores.

INSECTA		**Coleoptera** (cont.)		**Trichoptera** (cont.)	
Ephemeroptera		Staphylinidae	5	*Hydrochorema*	9
Ameletopis	10	**Diptera**		*Kokiria*	9
Arachnocolus	8	Anthomyiidae	3	*Neurochorema*	6
Atalophlebioides	9	*Aphrophila*	5	*Oeconesidae*	9
Austroclima	9	*Austrosimulium*	3	*Olinga*	9
Coloburiscus	9	*Calopsectra*	4	*Orthopsyche*	9
Deleatidium	8	Ceratopogonidae	3	*Oxyethira*	2
Ichthybotus	8	*Chironomus*	1	*Paroxythira*	2
Isothraulus	8	*Cryptochironomus*	3	*Philorheithrus*	8
Mauiulus	5	*Culex*	3	*Plectrocnemia*	8
Neozephlebia	7	Empididae	3	*Polyplectropus*	8
Nesameletus	9	Ephydridae	4	*Psilochorema*	8
Oniscigaster	10	Eriopterini	9	*Pycnocentrella*	9
Rallidens	9	*Harrisius*	6	*Pycnocentria*	7
Siphlaenigma	9	Hexatomini	5	*Pycnocentrodes*	5
Zephlebia	7	*Limonia*	6	*Rakiura*	10
Plecoptera		*Lobodiamesa*	5	*Tiphobiosis*	6
Acroperla	5	*Maoridiamesa*	3	*Triplectides*	5
Austroperla	9	*Microchorista*	4	*Zelolessica*	10
Cristaperla	8	*Mischoderus*	4	**Lepidoptera**	
Halticoperla	8	*Molophilus*	5	*Hygraula*	4
Megaleptoperla	9	*Neocurupira*	7	**Collembola**	6
Spaniocercoides	8	*Orthocladiinae*	2	**ACARINA**	5
Stenoperla	10	*Parochlus*	8	**CRUSTACEA**	
Zelandobius	5	*Paradixa*	4	Amphipoda	5
Zelandoperla	10	*Paralimnophila*	6	Copepoda	5
Megaloptera		*Paucispinigera*	6	Cladocera	5
Archichauliodes	7	*Peritheates*	7	Isopoda	5
Odonata		*Podonominae*	8	Ostrocoda	3
Aeshna	5	*Polypedilum*	3	*Paranephrops*	5
Antipodochlora	6	Psychodidae	1	*Paratya*	5
Austrolestes	6	Sciomyzidae	3	Tanaidacea	4
Hemicordulia	5	Stratiomyidae	5	**MOLLUSCA**	
Xanthocnemis	5	Syrphidae	1	*Ferrissia*	3
Procordulia	6	Tabanidae	3	*Gyraulus*	3
Hemiptera		Tanypodinae	5	*Latia*	3
Diaprepocoris	5	Tanytarsini	3	*Lymnaeidae*	3
Microvelia	5	*Tanytarsus*	3	*Melanopsis*	3
Sigara	5	*Zelandoptipula*	6	*Physa*	3
Coleoptera		**Trichoptera**		*Glyptophysa*	5
Antiporus	5	*Aoteapsyche*	4	*Potamopyrgus*	4
Berosus	5	*Beraeoptera*	8	Sphaeriidae	3
Dytiscus	5	*Confluens*	5	**OLIGOCHAETA**	1
Elmidae	6	*Conuxia*	8	**HIRUDINEA**	3
Hydraenidae	8	*Costachorema*	7	**PLATYHELMINTHES**	3
Hydrophilidae	5	*Ecnominidae*	8	**NEMATODA**	3
Liodessus	5	*Helicopsyche*	10	**NEMATOMORPHA**	3
Ptilodactylidae	8	*Hudsonema*	6	**NEMERTEA**	3
Rhantus	5	*Hydrobiosella*	9	**COELENTERATA**	
Scirtidae	8	*Hydrobiosis*	5	*Hydra*	3

Table 4.2 MCI, QMCI, SQMCI and OQMCI indices and their formulae.

Biotic Index	Formulae	Definitions of terms
MCI (Stark 1985)	$20 \sum a_i / S$	a_i = MCI tolerance score for the i^{th} taxon n_i = the number of individuals in the i^{th} taxon S = total number of taxa
QMCI (Stark 1985)	$\sum (n_i a_i)/N$	N = total number of individuals c_i = coded abundance for the ith scoring taxon[1] M = total coded abundances
SQMCI (Stark 1998)	$\sum (c_i a_i)/M$	o_i = the number of individuals in the ith order b_i = MCI toterance score for the ith order[2]
OQMCI (Present chapter)	$\sum o_i b_i / N$	

[1], Values of c_i are coded as follows (from Stark 1998): c_i = 1 if n_i is between 0 and 1%, ci = 5 if n_i is between 1.01 and 5%, c_i = 20 if n_i is between 5.01 and 20%, c_i = 100 if n_i is between 20.01 and 50%, and c_i = 500 if n_i is greater than 50% of the sample.
[2], Values of *bi* are average tolerance scores for: Ephemeroptera, 8.3; Plecoptera, 7.9; Trichoptera, 7.2; Diptera, 4.4; Coleoptera, 5.9; Crustacea, 4.6; Odonata, 5.5; Mollusca, 3.3; Oligochaeta,1.0; Lepidoptera, 4.0; Megaloptera, 7.0; Mecoptera, 7.0; Acarina, 5.0; Hemiptera, 5.0; Other worms (Nematoda, Platyhelminthes, Hirudinea), 3.0.

Although a number of studies have examined relationships between MCI, QMCI, SQMCI and stream health (e.g. Quinn and Hickey 1990a, Collier et al. 1998, Scarsbrook et al. 2000) few have compared the way in which the three indices assess the health of the same streams (Stark 1993, Stark 1998). In this chapter, I examine relationships between MCI, QMCI and SQMCI and how they rank sites into degradation bands. Explanations for observed differences are explored. In addition, I investigate the utility of a quantitative MCI with low-level (order, class, phylum) identification (referred to as Ordinal QMCI; OQMCI).

4.2 METHODS

4.2.1 Data collection

Full descriptions of study areas, study sites, invertebrate sampling techniques and environmental measurement techniques are given in Chapter 2. In summary, benthic macroinvertebrates were surveyed using a "kick-net" sampling technique (0.5 mm mesh) at each of 230 3rd and 4th order stream sites in Canterbury between November 1999 and March 2000. In the laboratory, each sample was sub-sampled and a modified 100-count technique was used to estimate relative abundances of invertebrate taxa and community composition. This involved identifying and counting at least 100 animals (using a Bogoroff tray, Winterbourn and Gregson 1989) from each kick-net sample, then scanning the rest of the sub-sample for invertebrates that had not been found in the 100-count.

At each sampling site, habitat variables were recorded. They were later combined into an overall habitat assessment score, which included the following weighted variables; landuse, bank vegetation (abundance and composition), bank heterogeneity, channel heterogeneity, macrophyte abundance and composition, siltation/embeddedness, substrate packing and substrate heterogeneity (Table 2.2).

4.2.2 Data analysis

For each site MCI, QMCI and SQMCI scores were calculated. SQMCI taxon coded abundances (Rare, Common, Abundant, Very Abundant, Very Very Abundant) were derived from percentages (Table 4.2). Pearson correlations were used to test for relationships between MCI, QMCI and SQMCI scores for the 230 sites. Sites were also grouped into Stark's (1998) degradation categories (Table 4.3) based on their MCI, QMCI and SQMCI scores and compared using frequency distributions.

Table 4.3 Degradation categories defined by MCI, QMCI and SQMCI values. From Stark (1998).

Degradation category	MCI	QMCI & SQMCI
Clean Water	>120	>6
Doubtful quality of possible mild degradation	100-119	5 to 5.9
Probable moderate degradation	80-99	4 to 4.9
Probable severe degradation	<80	<4

4.3 RESULTS AND DISCUSSION

4.3.1 Relationships between MCI, QMCI and SQMCI

Correlations

Strong positive linear relationships were found between MCI and QMCI (r = 0.86, p<0.001, Fig. 4.1), MCI and SQMCI (r = 0.81, p<0.001, Fig. 4.2) and QMCI and SQMCI (r = 0.97, p<0.001, Fig. 4.3) based upon data for the 230 Canterbury streams. These highly significant correlations suggest that the three indices rank the same sites similarly, and that the QMCI and SQMCI assessments are more similar to each other than they are to MCI assessments. This also indicates that the quantitative count used by QMCI produces results consistent with coded abundance counts used by SQMCI, but that the invertebrate presence-absence data used in the MCI may rank sites slightly differently. It was also apparent that rankings of sites with QMCI and SQMCI were more similar to each other than either was to ranking with MCI (Table 4.4).

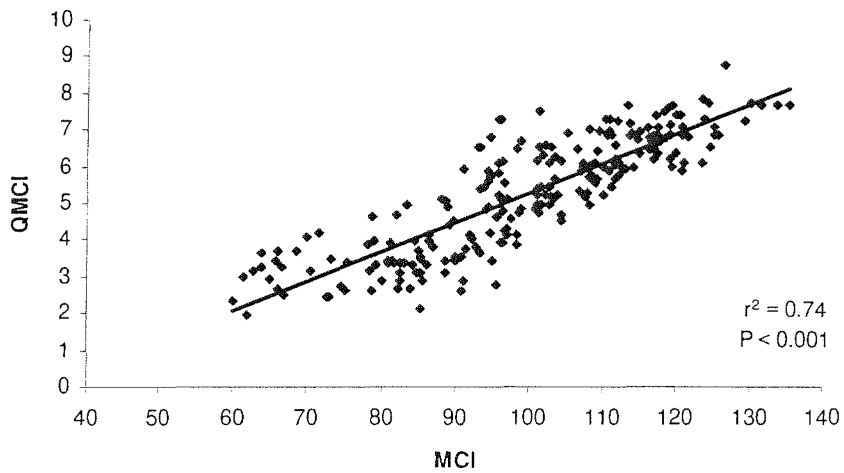


Fig. 4.1 Linear relationship between Macroinvertebrate Community Index (MCI) and Quantitative Macroinvertebrate Community Index (QMCI) based on kick-net data from 230 sites.

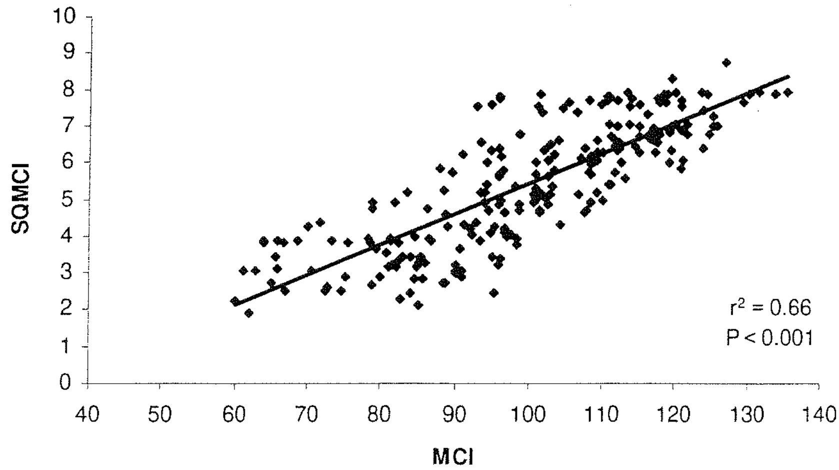


Fig. 4.2 Linear relationship between Macroinvertebrate Community Index (MCI) and Semi Quantitative Macroinvertebrate Community Index (SQMCI) based on kick-net data from 230 sites.

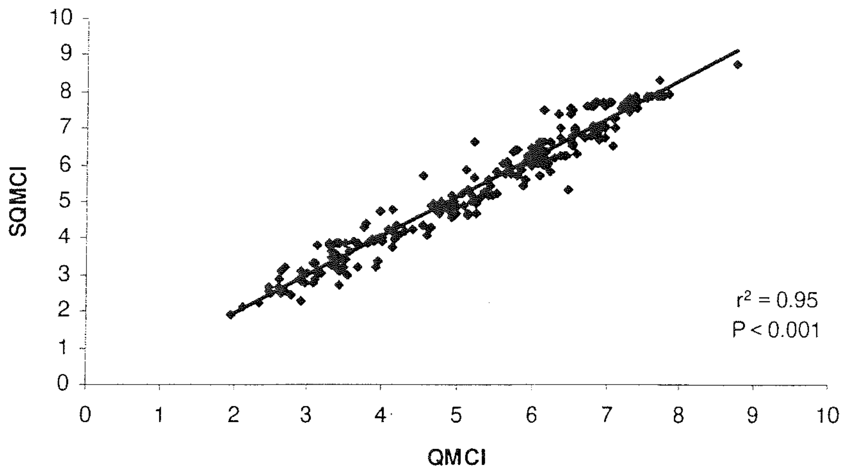


Fig. 4.3 Linear relationship between Quantitative Macroinvertebrate Community Index (QMCI) and Semi-Quantitative Macroinvertebrate Community Index (QMCI) based on kick-net data from 230 sites.

Table 4.4 The three highest and lowest ranked sites according to MCI, QMCI and SQMCI.

		Highest scores			
MCI	**Score**	**QMCI**	**Score**	**SQMCI**	**Score**
Ant Stream	135.5	Ohau Stream	8.8	Ohau Stream	8.7
Andrews Stream	133.7	Sudden Valley Stream	7.8	Maori Stream	8.3
Waiangarara	131.6	Cox River	7.7	Waiangarara	7.9
		Lowest scores			
MCI	**Score**	**QMCI**	**Score**	**SQMCI**	**Score**
Upper Heathcote River	62	Stony Creek	2.3	Stony Creek	2.3
Wainiwaniwa	61.3	Elephant Hill Stream	2.1	Elephant Hill Stream	2.1
Stony Creek	60	Upper Heathcote River	2	Upper Heathcote River	1.9

The finding of strong linear correlations between QMCI and SQMCI is consistent with the work of Stark (1998). In his study of 1356 North and South Island stony stream sites Stark obtained an r^2 of 0.953, whereas in the present study of 230 Canterbury streams an r^2 of 0.951 was obtained for QMCI and SQMCI. This suggests, strongly that the QMCI and SQMCI were ranking sites, similarly. The correlations between MCI and QMCI, and MCI and SQMCI were not as strong (r^2 = 0.735 and 0.661, respectively), although considerably stronger than the relationship Stark (1993) found between MCI and QMCI from 523 Surber samples (r^2=0.477).

These analyses indicate that ranking of sites by MCI differs a little from ranking with QMCI and SQMCI. An example of this is given in Table 4.4 which shows that the three highest scoring and three lowest scoring sites identified by QMCI and SQMCI were more similar than those identified by MCI.

4.3.2 Hypothetical example of reasons for differences between MCI, QMCI and SQMCI

One explanation for differences in the way the three indices grouped sites into degradation categories is shown by an hypothetical example given by Stark (1998, Table 4.5). In this example, Stark (1998) showed that two sites with the same MCI score can have high and low QMCI and SQMCI scores (Site A with high numbers of pollution sensitive taxa but few tolerant taxa, and Site B with high numbers of tolerant taxa but few sensitive taxa). Generally, 'clean water' faunas are dominated by Ephemeroptera and Trichoptera with Coleoptera and Diptera (mostly Chironomidae) also usually present. Degraded waters are often dominated by Mollusca, Crustacea, Chironomidae, other Diptera and Oligochaeta even though some Ephemeroptera, Trichoptera and Coleoptera may also be present (Chapter 3). The presence-absence invertebrate data used by MCI may not pick up these subtle differences in community structure (Table 4.5).

Table 4.5 MCI, QMCI and SQMCI scores for two hypothetical stream communities. From Stark (1998).

Taxon	Taxon Score	Site A	Site B
Mayflies			
Coloburiscus	9	100	3
Deleatidium	8	250	15
Nesameletus	9	15	2
Stoneflies			
Zelandoperla	10	15	2
Beetles			
Elmidae	6	3	15
True flies			
Maoridiamesa	3	2	100
Orthocladiinae	2	2	250
Caddisflies			
Beraeoptera	8	50	2
Helicopsyche	10	60	1
Oxyethira	2	2	60
Oligochaeta	1	1	50
Number of taxa		11	11
Number of individuals		500	500
MCI		124	124
QMCI		8.44	2.54
SQMCI		8.28	2.49

4.3.3 Frequency distribution of degradation categories

When my 230 sites were grouped according to Stark's (1998) degradation categories (Table 4.3), MCI placed the majority of sites in the 'doubtful quality of possible mild degradation' and 'probable moderate degradation' categories (Table 4.6, Fig. 4.4). In contrast, QMCI and SQMCI assigned the majority of sites to 'clean' and 'probable severe degradation' categories (Table 4.6, Figs. 4.5 and 4.6). This implies that category-assessment was more conservative with MCI, or that the boundaries between categories are not exactly equivalent among the assessment methods.

Table 4.6 The percentage of sites assigned to each water quality category by MCI, QMCI and SQMCI (n = 230).

	%		
Water quality category	**MCI**	**QMCI**	**SQMCI**
Clean	10	41	43
Doubtful quality of possible mild degradation	42	19	15
Probable moderate degradation	36	13	16
Probable severe degradation	12	27	26

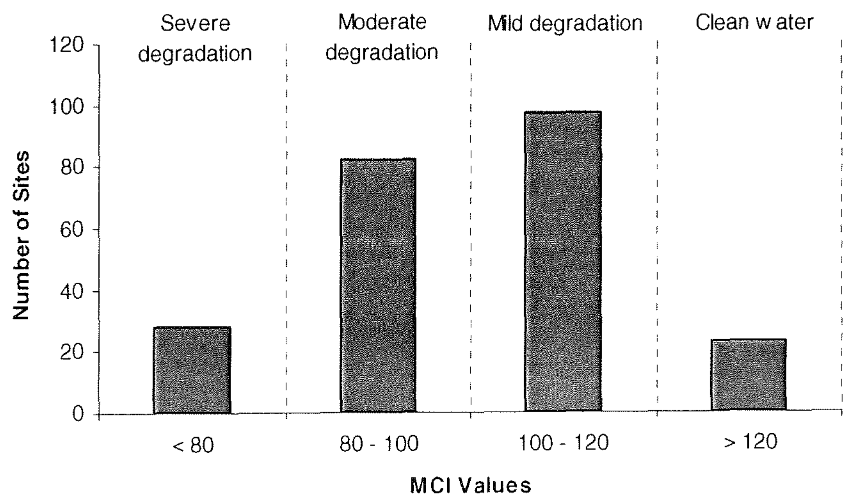


Fig. 4.4 Numbers of sites assigned to Stark's (1998) degradation categories by MCI.

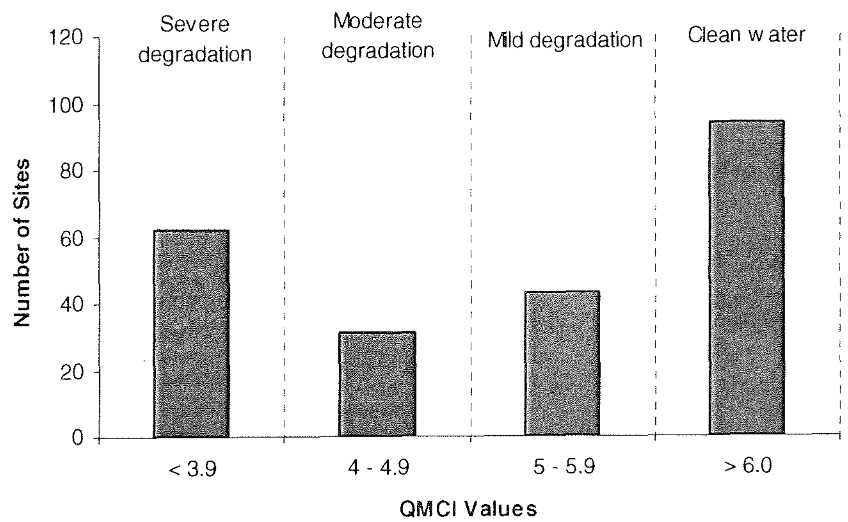


Fig. 4.5 Numbers of sites assigned to Stark's (1998) degradation categories by QMCI.

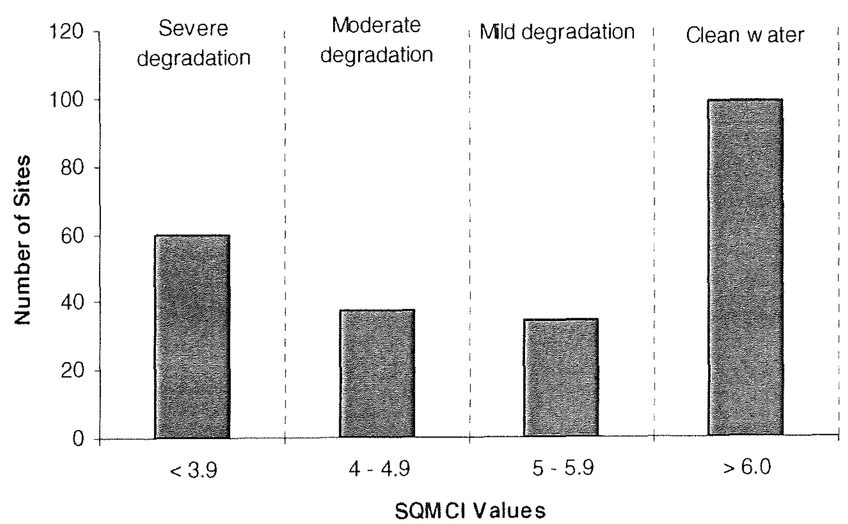


Fig. 4.6 Numbers of sites assigned to Stark's (1998) degradation categories by SQMCI.

Wellington and Southland Regional Councils have obtained similar results (Ryder 1998, Stansfield 1999) that also suggest MCI is more conservative in assigning sites to degradation categories or inappropriate band placement. To explore this further, Stark's (1998) QMCI and SQMCI degradation categories were changed from <3.9 to <3.8 (severe degradation), from 4-4.9 to 3.9-5.5 (moderate degradation), from 5-5.9 to 5.6-7 (mild degradation) and from >6 to >7 (clean). When this was done, more even frequency distributions (closer to that obtained with the MCI, Fig. 4.4) were obtained (Table 4.7). This is not entirely surprising since the indices are calculated in different ways and the degradation bands provide only an indication of a stream's condition, not its absolute state. Because the divisions fall on an even gradient (Fig. 4.7) the essentially arbitrary nature of their positions is apparent. Also, the upper and lower limits of the degradation bands were not drawn with reference to independent, objective criteria, such as physico-chemical and/or habitat data. In fact, in the publication in which the MCI was formally introduced, the degradation categories identified by Stark (1985) were "based upon subjective assessments of water quality". The indicative rather than absolute nature of the cut-off points between the groups was clearly appreciated by Stark who showed them as bands covering 10 MCI units.

Table 4.7 The percentage of sites assigned to each water quality category by MCI, and by QMCI and SQMCI with altered degradation categories (n = 230).

	%		
Water quality category	**MCI**	**QMCI**	**SQMCI**
Clean	10	25	22
Doubtful quality of possible mild degradation	42	27	28
Probable moderate degradation	36	36	31
Probable severe degradation	12	12	19

Another approach to evaluating the relative 'health' of a stream site in Canterbury is to consider its position in relation to all other sites. The appropriate 'continuum' in Fig. 4.7 can be used for this purpose with percentiles (shown) used as reference points.

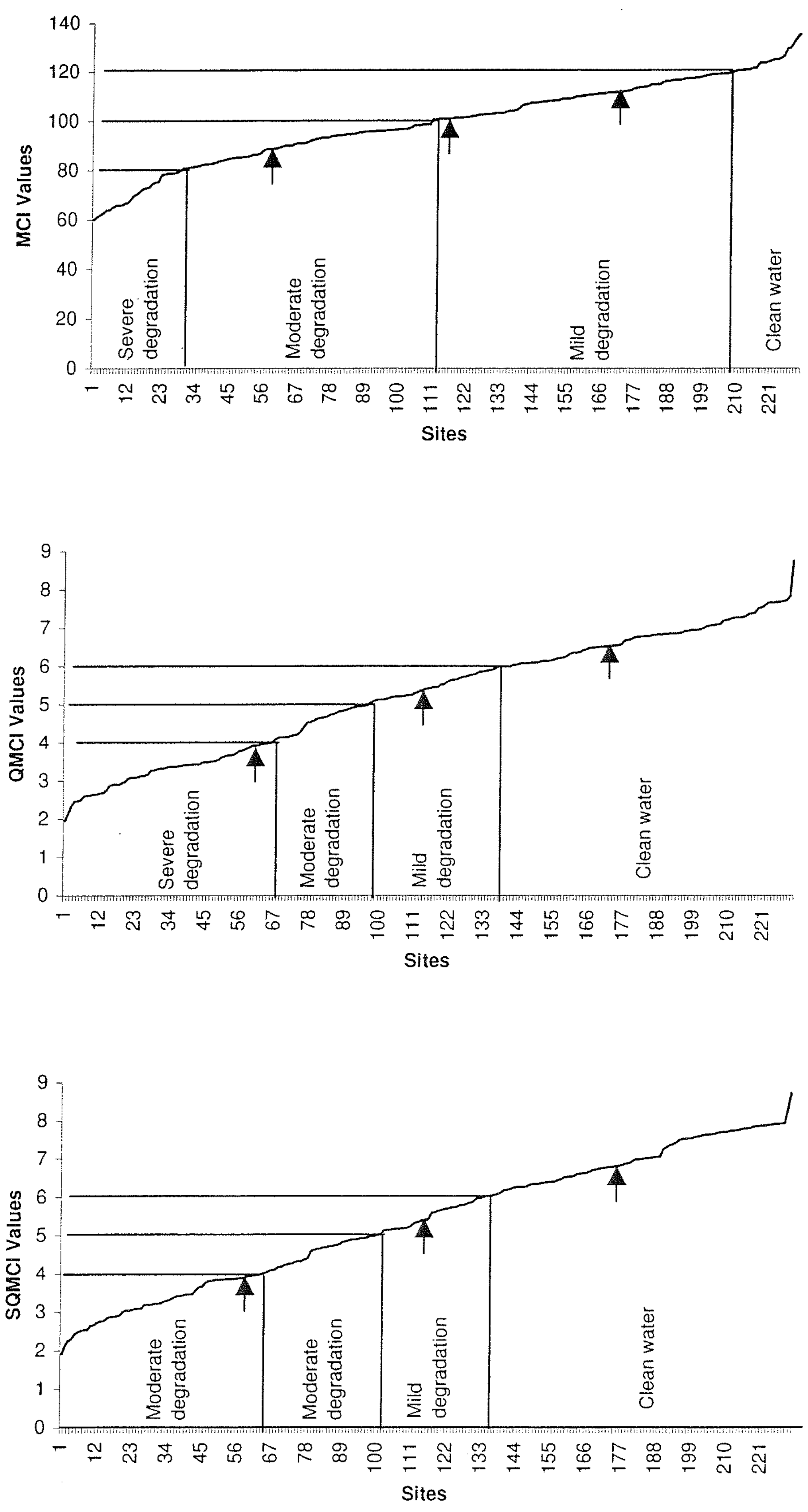


Fig. 4.7 Frequency distribution of MCI, QMCI and SQMCI scores arranged in ascending order with Stark's (1998) degradation bands shown. Also shown are the 25th, 50th and 75th percentiles (arrows). Note the even gradient on which the divisions fall, and therefore their essentially arbitrary positions.

4.3.4 An alternate approach: QMCI at coarse-level identification (OQMCI)

A potentially viable alternative to the MCI, QMCI and SQMCI, which requires less taxonomic expertise, is an Ordinal QMCI (OQMCI; Table 4.2). Because most conventional MCI scores are allocated to genera or families (not orders, classes or phyla) scores had to be given to these groups. This was done by averaging all values for taxa listed in each group by Stark (Table 4.1) and are shown in Table 4.8. Whilst there is a reasonable amount of variation in the sensitivity or tolerance values among genera or species in the same group, overall the OQMCI grouped sites similarly to MCI, QMCI and SQMCI (and in fact more similarly than Stark (1993) found between MCI and QMCI). This promising result indicates that coarse-level identification ranked the 'health' of sites in a comparable way to the other indices (Fig. 4.8). Use of the OQMCI could be considered when low-cost rapid assessments of large numbers of sites are required and either resources or expertise are limited.

Table 4.8 Average tolerance scores for OQMCI groups.

Group	OQMCI Value
Ephemeroptera	8.3
Plecoptera	7.9
Trichoptera	7.2
Diptera	4.4
Coleoptera	5.9
Crustacea	4.6
Odonata	5.5
Mollusca	3.3
Oligochaeta	1
Lepidoptera	4
Megaloptera	7
Mecoptera	7
Acarina	5
Hemiptera	5
Other Worms[1]	3

1, Nematoda, Platyhelminthes, Hirudinea

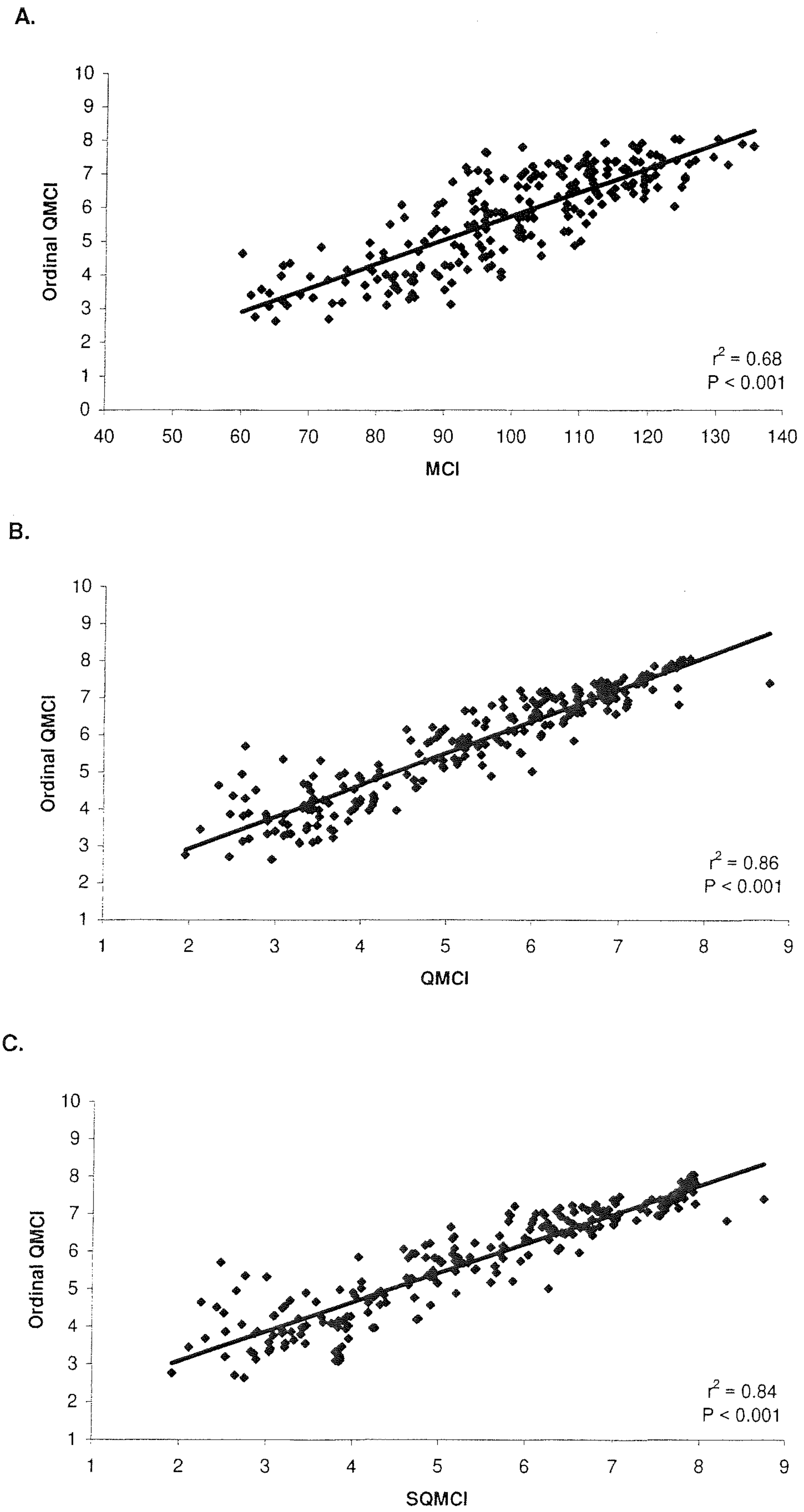


Fig. 4.8 Linear relationships between the new Ordinal Quantitative Macroinvertebrate Community Index (OQMCI) and MCI, QMCI and SQMCI.

4.3.5 Summary

The MCI and its derivatives (QMCI, SQMCI) are useful for determining water quality and the assessment of enrichment in stony streams (Quinn & Hickey 1990, Stark 1993, Stark 1998) and are widely used in New Zealand. The MCI is generally most cost-efficient, since fewer samples are required and sample collection and processing are least time consuming. However, the QMCI and SQMCI are preferred by some workers (Scott et al. 1994) since they reflect changes in the numerical composition of macroinvertebrate communities, which may be a consequence of enrichment or organic pollution (Stark 1998). My comparative investigation indicates that the MCI, QMCI and SQMCI do not assess the health of all streams consistently, and that the reasons for this inconsistency are likely to be a combination of two factors. Firstly, the presence-absence data used in calculating the MCI may not detect subtle changes in community structure, whereas the quantitative data used by QMCI and SQMCI may pick up changes and therefore group sites differently. Stark's (1998) hypothetical example showed how sites with the same MCI score can have low and high QMCI and SQMCI scores. Nevertheless, the correlations I obtained between MCI and QMCI, and MCI and SQMCI were reasonable strong. Secondly, the three indices did not always assign sites to the same degradation bands. Changes to the positions of the divisions may need to be contemplated so the MCI, QMCI and SQMCI provide more consistent assignments. However, because the Canterbury MCI, QMCI and SQMCI scores fell along a continuous and even gradient, the cut-off points must be arbitrary to a certain extent. Reference to the percentiles within which sites of interest fall may be a useful additional evaluation tool (Fig. 4.7).

4.3.6 Evaluation of biomonitoring approaches and their ability to assess stream health

The quantitative versions of MCI are dependent on numbers as well as kinds of taxa present, which means they are more likely to pick up subtle changes in community composition resulting from enrichment or pollution. More enriched sites may be dominated by Diptera including Chironomidae, Crustacea, Mollusca and Oligochaeta although some Ephemeroptera, and Trichoptera may still be present. Genera in these last two insect orders can have a disproportionate effect on health scores if quantitative or semi-quantitative data are not used. MCI values are less influenced by sampling effort (Stark 1998) however, and are probably best suited for gathering general health information on, for example, a regional basis. Furthermore, correlations between MCI and conductivity (a surrogate measure of enrichment) within the Canterbury data set were stronger than correlations between either QMCI or SQMCI and conductivity (see Chapter 5, Appendix 6) indicating that the MCI may be the more useful index for investigating general organic enrichment. Quinn and Hickey (1990a) also suggested that MCI may be a more sensitive index of water enrichment than QMCI for 88 New Zealand rivers because it had higher correlations with indicators of enrichment (e.g., nitrogen, periphyton chlorophyll a and AFDW). They suggest however, that the extra effort required to obtain QMCI values may be warranted where water quality is suspected of changing over short distances (such as above and below a discharge).

If a low-cost, rapid assessment technique is needed and a means for expert identification of invertebrates is unavailable, the ordinal QMCI could be considered since it produces results similar to the other three indices in Canterbury. The relative abundance of Ephemeroptera within this index is likely to give a particularly good indication of stream 'health' (Suren et al. 2000, Chapter 3). The OQMCI works because of general similarities in pollution tolerance and habitat preference of many taxa within the higher groups. For example, the ephemeropterans *Deleatidium, Coloburiscus, Nesameletus* and *Zephlebia* all prefer relatively pristine streams, whereas dipteran taxa are often found in enriched, silty streams (Chapter 3). Where taxa have MCI scores outside the norm for their group (e.g., Eriopterini is a dipteran with a MCI score of 9), their abundance at a site is likely to be relatively low and therefore, the quantitative aspect of the OQMCI reduces their influence on the overall score. Potential problems with a coarse-level identification approach could occur of course where a community consisted primarily of taxa with high MCI scores, yet the high-level group score was lower. For example, *Helicopsyche* and *Olinga* have MCI scores of 10 and 9, whereas the score for Trichoptera is 7.2 (Table 4.9). Conversely, the OQMCI for a site dominated by *Oxyethira,* which has an MCI score of 2, would be unrealistically high. Coarse-level identification however, is considered useful in Australia (Chessman 1995) and RIVPACS in the UK uses families, not genera as scoring taxa (Coysh and Norris 1999). Finally, I do not consider the OQMCI to be a substitute for more detailed quantitative studies with higher-level identifications since quantitative biotic indices have the capacity to identify specific effects (e.g., of a discharge) more accurately.

In the following chapter, I use the MCI to infer the general state of stream health in the Canterbury region and consider environmental and habitat factors potentially affecting the condition of streams.

CHAPTER FIVE

CANTERBURY STREAM HEALTH: BIOMONITORING SURVEY RESULTS

5.1 INTRODUCTION

The introduction of the Resource Management Act 1991 (RMA) in New Zealand has focussed attention on sustainable management of resources at the level of the ecosystem. The purpose of the Act is to promote the sustainable management of natural and physical resources, while safeguarding the life-supporting capacity of air, water, soil, and ecosystems; and to avoid, remedy, or mitigate any adverse effects of activities on the environment. Responsibility for implementing the RMA was given to regional councils through the development of policy statements, regional plans and resource consents. The RMA differs from its predecessor, the Water and Soil Conservation Act 1967 (WSCA), in that it strongly emphasises integrative management whereby the whole ecosystem is sustainably managed for use, development and protection of water processes. The WSCA dealt with water quality issues primarily in the context of use and development of water resources (Winterbourn 1999). Implementing the RMA, which includes reporting on State of the Environment, requires regular monitoring by regional councils. This thesis stems from Environment Canterbury's freshwater biomonitoring survey of the Canterbury region.

Macroinvertebrates are commonly used, at least in part, to assess stream health. Twelve of the 16 regional councils in New Zealand, as well as the Nelson City Council, and Marlborough District Council (equivalent to regional councils) presently use macroinvertebrates for biomonitoring or State of the Environment Monitoring (Winterbourn 1999, Jon Harding pers. comm.). The Macroinvertebrate Community Index (MCI) (Stark 1985) and its allies (Quantitative Macroinvertebrate Community Index (QMCI) and Semi-Quantitative Macroinvertebrate Community Index (SQMCI) are currently the standard assessment techniques for monitoring stream and river "health" in New Zealand.

Regional councils are using macroinvertebrates in objective and policy setting to assist their water management functions adhering to the RMA (Miller and Berry 1999). For example, the Wellington Regional Council uses macroinvertebrates as indicators of the overall status of water quality, to identify trends in water quality over space and time, to identify water quality related issues and to improve catchment management. Macroinvertebrates also provide information used to assess the effectiveness of objectives, policies, rules and methods in the Regional Policy Statement, Regional Plans and resource consents (Stansfield 1999). The Southland Regional Council uses the MCI and the SQMCI as indicators of stream quality to fulfil its obligation under the RMA. Additionally, macroinvertebrates were used to detect trends in the ecology of Southland's river ecosystems and help provide a level of understanding necessary for effective resource management (Ryder 1998). The Otago Regional Council is unique in that it has a specific policy that aims to see an increase in the MCI values of a number of waterways (Moore 1999). The data used in this thesis were collected for Environment Canterbury to facilitate obligations under the RMA in accordance with objectives and policy statements, and to aid in determining water quality and the processing of resource consents.

Biotic indices such as the MCI are useful, and amongst the most commonly used techniques for biomonitoring in streams and rivers (Norris and Georges 1993, Quinn et al. 1997). This is because (i) indices condense data and therefore aid interpretation; (ii) they are easily understood by persons with little biological expertise; (iii) they have more general value than physico-chemical measurements; (iv) they allow spatial and temporal comparisons even when sampling methods and habitats differ; and (v) their data requirements are often more modest than traditional statistical approaches (Norris and Georges 1993).

Biotic indices assign pollution scores to taxa based on accepted organism sensitivities to pollution and habitat disturbance, or numerical procedures based on distributions of taxa at a range of river sites grouped or ranked according to their degree of impact (Stark 1999). The MCI pollution scores used in New Zealand were based on the latter procedure and have been modified where it has been considered necessary to suit local conditions or integrate newly discovered taxa. For example, Otago Regional Council discovered the occurrence of pelecorhynchid flies and assigned sensitivity scores based on known habitat records in consultation with professional stream ecologists (Moore 1999).

Because biotic indices embody the pollution tolerances of indigenous taxa, they are regionally and geographically specific (Stark 1998). This means a biotic index developed in one country cannot be applied in another without modification. Thus, the MCI was designed to assess organic enrichment in stony streams and rivers in Taranaki, New Zealand, and was derived and modified from the British Biological Monitoring Party Score (BMWP). More specifically, it is analogous to a variant of the BMWP, the Average Score Per Taxon (ASPT), although scoring taxa in New Zealand are primarily genera rather than families as used in the BMWP system. Other biotic indices include the South African System (SASS); Hilsenhoff's Biotic Index (BI); Wisconsin Biotic Index and Family Biotic Index (FBI); the Spanish Biological Monitoring Water Quality (BMWQ) score system; and the Australian Stream Invertebrate Grade Number – Average Level (Stark 1998).

Biotic indices can be classified into three groups. Firstly, qualitative indices, use presence-absence data only and ignore abundances (e.g. BMWP, BMWP ASPT, MCI). Secondly, semi-quantitative indices include information on taxonomic composition and relative abundances (e.g. Chandler Score System, SQMCI), and thirdly, quantitative indices consider both taxonomic richness and numerical abundance (e.g. Hilsenhoff's BI and FBI, SASS, QMCI) (Stark 1999). Since the MCI does not take into account invertebrate abundance, it may be less sensitive to subtle changes in community composition than indices that do (Stark 1998). New Zealand appears to be the only place with qualitative, semi-quantitative and quantitative variants of the same biotic index (Stark 1999). The MCI, QMCI and SQMCI have been shown to be little affected by sampling method, water depth, current velocity and substratum (Stark 1993), and thus allow effects of differing environmental variables to be compared across stream types (within reason).

This chapter focuses on Canterbury stream health as assessed by the MCI and relates it to several environmental variables. Chapter 4 showed that except for the degradation bands, the relationships between MCI, QMCI and SQMCI were very similar, and subsequent analysis indicated that both QMCI and SQMCI health scores respond to environmental variables in a similar manner to the MCI.

Therefore, only MCI data are presented here and the QMCI and SQMCI results are given in Appendix 6. "Stream health" is considered to reflect a gradient of human disturbance and a subsequent gradient of biological condition (Fig. 5.1). The environmental variables considered here are those that were correlated significantly with MCI scores and include; altitude, landuse, conductivity, clarity, temperature, stream-bank vegetation, stream-substrate composition, algal composition, and a measure of overall habitat assessment. Habitat assessment scores are multi-factor scores and consider landuse, bank vegetation cover, stream bank heterogeneity and roughness elements, macrophyte biomass, siltation, and substrate size, packing, stability and heterogeneity (see Table 2.6 for their calculation).

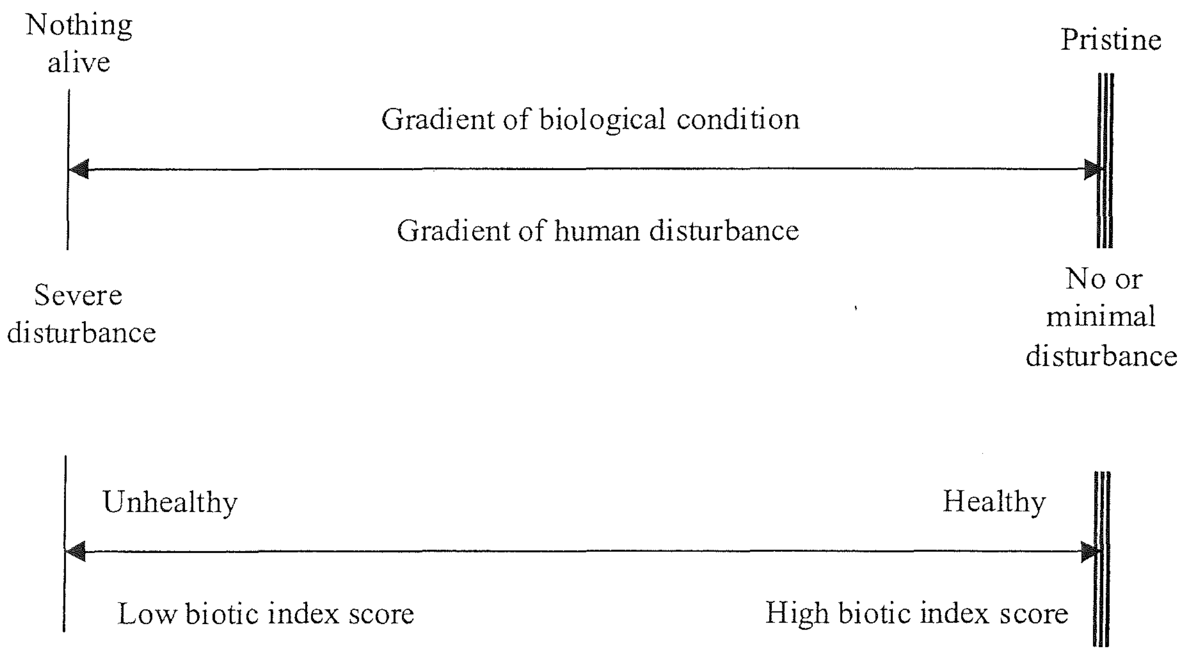


Fig. 5.1 Continuum of "stream health" and how human disturbance relates to biological condition. At one end of the continuum human disturbance eliminates all life, at the other end of the gradient are pristine or minimally disturbed living systems (top). A parallel gradient (bottom) defines streams as "healthy" or "unhealthy" depending on their biotic index score. Modified from Karr and Chu (1999).

5.2 METHODS

Sampling protocols, macroinvertebrate sample analysis, general macroinvertebrate data analysis, and the collection of environmental variables are described in Chapter 2.

5.2.1 Calculation of the MCI

The MCI was calculated from presence-absence data following Stark (1998) as follows:

$$MCI = 20\sum a_i/S$$

where S = the total number of taxa in the sample, and a_i is the MCI score for the i^{th} taxon (Table 4.1).

5.2.2 Data analysis

MCI scores were compared to other biological and physico-chemical variables using Spearman's rank-correlation coefficients (STATISTICA 5.0), regressions and frequency distributions. Correlations were considered significant at $P<0.001$. I used this very conservative level of significance to select and concentrate on the factors that were having the greatest effect on stream health, and to reduce the likelihood of Type II statistical errors.

The ordination technique PCA (Principal Components Analysis) was used to produce axis 1 scores, which were used as a surrogate for community structure. PCA was used because it is probably the best technique when data approximates multivariate normality and variables have linear relationships (McCune and Mefford 1999). Relationships between some environmental variables and PCA axis 1 scores were examined using Spearman's rank-correlation coefficients.

5.3 RESULTS

5.3.1 Invertebrate community characteristics

Descriptions of invertebrate community characteristics are given in Chapter 3, section 3.3.1. Invertebrate taxonomic richness for all sites is shown in Figure 5.2.

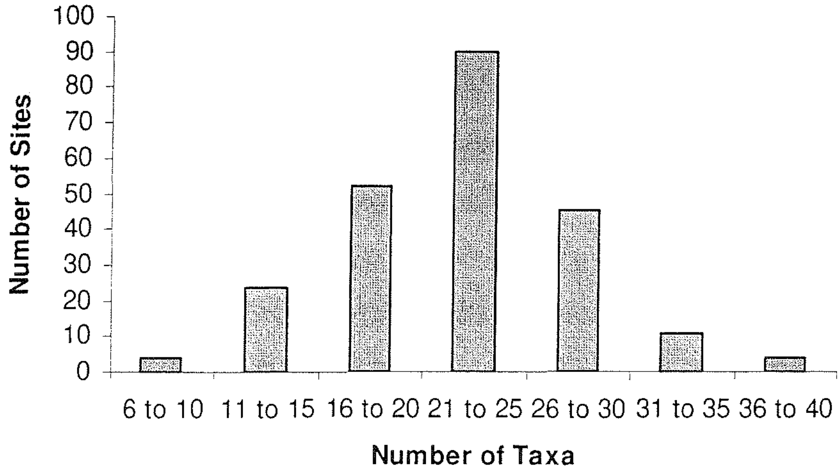


Fig. 5.2 Invertebrate taxonomic richness at all sites.

5.3.2 Macroinvertebrate Community Index (MCI)

MCI scores ranged from 60.0 to 135.5, with a median of 101.0 and an average of 99.6. The large variation in MCI values obtained reflects the diverse range of habitats sampled and the diverse effects that different landuses have on stream macroinvertebrates. The sites with the lowest MCI scores were on Stony Creek (MCI = 60.0, Site # 39), Wainiwaniwa River (MCI = 61.3, Site 147) and Upper Heathcote River (MCI = 62.0, Site # 141). The sites with the highest MCI scores were Waiangarara (MCI = 131.6, Site # 163), Andrews Stream (MCI = 133.7, Site # 13) and Ant Stream (MCI = 135.5, Site # 64). The higher scoring sites all occurred within indigenous forest and generally had a high proportion of the banks and riparian zone covered in vegetation. They all lacked fine sediment accumulations, and had heterogeneous substrate size class compositions. They were all found in steep catchments (>20^0), generally at high altitudes. In contrast, the low scoring sites had agriculture and exotic forest as the dominant landuses, and generally had low amounts of native vegetation on the immediate stream banks and in the riparian zone. Sedimentation was pronounced at two of the three sites, and substrate composition was more homogeneous. Low scoring sites occurred at lower altitudes, and were commonly on low gradient streams.

Generally, MCI values were higher inland, while many of the lower values were closer to the coast and in populated centres (Fig. 5.3). Streams at high altitudes (>500 m a.s.l.) generally had high MCI values while streams at low altitudes had low MCI values (Fig. 5.4a and b). Streams found in the 0-200 and 200-500 m a.s.l. brackets had a wide range of MCI and taxonomic richness values. Taxonomic richness ranged from 15 to 34 in the high altitude bracket. MCI and invertebrate taxon richness were related ($r = 0.27$, $P < 0.001$, Fig. 5.5) and although significant, outliers reduced the strength of the correlation. Raincliff Stream (Site 214) for example, had 27 taxa and a MCI value of only 79.3, and the nine sites with the lowest MCI scores all had 15 or fewer taxa. Little Kowai River, which had one of the five highest MCI scores (site 16) also had 15 taxa. Taxonomic richness ranged from 15 to 34 in the high altitude bracket.

Streams with catchments dominated by forest and tussock generally had higher MCI values (Fig. 5.6), whereas streams passing through urban/city catchments generally had lower MCI values and low taxonomic richness. The faunas of the latter comprised a small number of tolerant species, mainly chironomids, snails and oligochaetes. The wide range of MCI and taxon richness scores for streams with pasture as the dominant catchment landuse (Fig. 5.6) indicates that stream health varies considerably within this category. However, 42 pasture sites with MCI values greater than 100 and taxonomic richness greater than 25 indicate that healthy streams are attainable in agriculturally developed land.

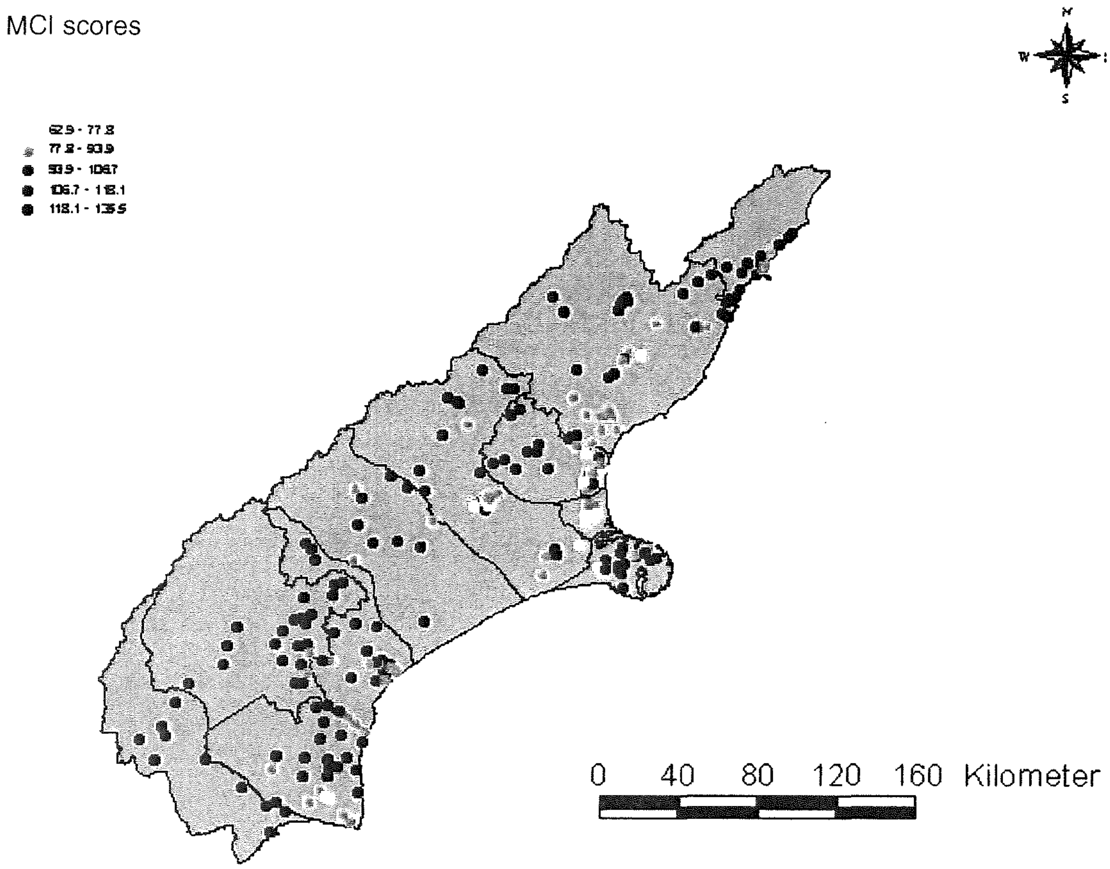
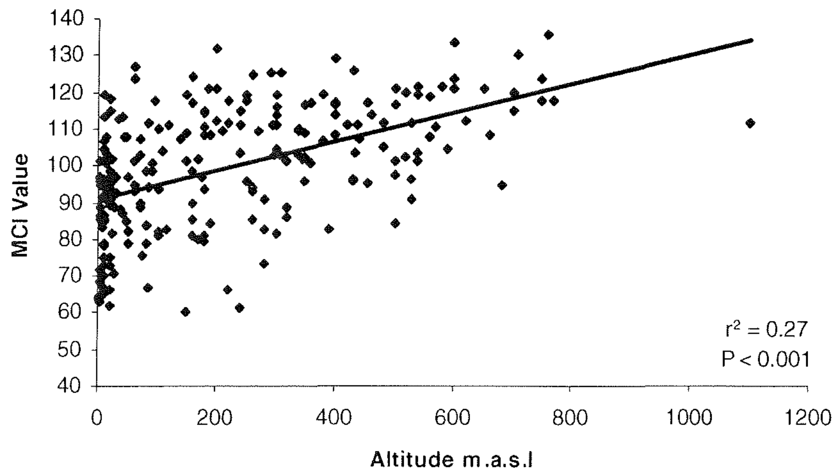


Fig. 5.3 Sites sampled within the Canterbury region and their MCI categories. Darker dots represent higher scoring streams.

a.



b.

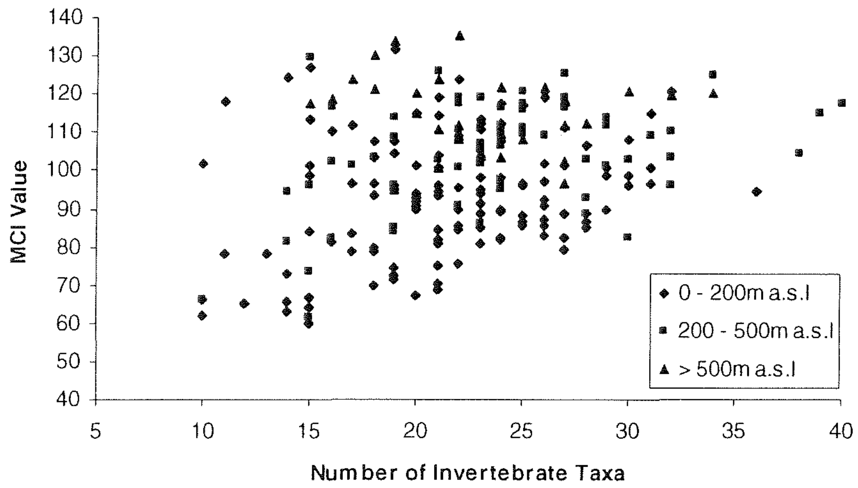


Fig. 5.4 Correlations between MCI values and (a) altitude, and (b) altitude bands for all sites.

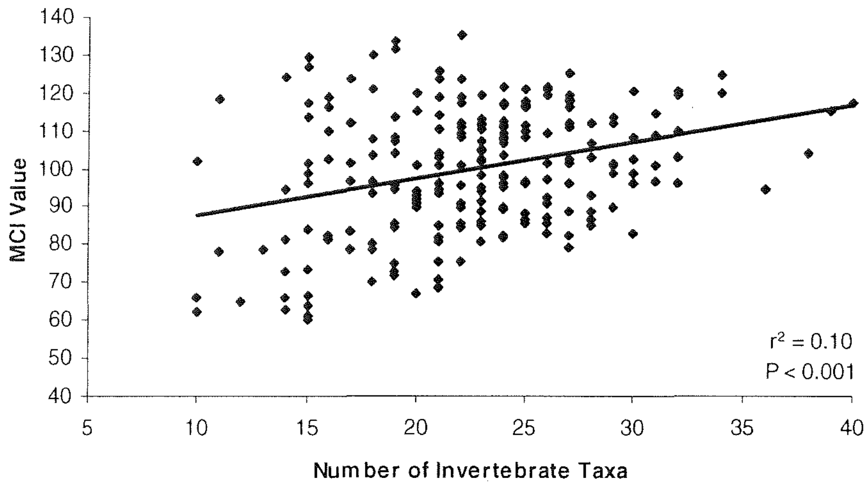


Fig. 5.5 Correlation between taxon richness and MCI values for all sites.

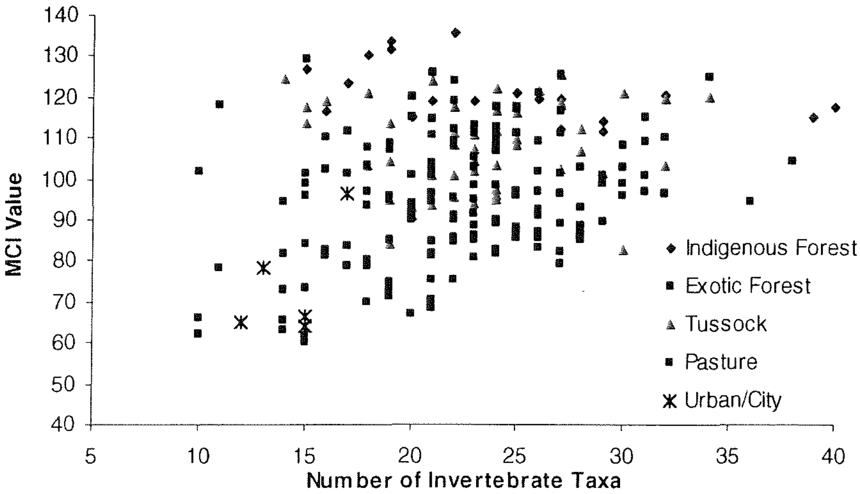


Fig. 5.6 Relationship between MCI values and taxon richness, with the dominant catchment landuse of all sites shown.

Associations with environmental factors

Of the environmental factors measured, conductivity was most strongly correlated with MCI (r = -0.56, P < 0.001, Fig. 5.7, Table 5.1, 5.2). Generally, an increase in water clarity was associated with an increase in MCI value (Fig. 5.7, r = 0.27, P<0.001), although the correlation was not strong. No significant correlation was found between pH and MCI, and although MCI was negatively correlated with water temperature (Fig. 5.7, r = -0.35, P<0.001) the relationship was not strong.

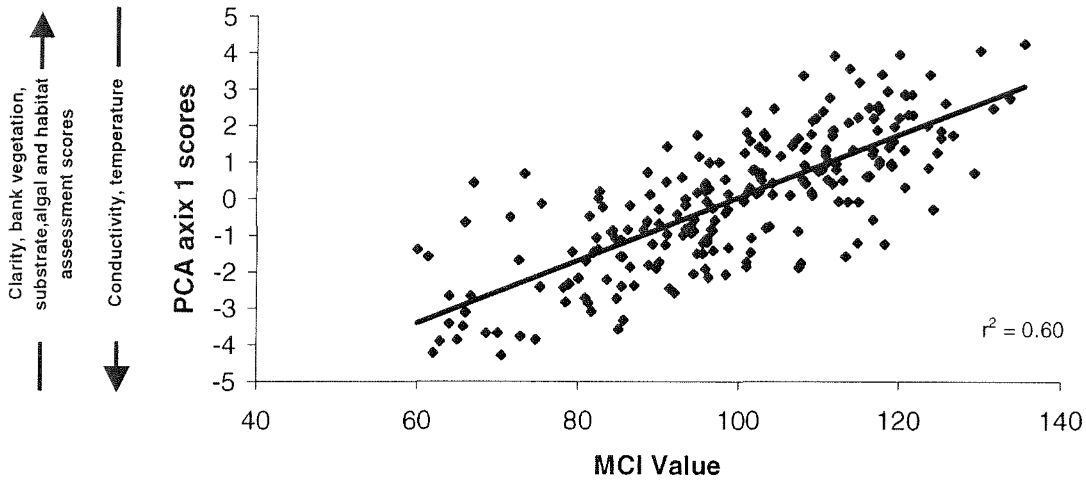


Fig. 5.7 Correlation between MCI values and PCA axis 1 scores. Environmental associations factors significantly associated with axis 1 scores are also show on the y-axis (and see Table 5.2).

Table 5.1 Matrix of Spearman rank-correlation coefficients between measured environmental factors and some invertebrate community indices and PCA axis 1 scores. **Bold, P<0.001**.

	Conductivity	Clarity	pH	Temperature	Habitat assessment score	Substrate score	Bank vegetation score	Algal score	No. taxa	Altitude	MCI	PCA axis 1 score
Conductivity	1											
Clarity	**-0.34**	1										
pH	0.05	-0.04	1									
Temperature	0.17	-0.09	**0.28**	1								
Habitat assessment score	**-0.43**	**0.31**	0.20	**-0.28**	1							
Substrate score	**-0.38**	**0.23**	**0.23**	-0.20	**0.73**	1						
Bank vegetation score	**-0.31**	0.18	0.12	-0.16	**0.48**	**0.28**	1					
Algal score	**-0.28**	0.08	-0.06	-0.14	0.20	0.23	0.14	1				
No. taxa	**-0.28**	**0.41**	0.02	-0.03	**0.28**	0.19	-0.03	0.10	1			
Altitude	**-0.68**	**0.23**	0.06	-0.18	**0.51**	**0.46**	**0.47**	0.22	0.16	1		
MCI	**-0.56**	**0.27**	0.05	**-0.35**	**0.62**	**0.47**	**0.50**	**0.46**	**0.27**	**0.55**	1	
PCA axis 1 score	**-0.78**	**0.43**	0.05	**-0.32**	**0.76**	**0.65**	**0.61**	**0.43**	**0.30**	**0.77**	**0.77**	1

Table 5.2 Spearman rank-correlations between PCA axis 1 scores and environmental factors and invertebrate community indices; ***, P<0.001.

Descriptors	Axis 1 score
Conductivity	-0.78***
Clarity	0.43***
pH	0.05
Temperature	-0.32***
Bank vegetation score	0.61***
Substrate score	0.65***
Algal score	0.43***
Habitat score	0.76***
MCI	0.77***

Bank vegetation scores were significantly correlated with MCI values ($r = 0.50$, $P<0.001$, Fig 5.7) indicating that intolerant taxa are more likely to be associated with indigenous, well vegetated stream banks. Similarly, substrate scores were significantly correlated with MCI values ($r = 0.47$, $P<0.001$, Fig. 5.7), with the larger, more heterogeneous, substrates supporting higher scoring invertebrate communities. Algal scores were significantly correlated with MCI values ($r = 0.46$, P<0.001, Fig. 5.7). Low algal scores denote the presence of algal forms such as long green filaments, and indicate high nutrient conditions and/or the occurrence of prolonged slow flows. High algal scores indicate the presence of diatoms and blue-green algae that are able to grow under moderate to low nutrient conditions and may be intensively grazed by invertebrates. Similarly, habitat assessment scores were significantly correlated with MCI values ($r = 0.62$, $P<0.001$, Fig. 5.7, Table 5.1).

5.4 DISCUSSION

The MCI, QMCI and SQMCI all responded in similar ways to the environmental variables considered in this chapter (Appendix 6) therefore only MCI data was presented. Additionally, the strong correlations between PCA axis 1 scores (ordination on environmental measurements; Fig. 5.7) suggest that the MCI responds in a predictable way to gradients of environmental conditions (also see Chapter 3). The condition of the macroinvertebrate communities (and therefore the perceived condition of the streams) was affected by geographic position, altitude, landuse, conductivity, clarity, temperature, bank vegetation, substrate, and algal composition in Canterbury. MCI indicated that streams were generally more healthy, if they were further inland, at higher altitudes, were in forested or unmodified catchments, had low conductivity and temperature, high clarity and high bank vegetation, substrate and algal scores. This discussion examines ways in which Canterbury stream health, as measured by MCI, was affected by the measured physico-chemical and environmental factors.

5.4.1 Canterbury Stream Health

As our understanding of stream health has improved over the last few decades, so has our attitude towards the quality of New Zealand waterways. The Resource Management Act (RMA) has changed thinking about stream management, such that a whole ecosystem approach, rather than just a water quality and quantity approach is needed. To this end, monitoring of the state of the country's freshwaters resources is required to determine whether objectives are being met. As indicated by Moore (1999) biological monitoring is useful because it provides information on the life-supporting capacity of water resources, which reflect hydrological, physico-chemical, and other habitat characteristics.

Geographic location and altitude

Both the locations of streams and their altitude influence stream health. Higher scores were generally obtained for streams that were further away from the coast, although this was almost certainly influenced by the extent of anthropogenic land development and land use intensity rather than geographic position *per se*. Sites occurring at >500 m a.s.l generally had higher health scores than low altitude sites. Charvet et al. (2000) also found that altitude could be used to discriminate mountain from lowland streams in France with respect to invertebrate taxonomic composition. The increase in MCI with altitude is consistent with findings in the Taranaki Region (Taranaki Regional Council 1999) and reflects the relative ease with which lower altitude, lower gradient land can be developed for forestry and agriculture. However, causal relationships between geographic location and altitude, and stream health are often unclear because of their complex interrelationships.

High scores were observed across the altitude spectrum, suggesting that healthy streams are attainable at low altitudes where landuse intensity is generally greater. In fact, a reasonable number of low altitude sites had high MCI scores. The Ohau Stream near Kaikoura for example, had a high MCI score and was located at only 60 m a.s.l. Ohau Stream is located in a Department of Conservation reserve and has a catchment clothed in indigenous forest, which implies that factors other than exclusively altitude (such as canopy cover) are important in determining stream health.

Landuse

Because a variety of factors were correlated significantly with stream health scores (i.e. altitude, dominant catchment landuse, conductivity, clarity, temperature, bank vegetation score, algal score, substrate score, overall habitat score) it was difficult to pin-point exactly what was driving stream health. In reality, it is probable that combinations of factors are involved, as well as others not tested here. However, correlations between stream health and conductivity, bank vegetation score, substrate score, algal score and habitat score were particularly strong, suggesting that they are important determinants of stream health. These are all driven to a certain extent by landuse (Quinn and Hickey 1990a).

Links between degree of land development and landuse, and invertebrate communities (and therefore stream health) are complex because multiple physico-chemical and environmental factors affect stream health as indicated above. However, macroinvertebrate scores almost certainly reflect dominant catchment landuse to some degree. The low MCI scores of streams in urban/city locations reflect the impact that development is having on invertebrate communities (Chapter 3). In contrast, streams with minimal catchment development, such as those in forested catchments had higher scores. Observations of stream health being affected by landuse is consistent with the results of several other New Zealand studies (Quinn and Hickey 1990a, Harding 1994, Collier 1995, Harding and Winterbourn 1995, Quinn et al. 1997, Harding et al. 1999, Vant et al. 2000) and research in other countries (Cao et al. 1996, Fore and Karr 1996, Doledec et al. 1999, Karr and Chu 1999, Charvet et al. 2000, Maxted et al. 2000).

Land development can change channel morphology (Davies-Colley 1997), increase water temperature and degree of enrichment, increase suspended solids and fine sediments and reduce amounts of coarse woody debris in streams (Quinn and Hickey 1990a). It can also change light levels and increase flow variability (Quinn and Hickey 1990b). The degree to which these changes occur in developed areas is reflected in the large amount of variation both in stream health and taxon richness. A reasonable proportion of sites with agriculture (sheep, dairy, deer, crops) as their dominant landuse attained high health scores, nevertheless, suggesting that many streams that pass through agricultural land and exhibited low health scores could be improved. High scoring agricultural sites typically had low conductivity and a relatively well vegetated riparian zone. Furthermore, the substrate within these streams was reasonably heterogeneous, and algal scores were relatively high. This suggests that by decreasing nutrient additions (and by implication conductivity levels and prolific algal growth), improvements in stream health including higher taxon richness are attainable. Protecting the vegetation in the riparian zone and increasing substrate heterogeneity can further enhance taxon richness and health scores. The latter may be achieved, or at least encouraged, by retaining vegetated riparian zones and changing farming practices to reduce sediment accumulation (Waters 1995, Biggs et al. 1998).

Some variations in stream health maybe related to the proportion of land developed. Low health scores generally occurred near populated areas, which were generally at lower altitudes. The relationship between stream health and proximity to urban areas is consistent with the findings of other studies; for example, Karr and Chu (1999) found a negative relationship between urban development (% impervious area) and richness of Ephemeroptera. Quinn and Hickey (1990b) found that sites with >30% of their catchment agriculturally development had significantly lower health indices (diversity, taxon richness, numbers of Ephemeroptera, Plecoptera and Trichoptera, biomass of species that are sensitive to changes in water quality related to eutrophication), compared to sites with less than 30% development. Quinn et al. (1997) established that invertebrate community composition varied most between streams draining pasture and native forest catchments in the Waikato, and Harding et al. (1999) found that the effect of agriculture on the Pomahaka River (Otago) significantly changed species composition. The change in community type with alteration in landuse could result from shifting food supply (Quinn and Hickey 1990b). For example, it is likely that increased concentrations of nutrients, elevated water temperatures and higher light levels are responsible for increased algal biomass at at least some of the more developed sites. Consequently, increased numbers and biomass of functional feeding groups such as algal piercing Trichoptera and collector-browser chironomids and molluscs may occur at the expense of facultative shredders such as *Austroperla* and *Olinga*. Additionally, the amount of "clean" stone surface habitat that is preferred by insects such as *Deleatidium* is likely to be reduced where algae are prolific, resulting in lower stream health scores. Not all streams however, are unhealthy in modified catchments.

Some sites in Canterbury showed exceptions to the general trends of low health scores in developed catchments. The Styx River at Styx Mill Reserve is located within an area dominated by city/urban development, and had relatively high stream health values compared with other city/urban sites. In 1980 the Christchurch Drainage Board made a biological survey of rivers in the metropolitan Christchurch area and outlying districts (CCC 1980). It included the Styx River, with three sites close to my study site. Presence-absence data were given, and from them MCI scores were calculated. The three sites had an average MCI score of 102.2, and ranged from 93.7 to 107.8. The value of 96.5 obtained in the present study implies that stream health has not changed significantly in the last twenty years. Although much of the Styx River catchment is dominated by city/urban development, the actual sampling site was located within a reserve on the outskirts of Christchurch. The increased vegetation, reduced sedimentation, lower temperatures and lower algal scores compared with other urban/city sites and the Styx River at a site further downstream, presumably as a result of the reserve, may account for the higher health scores. The Styx River was also sampled closer to the coast in this study and by the Christchurch Drainage Board. In 1980 a MCI score of 77.8 was obtained, compared with 75.2 in the present study. Again, this suggests that stream health has not changed significantly in the last twenty years, but was lower than at the upstream site.

The Christchurch Drainage Board study also included sites on the Heathcote River, the Avon and upper Avon River, the Wairarapa Stream and the Waimairi Stream close to my sampling sites. In 1980 MCI for the Heathcote averaged 67.0 (n=3), whereas in 1999-2000 it was 62.0, suggesting a slight reduction in stream health over the last twenty years. The Avon River and the Waimairi Stream had greater reductions in MCI (76.0 c.f. 64.0, 84.0 c.f. 66.7, respectively), whereas Dudley Creek, the upper Avon River and the Wairarapa Stream all had higher MCI scores in the current survey (60.0 c.f. 65.0, 57.1 c.f. 64.0 and 65.0 c.f. 78.5, respectively). The changes in scores that have occurred at these sites is not large, and because they indicate increases and decreases in stream health over time, it is apparent that streams and factors affecting their health need to be considered individually to determine reasons for change.

Two sites with low stream health scores worthy of comment were on Makerikeri and Stony Creeks. Both these sites were in exotic forest, and close to areas where logging was or had been occurring, recently. Additionally, Makerikeri was dry at the time of the second sampling (February) suggesting it may have been under low flow stress when sampled initially. Logging can have a significant effect on streams; for example, it can result in higher water temperatures, reduce substrate size and colonisable habitat through the addition of sediments, change flow regimes and primary production, change organic matter dynamics and alter macroinvertebrate community structure (Collier 1989, Corn and Bury 1989, Stone and Wallace 1998). Even after logging has stopped, there may be a long recovery lag time during which fine sediment continues to accrue from roads, and fine sediments that remains from earlier logging continues to be supplied to, and deposited in the stream. The result is often a decrease in invertebrate species richness (Haynes 1999) although in some cases, removal of riparian vegetation allows more light to reach the stream, thereby increasing primary production, and enhancing invertebrate numbers (Waters 1995).

Associations with environmental factors

The decline in stream health score with increasing conductivity probably reflects the effects of nutrient inputs. Conductivity was considered to be a surrogate for nutrient loading in New Zealand streams by Biggs et al. (1990), and Harding et al. (1999) found that it was significantly correlated with the intensity of agriculture in a New Zealand catchment. Conductivity is considered to be one of the best chemical predictors of water quality (Close and Davies-Colley 1990b), although it is influenced by geology and other geographic factors such as climate and location (Friberg et al. 1997). The slope of the regression lines relating conductivity and altitude to stream health scores in the present study were almost the inverse of each other reflecting that at lower altitudes, where there is generally a greater degree of anthropogenic development, conductivity is higher, probably because of a greater nutrient supply. Furthermore, although geology is the primary determinant of conductivity, most of the stream types sampled in the present survey (excluding Banks Peninsula and Geraldine) have greywacke as the underlying geology type which "factors" out geology to a large extent (Chapter 2).

The occurrence of lower MCI health scores where water clarity was low may also indicate that invertebrate composition is affected by turbidity in some way, although the interactions are likely to be complex. Clarity is to a large extent a function of suspended solids in the water (Davies-Colley and Close 1990), and it is likely that the presence of suspensoids affect invertebrate distribution (Biggs et al. 1990). Dissolved organic carbon (DOC) also affects clarity, and is associated with enrichment through both extent of development of the catchment (Quinn and Hickey 1990a) and periphyton (probably because of increased sloughing of organic matter at higher periphyton biomasses (Close and Davies-Colley 1990b)). The major effect of clarity (or turbidity) is probably on photosynthetic activity of plants and therefore primary production. Photosynthetic production will be reduced by turbidity if light is a limiting factor, and its biomass may affect grazer populations and other aspects of community structure (Ryan 1991). The effect on visual predators such as *Stenoperla* and *Hydrobiosis* may be important for example, since high turbidity may reduce their ability to detect and catch prey. Additionally, turbidity can decrease water temperature through the reflection of heat, and so may affect temperature sensitive species (Ryan 1991), such as *Zelandoperla*. Clarity, however, is also a reflection of catchment geology (Close and Davies-Colley 1990a, Close and Davies-Colley 1990b). It can vary considerably between streams both spatially and temporally but is generally related to flow (Close and Davies-Colley 1990a). This reflects the importance of comparing similar streams at appropriate spatial scales and under comparable flow conditions.

The general decline in stream health with increasing temperature found in the Canterbury surveys implies that in some cases invertebrate composition may be influenced by temperature (Chapter 3). This would be consistent with the findings of some overseas studies (Hawkins et al. 1997, Jacobsen et al. 1997). It is also consistent with the findings of other studies in New Zealand in which high scoring taxa were more often associated with cool streams, whereas low scoring taxa were often associated with warm ones (Biggs et al. 1990, Collier et al. 1998). Interestingly, I found no relationship between temperature and the number of invertebrate taxa per stream ($r^2 = 0.01$) in contrast to the results of other studies in which an increase in temperature (and a corresponding decrease in altitude) was associated with increasing numbers of taxa (see review by Jacobsen et al. 1997). Because a considerable component of the New Zealand stream fauna is considered to be 'cold adapted' (Boothroyd 2000) it is perhaps not surprising that no strong relationship between species richness and temperature was found.

Effects of bank, vegetation, substrate and algae on Canterbury stream health

The natural feature found to have the most influence on Canterbury stream health was streambed composition, which is largely a product of large-scale natural factors including climate, geology and topography (Biggs et al. 1998). The pattern of increasing health scores with increasing substrate scores indicates that healthier invertebrate communities are associated with more heterogeneous substrates (Chapter 3). This conclusion is consistent with the findings of some other New Zealand surveys in which invertebrate abundance and taxonomic richness were lowest in rivers with beds of silt or sand, or cobbles overlain with sand deposits (Quinn and Hickey 1990a, Quinn and Hickey 1990b, Collier et al. 1998). Additionally, in a sediment addition experiment (Dunning 1998) found that high sediment levels caused a reduction in invertebrate density and diversity in four North Island pine forest and pasture steams. Silt, mud and sand levels are often exaggerated by practices such as agriculture (Ryan 1991) and forestry (Waters 1995). Sediments incorporating a substantial fine fraction provide poor habitat for most stream invertebrates (apart from degradation-tolerant taxa such as some chironomids and oligochaetes)(Cobb et al. 1992), and can change periphyton community composition (Kutka and Richards 1996). Excessive deposited sediments change the character of surface and hyporheic habitats, increase embeddedness and reduce interstitial spaces (Ryan 1991, Davies-Colley 1997), where many invertebrates live (Waters 1995). Mud and silt can become stagnant because of high benthic respiration from accumulated organic matter and reduce exchanges between surface waters and the hyporheic zone (Davies-Colley 1997, Bunn et al. 1998). Low levels of oxygen can result to the detriment of many forms of stream life.

Because the nature of streambed material has a major influence on invertebrate communities (Chapter 3) it is a useful variable to include in assessments of stream health. The apparent lack of relationship between health score and substrate type beyond a substrate score of 2-3 (incorporates gravel, small cobbles, large cobbles, boulders) implies that bed material may be more useful for assessing sites composed of sand, mud, bedrock and man-made material. Thus, riffles with low substrate scores are likely to be degraded, whereas stream health at sites with higher scores are likely to be affected to a greater extent by other variables. Substrate composition and stability are, however, a product of a river's hydrological regime (bed slope, frequency and intensity of freshes, sediment supply regime) and geology (Biggs et al. 1990, Cobb et al. 1992) and hydrology can influence macroinvertebrate assemblages (Richards et al. 1997). Thus, some riverbeds will consist naturally of fine unstable sediments, and a paucity of associated invertebrates may wrongly suggest an unhealthy stream.

The importance of bank vegetation to overall stream condition was highlighted by the strong correlations found between bank vegetation scores and MCI scores in Canterbury. The riparian zone is important because it provides habitat for adult insects (Collier and Smith 1998), and also shade (Davies-Colley and Quinn 1998), which helps to reduce stream temperatures, especially in slow-flowing streams (Biggs et al. 1998). Thus, it may prevent the proliferation of invasive macrophytes (Bunn et al. 1998) and limit periphyton growth (Towns 1981, Davies-Colley and Quinn 1998). Root systems stabilise banks, reducing erosion (Cottam 1999) and riparian vegetation reduces stream exposure to sunshine and winds (Waters 1995). Allochthonous inputs (leaves, twigs) from bank vegetation provide food for some invertebrates. Additionally, the riparian zone filters runoff from the surrounding land, removing nutrients from ground and surface water (Biggs et al. 1998, MacGibbon 2000). The importance of interactions between the stream and its banks, means that a reduction in bank vegetation generally results in a reduction in stream health scores, reduced invertebrate densities (Clenaghan et al. 1998) and changes in stream ecosystem function (Stone and Wallace 1998).

Algal scores of Canterbury showed similar trends to substrate and bank vegetation scores, with increases generally being associated with increases in MCI values. Periphyton is a useful indicator of stream health (Kutka and Richards 1996, Pan et al. 1996, Biggs et al. 1998, but see Kutka and Richards 1997) for a number of reasons (see below) and therefore one might expect it to be correlated with invertebrate health scores. Algae, like invertebrates, have limited mobility, and therefore are indicative of pollution and disturbance events even after they have passed, while the amount and types of periphyton present can be useful indicators of the nutrient status (enrichment) of water (Kutka and Richards 1997, Biggs et al. 1998).

The habitat assessment scores I calculated incorporated measurements of a number of factors (landuse, stream bank vegetation cover, bank heterogeneity, macrophyte composition, siltation/embeddedness, packing of substrate material, and substrate heterogeneity; Table 2.6) and the strong correlation between them and MCI scores suggests that invertebrate community composition is indeed affected by combinations of physical factors. Of these siltation, landuse, channel heterogeneity and substrate heterogeneity were identified as particularly important because of their strong correlations with MCI scores.

Comparisons with other regions

Biotic indices calculated by different organisations cannot always be compared directly because different sampling and processing methods are sometimes employed. For example, different regional councils in New Zealand use different sampling methods (Surber sampler, kicknet), mesh sizes, sample at different times of the year, and in different places (pools, riffles and runs, productive areas, representative areas). Additionally, many biologists use modified taxonomic lists and may not identify certain groups, particularly the Chironomidae, to the taxonomic level listed by Stark (1993, 1998). In most cases, these modified taxonomic lists result in higher MCI values (Moore 1999). Consequently, Wellington and Southland Regional Councils tend to have higher MCI values than Taranaki, Canterbury and Otago Regional Councils. Here I tentatively evaluate Canterbury stream health in comparison to some other regions.

Median and mean MCI values for Canterbury streams were similar to those obtained in comparable invertebrate surveys by the Taranaki Regional Council (Taranaki Regional Council 1999), and higher than those obtained by the Otago Regional Council (Moore 1999). Wellington (Stansfield 1999), Southland (Ryder 1998) and Bay of Plenty (Donald 1995) Regional Councils mean and median MCIs are similar to each other, but higher than those for Otago, Taranaki and Canterbury. Hawke's Bay values were higher again in 1996-1999 (unpublished data). These results suggest that Canterbury stream health is similar to that in Taranaki, above that of Otago, but lower than in Hawke's Bay, Wellington, Southland and Bay of Plenty. Wellington and Southland stream health results may be an artefact of modifying the original taxonomic lists in an effort to save time/money however. National comparisons are also difficult because a regional council's rationale for sampling varies. For example, Otago and Taranaki Regional Councils sample the majority of their sites for resource consent purposes (Adrian Meredith pers. comm.) whereas Canterbury sites were chosen to cover all stream types.

5.4.2 *Conclusion*

Overall, sites in Canterbury with high intensity landuse had lower stream health than streams with low intensity landuse according to the MCI. However, it was possible for high scores to be attained in high intensity landuse catchments. Management of point source wastewaters has improved over the past few decades, so that non-point sources such as runoff and leaching from land have become relatively more important than they were in the past (Vant et al. 2000). Presently, point source discharges require resource consents, but landuses such as pastoral farming do not. A reduction in non-point source additions is necessary to further improve stream heath in modified landscapes since pressure on rivers will continue to grow as numbers of people and livestock (especially dairy) increase.

Landuse practices are often the main cause of degraded water quality and therefore improvements in river condition are likely to depend on changing farming practices including riparian restoration and streamside enhancement, rather than improving wastewater treatment (Vant et al. 2000). A variety of factors influence invertebrate communities however, and make it difficult to determine the exact causes. More experimental manipulations are required to confidently identify causative mechanisms, however, stream health scores in Canterbury were most strongly correlated with conductivity, substrate composition, bank vegetation and algal scores and altitude, suggesting that these play important roles either directly or indirectly in determining stream health.

A significant problem in interpreting stream health based on scores such as MCI is the difficulty in determining the degree to which perceived stream health is related to degradation and how much observed results are related to natural variation. Over reliance on one index at the expense of other independent measures is a risky strategy when trying to assess stream health. Different measurements have strengths and weaknesses, and relying on a single measurement can easily bias assessments (see Chapter 4). Because stream ecosystems are complex and multidimensional, the more measurements one takes, the more confidently one can gauge stream health, providing the measurements can be integrated and evaluated together. This is an important issue when using multimetric scores (see Chapter 6).

CHAPTER SIX

ASSESSMENT OF BANKS PENINSULA STREAM HEALTH USING BENTHIC MACROINVERTEBATES: A MULTIMETRIC APPROACH

6.1 INTRODUCTION

The purpose of biological assessment is to characterise the status of water resources and to monitor trends in the condition of biological communities so that stream condition can be inferred (Resh et al. 1995a). While a variety of methods have been developed for inferring stream condition from invertebrate monitoring data, until recently, only a limited number of diversity, comparative and biotic indices have been used in New Zealand (Boothroyd and Stark 2000). Multimetric methods were developed in the United States from Rapid Bioassessment Protocols (Plafkin et al. 1989), which were originally designed to determine whether a stream contained an expected faunal community. Protocols have since been developed for determining the degree of impairment, evaluating the effectiveness of management and regulation, and for trend monitoring (Boothroyd and Stark 2000).

As stated above, the goal of biological assessment is to detect and characterise the status of water resources, and understand change in the biological system that results from human actions. Thus, biological assessment must have a standard against which the conditions of one or more sites of interest can be evaluated (Karr and Chu 1999). Fundamental to multimetric bioassessment methods is the classification of reference areas (Gerritsen 1995). Exactly what the 'reference condition' is, has been the subject of considerable debate (e.g. Reynoldson et al. 1997) and is outside the scope of this study. However, in this chapter, the reference condition is considered to be the best available condition that can be expected at similar sites given acceptable land or water use. Reference streams are used to predict the fauna expected at a test site, and the health of that site is then inferred from its degree of divergence from the reference condition. Biological metrics are used to measure this divergence, a metric being defined as an ecological attribute of an assemblage estimated from a collection of organisms, and responsive to perturbation or disturbance (Barbour et al. 1995). The multimetric approach involves calculating an array of invertebrate structural and functional metrics that individually provide information on diverse biological attributes and when combined give an overall indication of the condition of a biological community (Barbour et al. 1996). By using several metrics a variety of attributes of a benthic macroinvertebrate community can be measured, and can be expected to provide a stronger assessment than a single metric, which may fail to reveal the effects of multiple stressors (Boothroyd 1999). Indices have a range of sensitivities to different kinds of stress and therefore must be chosen or calibrated for the area and conditions of interest.

The research discussed in this chapter was confined to Banks Peninsula, which forms a relatively homogeneous geographical unit (Chapter 2) and was discriminated as a discrete ecoregion by Harding (1994). The aim of the study was to select biological indices that best discriminated reference sites from sites impaired by habitat disturbance and organic pollution in Banks Peninsula streams, and to combine the metrics into an index of biological integrity, the Banks Peninsula Macroinvertebrate Index (BPMI). Whilst the multimetric approach has been used for stream biomonitoring in New Zealand (Quinn et al. 1997, Collier et al. 1998), no studies have specifically developed multimetric indices for use in New Zealand ecoregions.

6.2 METHODS

Sites were sampled on 16 Banks Peninsula streams (Fig. 6.1) and classified as either reference or degraded. Descriptions of study areas, study sites, invertebrate sampling techniques and environmental measurement techniques are given in Chapter 2. The criteria for selecting a site as 'reference' or 'degraded' was based on non-biological factors, including landuse, composition of riparian vegetation, water quality as indicated by conductivity and personal observations. Thus, sites with combinations of low intensity landuse (e.g. sheep farming, native and exotic trees), high quality, well vegetated riparian zones and low conductivity were considered to be in good condition and were designated reference sites. Conversely, sites with high intensity landuse (e.g. dairy, deer, urban), with low quality, poorly vegetated riparian zones and high conductivities were considered to be in poor condition and were designated degraded. Non-biological factors were used to avoid circular reasoning that can occur if biological factors are used; i.e., sites are healthy because the fauna is intolerant of pollution, and the fauna is

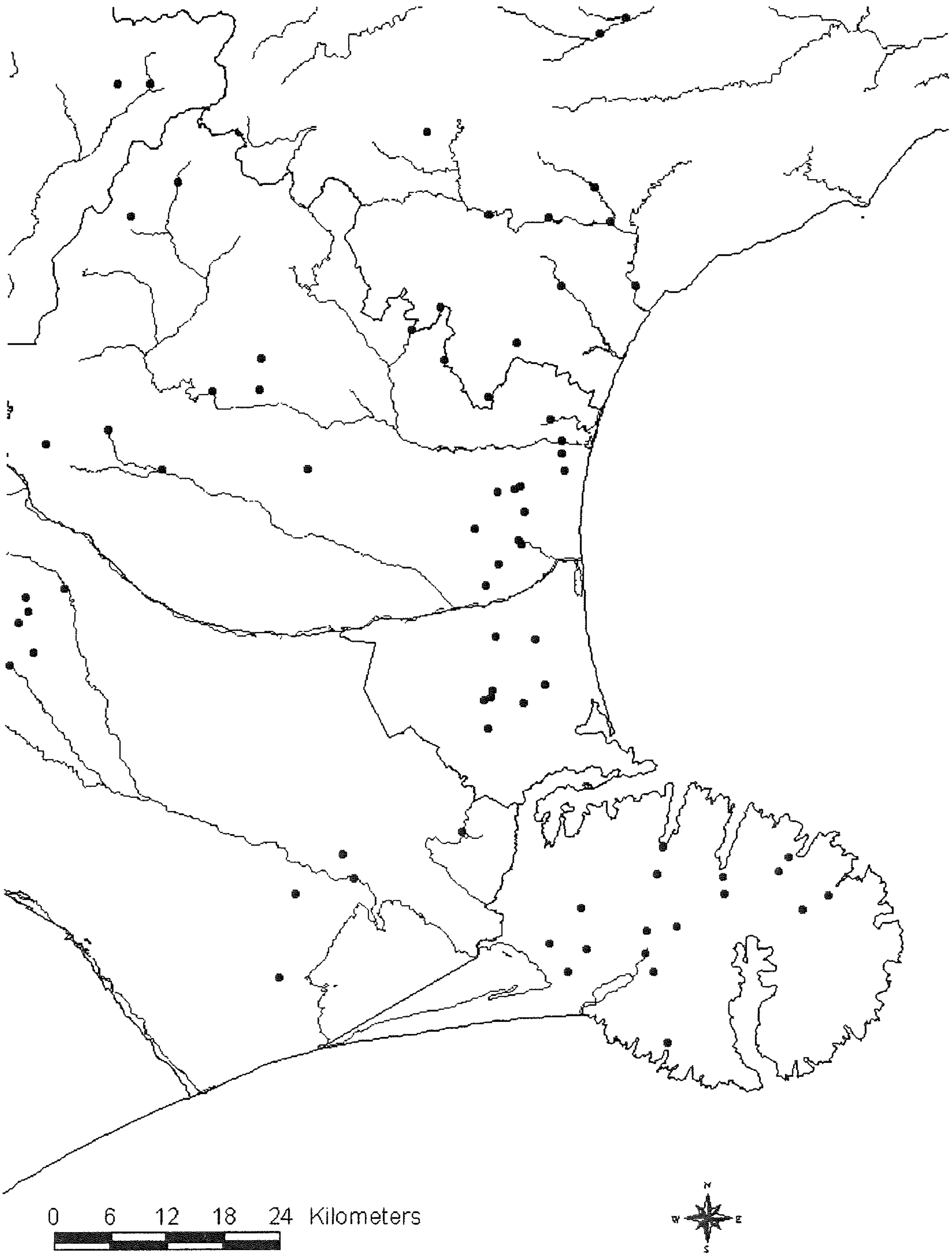


Fig. 6.1 Locations of the 16 Banks Peninsula sites (shown in red) used in the development of the BPMI. Blue dots represent some of the other sites sampled in this survey. Major rivers are also shown.

healthy because the sites are healthy. Summaries of site and habitat variables for reference and degraded streams are given in Table 6.1.

Table 6.1 Summary of habitat and site variables for reference (Ref) and degraded (Deg) sites. See Chapter 2 for calculations of substrate, bank vegetation, algal and habitat scores.

	Median		25 percentile		75 percentile		Range	
	Ref	Deg	Ref	Deg	Ref	Deg	Ref	Deg
Conductivity ($\mu S\ cm^{-1}$)	127.5	133.6	116.9	125.7	136.6	142.1	63.3 - 152.3	123.4 - 203.0
Substrate score	3.9	3.6	3.9	3	4	4.1	3.8 - 4.1	0.8 - 4.2
Bank vegetation score	1.9	1.7	1.8	1.6	2.1	1.9	1.7 - 2.3	1.3 - 2.2
Algal score	4.2	2.5	4	1.8	4.3	3.1	3.2 - 4.9	1.0 - 4.1
Habitat score	130	120	125	120	130	130	110 - 130	63 - 130
Width (m)	3.7	4.4	3.3	2.9	4.4	4.6	3.0 - 5.2	2.5 - 6.0
Depth (m)	0.4	0.3	0.2	0.2	0.4	0.4	0.2 - 0.4	0.2 - 0.6
Altitude (m a.s.l)	60	20	50	20	90	20	35 - 150	5 - 60

6.2.1 Selection and testing of metrics

Metrics were selected by comparing reference sites to sites of poor quality and/or known anthropogenic impacts. Twenty-nine measurements of macroinvertebrate benthic assemblages were considered initially (Table 6.2). These incorporated 5 richness measures, 14 composition measures and 10 tolerance measures.

Many of the metrics were based on the number of lower taxa within a taxonomic group and are self-explanatory. Richness measurements reflect the diversity of the invertebrate population. Increasing diversity is often correlated with increasing stream health (Chapter 5) and suggests that habitat, food source and niche space are adequate to support many species (Barbour et al. 1996). Number of taxa measures variety of the invertebrate assemblage. EPT taxa is a measure of the number of kinds of Ephemeroptera, Plecoptera and Trichoptera (Table 6.2). These insect orders are generally considered to be sensitive to pollution, and a loss of EPT taxa is indicative of degradation (Collier et al. 1998).

Measurements of relative abundance provide information on the numerical composition of the fauna and the importance of different taxa. Metrics that generally increase with degradation include percent Diptera, Chironomidae, Crustacea, Mollusca and Oligochaeta. Metrics that are likely to decrease include percent Ephemeroptera, Trichoptera, Plecoptera, Coleoptera and Odonata (Chapter 3), although certain individual taxa within these major groups may not conform to this pattern (e.g. *Oxyethira* is a trichopteran genus commonly associated with degraded stream conditions). Percent EPT/Mollusca, EPT/Oligochaeta and EPT/Mollusca + Oligochaeta provide ratios of clean water to degraded water faunal components (Table 6.2).

Table 6.2 The 29 measurements of macroinvertebrate assemblages (metrics) initially considered in the Banks Peninsula Macroinvertebrate Index (BPMI), and the expected response of each to increasing degradation.

Category	Metric	Definition	Expected response to increasing degradation
Composition	% EPT	Measures the abundance of taxa belonging to Ephemeroptera, Plecoptera and Trichoptera	Decrease
	% EPT2	As above, but excluding the pollution tolerant *Oxyethira* and *Paroxyethira*	Decrease
	% EPT3	As above, but excluding the pollution tolerant *Oxyethira*, *Paroxyethira* and *Aoteapsyche*	Decrease
	% Ephemeroptera	Percent of mayflies	Decrease
	% Plecoptera	Percent of stoneflies	Decrease
	% Trichoptera	Percent of caddisflies	Decrease
	% Diptera	Percent of dipterans, excluding chironomids	Increase
	% Coleoptera	Percent of beetle larvae and aquatic adults	Decrease
	% Chironomidae	Percent of chironomids	Increase
	% Crustacea	Percent of crustaceans	Increase
	% Odonata	Percent of dragonfly nymphs	Increase
	% Mollusca	Percent of molluscs	Increase
	% Oligochaeta	Percent of aquatic worms	Increase
	% Dominant taxa	Measures the abundance of the single most abundant taxon	Increase
Richness	No. EPT taxa	Number of taxa in the insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies)	Decrease
	No. Ephemeroptera	Number of taxa in the Ephemeroptera (mayfly)	Decrease
	No. Plecoptera	Number of taxa in the Plecoptera (stonefly)	Decrease
	No. Trichoptera	Number of taxa in the Trichoptera (caddisfly)	Decrease
	No. Taxa	Measures overall variety of the macroinvertebrate assemblage	Decrease
Tolerance	MCI	The Macroinvertebrate Community Index is a biotic index that assigns tolerance scores to individual taxa	Decrease
	QMCI	Quantitative version of MCI	Decrease
	SQMCI	Semi-quantitative version of MCI	Decrease
	OQMCI	Quantitative version of MCI with low-level (order, class, phyla) invertebrate identification	Decrease
	EPT as % of total taxa	Percent of individuals in Ephemeroptera, Plecoptera and Trichoptera combined	Decrease
	% EPT / % Mollusca + % Oligochaeta	Ratio of percent Ephemeroptera, Plecoptera and Trichoptera, to percent Mollusca plus percent Oligochaeta	Decrease
	% EPT / % Oligochaete	Ratio of percent Ephemeroptera, Plecoptera and Trichoptera, to percent Oligochaeta	Decrease
	% EPT / Mollusca	Ratio of percent Ephemeroptera, Plecoptera and Trichoptera, to percent Mollusca	Decrease
	% intolerant taxa	Percent of taxa with MCI tolerance scores >7	Decrease
	% tolerant taxa	Percent of taxa with MCI tolerance scores <3	Increase

Tolerance metrics assign sensitivity scores to invertebrate taxa. The MCI, QMCI, SQMCI and OQMCI were calculated as described in Chapter 4. Percent intolerant taxa and percent tolerant taxa were calculated using Stark's (1998) tolerance scores (Table 4.1). Taxa with scores ≤3 were considered to be intolerant, whereas those ≥7 were considered to be tolerant (Table 6.2).

A site was considered to be correctly assigned to the degraded category for a particular metric if its site score was less than or equal to the 25th percentile of the reference population (Maxted et al. 2000). The percentage of sites that were correctly assigned by each metric were calculated, and those metrics that classified the highest percentage of sites correctly were selected as the basis for development of an aggregated index (see below).

6.2.2 Index development

The Banks Peninsula Macroinvertebrate Index (BPMI) was further refined using Spearman rank correlation coefficients. Metrics that were strongly correlated (r-values < -0.75 and > 0.75) were considered redundant, indicating that they were contributing more or less the same information to the index. In these cases, only one member of a pair was included. Priority was given to selecting at least one metric from each of the three categories (i.e. tolerance, richness, composition), to further reduce redundancy. A summary flow diagram of the methods used in the development of the BPMI is given in Figure 6.2. The selected metrics and overall index were tested for sensitivity and ability to discriminate between reference and degraded sites using box-and-whisker plots and one-way analysis of variance (ANOVA). Comparisons between the BPMI and MCI/QMCI were tested with Spearman rank correlation.

6.3 RESULTS

6.3.1 Selection of metrics for Banks Peninsula streams

Sixteen metrics were selected as the basis of the Banks Peninsula Macroinvertebrate Index (BPMI) (Table 6.3). Metrics were eliminated if they did not correctly assign 100% of the selected impaired sites as degraded. The composition metrics, % Coleoptera and % Odonata, were not used because of the overall low abundances of taxa within these groups in the streams.

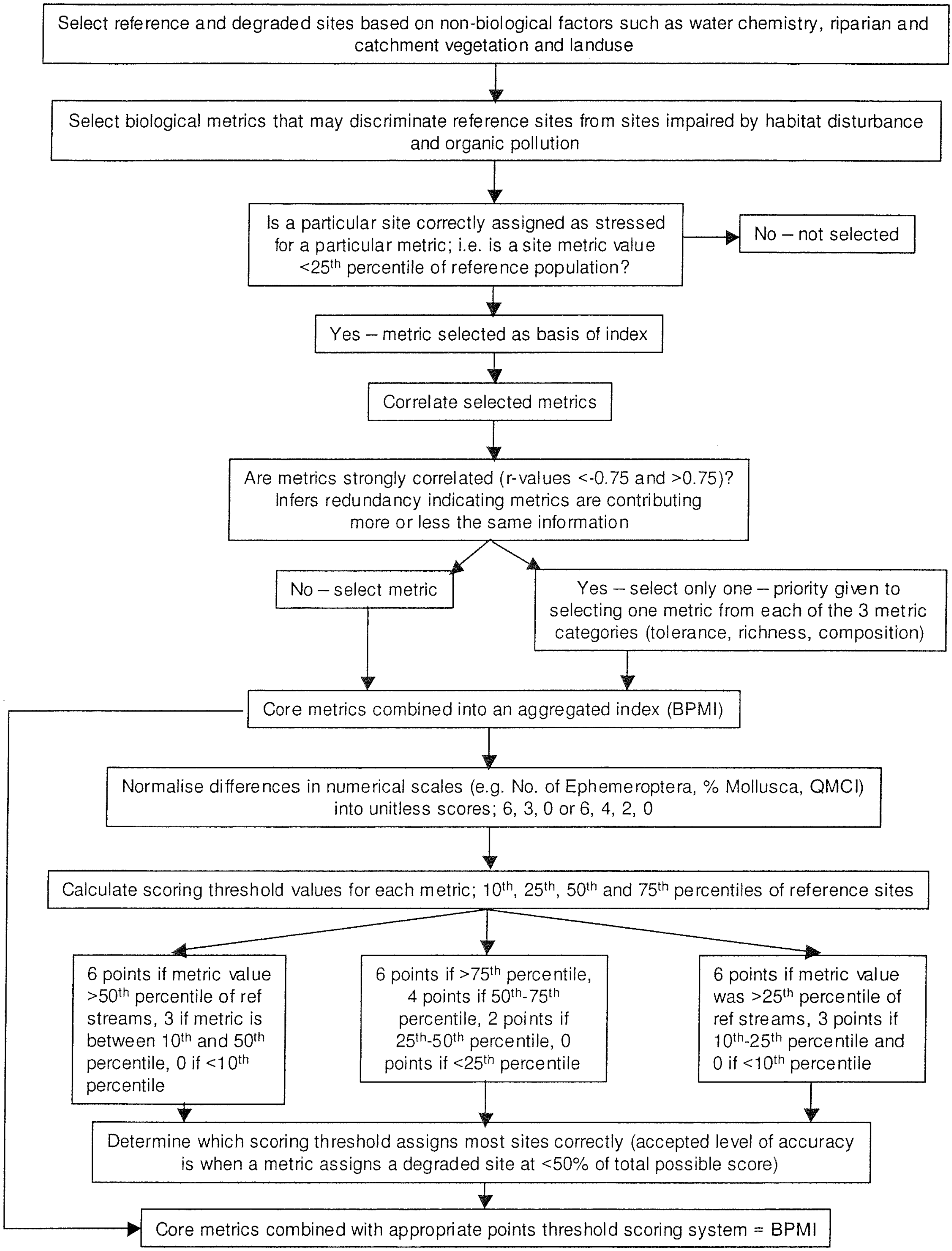


Fig. 6.2 Summary flow diagram of the steps taken in creating the BPMI

Table 6.3 Percentage of degraded sites correctly assigned as degraded (<25th percentile of reference population) for the 29 metrics. Metric types include composition (C) richness (R) and tolerance (T). EPT = Ephemeroptera, Plecoptera and Trichoptera. The 16 metrics selected as the basis of the Banks Peninsula Macroinvertebrate Index (BPMI) are shown.

Metric	**Type**	**% of sites correctly assigned as degraded**	**Selected**
% EPT in sample	C	100.0	X
% EPT in sample[2]	C	100.0	X
% EPT in sample[3]	C	100.0	X
% Ephemeroptera	C	100.0	X
% Plecoptera	C	0.0	
% Trichoptera	C	100.0	X
% Diptera	C	44.4	
% Coleoptera	C	100.0	
% Chironomidae	C	100.0	X
% Crustacea	C	77.8	
% Odonata	C	100.0	
% Mollusca	C	100.0	X
% Oligochaeta	C	88.9	
% Dominant taxa	C	44.4	
No. EPT taxa	R	55.6	X
No. Ephemeroptera	R	100.0	X
No. Plecoptera	R	0.0	
No. Trichoptera	R	44.0	
No. Taxa	R	44.0	
MCI	T	100.0	X
QMCI	T	100.0	X
SQMCI	T	100.0	X
Ordinal QMCI (OQMCI)	T	100.0	X
EPT taxa as % of total taxa	T	44.4	
%EPT / %Mollusc + %Oligochaeta	T	100.0	X
%EPT / %Oligochaeta	T	100.0	X
%EPT / %Mollusc	T	88.9	
% intolerant (MCI>=7)	T	100.0	X
% tolerant (MCI <=3)	T	100.0	X

[2] Excluding *Oxyethira* and *Paroxyethira*
[3] Excluding *Oxyethira*, *Paroxyethira* and *Aoteapsyche*

6.3.2 Development of the Banks Peninsula Macroinvertebrate Index

Spearman rank correlation analysis was performed to determine the amount of redundancy between the different metrics (Table 6.4). A strong correlation was found between QMCI and MCI, QMCI and SQMCI, and MCI and SQMCI (Table 6.4) indicating redundancy of information. The QMCI was selected over the MCI since it reflects numerical changes in the composition of macroinvertebrate communities and is therefore more likely to pick up subtle changes in community structure (Chapter 4). Additionally, the QMCI is likely to be more robust than the

SQMCI since it uses quantitative rather than semi-quantitative abundance data. The QMCI was strongly correlated with % tolerant taxa (r -0.80) and therefore the latter was omitted too. % EPT was selected because of its low redundancy with QMCI and because of its general applicability to a range of streams. % intolerant taxa, % EPT^2, % EPT^3, % Ephemeroptera, % Trichoptera and % EPT / % Mollusca + % Oligochaeta were not selected because of their strong correlations with % EPT (Table 6.4). Number of Ephemeroptera, % Mollusca and % Chironomidae were selected because of their low redundancies with all other metrics, indicating that they were providing additional information. Additionally, number of Ephemeroptera was selected because it was the sole richness metric.

Table 6.4 Spearman rank correlation coefficients between the 16 candidate metrics for the Banks Peninsula Macroinvertebrate Index (BPMI). Metric types, tolerance (T), composition (R) and richness (R) are also indicated. **Bold, r-values <0.75 and >0.75.**

	Type	MCI	QMCI	SQMCI	Ordinal QMCI (OQMCI)	% EPT in sample	% EPT in sample^2	% EPT in sample^3	% Ephemeroptera	% Trichoptera	% Chironomidae	% Mollusca	%EPT / %Mollusca + %Oligochaeta	%EPT / %Oligochaeta	% Intolerant (MCI>7)	% Tolerant (MCI<3)
MCI	T															
QMCI[a]	T	0.63														
SQMCI	T	**0.75**	**0.97**													
Ordinal QMCI (OQMCI)	T	0.45	0.53	0.50												
% EPT in sample[a]	C	0.47	0.45	0.43	**0.98**											
% EPT in sample^2	C	0.52	0.48	0.45	**0.97**	**0.98**										
% EPT in sample^3	C	0.52	0.48	0.45	**0.97**	**0.98**	**1.00**									
% Ephemeroptera	C	0.53	0.40	0.43	**0.87**	**0.88**	**0.85**	**0.85**								
% Trichoptera	C	0.47	0.45	0.43	**0.98**	**1.00**	**0.98**	**0.98**	**0.88**							
% Chironomidae[a]	C	0.17	-0.32	-0.17	0.28	0.32	0.25	0.25	0.50	0.32						
% Mollusca[a]	C	0.17	0.37	0.37	-0.53	-0.60	-0.57	-0.57	-0.47	-0.60	-0.60					
%EPT/%Mollusc+%Oligochaeta	T	0.33	0.30	0.30	**0.95**	**0.97**	**0.93**	**0.93**	**0.83**	**0.97**	0.45	**-0.75**				
%EPT%Oligochaeta	T	**0.93**	0.70	**0.77**	0.62	0.63	0.68	0.68	0.58	0.63	0.13	0.02	0.50			
% Intolerant (MCI>7)	T	0.43	0.33	0.28	**0.92**	**0.93**	**0.97**	**0.97**	**0.82**	**0.93**	0.28	-0.63	**0.90**	0.57		
% Tolerant (MCI<3)	T	-0.27	**-0.80**	-0.72	-0.03	0.07	0.03	0.03	0.15	0.07	0.73	**-0.75**	0.23	-0.30	0.17	
No. Ephemeroptera[a]	R	0.60	0.24	0.37	0.47	0.52	0.52	0.52	0.67	0.52	0.37	-0.30	0.52	0.45	0.52	0.22

[a] 5 core metrics selected for the Banks Peninsula Macroinvertebrate Community Index (BPMI)

The BPMI was constructed by summing the macroinvertebrate metrics that proved responsive to disturbance (i.e. assigned a high percentage of sites correctly as degraded or reference). Initially, differences in numerical scales (e.g. number of Ephemeroptera, % EPT, QMCI) were normalised into unitless scores. Scoring thresholds were determined by calculating the 10th, 25th, 50th and 75th percentiles for the reference site populations for each metric. Points were assigned to each section using either a 6, 3, 0 or a 6, 4, 2, 0 points system. Three scoring thresholds for each metric were considered initially following Maxted et al. (2000). The first method assigned 6 points to a stream if its metric value was >50th percentile of the reference streams, 3 points if the metric was between the 10th and 50th percentile, and 0 points if <10th percentile. The 75th, 50th and 25th percentile method assigned 6 points if a site was >75th percentile, 4 points if 50th-75th percentile, 2 points if 25th-50th percentile and 0 points if <25th percentile. The 25th and 10th percentile method assigned 6 points to a metric if its value was >25th percentile of reference streams, 3 points if 10th-25th percentile and 0 points if <10th percentile. The point scoring system that assigned the most sites correctly was selected for the Banks Peninsula Macroinvertebrate Index (BPMI). The accepted level of accuracy for the threshold points scoring system was when a metric assigned a degraded site at <50% of the total possible score (Maxted et al. 2000).

The 50th and 10th percentile method assigned the highest proportion of degraded sites correctly (77.8% degraded; Table 6.5) and was therefore selected for allocating points. Thus, 6 points were assigned to a metric if its value was >50th percentile of the reference population, 3 points if the metric value was between the 10th and 50th percentile and 0 points if <10th percentile of the reference population. The other two methods both assigned 66.7% of the sites, correctly (Table 6.5) and were therefore inferior to the 10-50 method. Summary statistics and scoring threshold values used to calculate the BPMI are shown in Table 6.6.

Table 6.5 Percentage of stressed sites correctly assigned as stressed (<50% of the total possible points) using the three threshold points scoring methods. See text and Fig. 6.2 for calculations.

	% of stressed sites correctly assigned		
n	50th and 10th	75th, 50th and 25th	25th and 10th
9	77.78	66.67	66.67

Table 6.6 Summary statistics for reference sites and scoring thresholds used to calculate the Banks Peninsula Macroinvertebrate Index (BPMI). The 50th and 10th percentile method was used to derive the three scoring thresholds.

	Statistics				Scoring thresholds		
	Median	25 percentile	75 percentile	Range	6 points	3 points	0 points
QMCI	5.9	5.4	6.0	4.8 - 6.1	>5.9	5.1 - 5.9	<5.1
% EPT	59.1	49.3	64.2	33.0 - 68.2	>59.1	41.8 - 59.1	<41.8
%Chironomidae	7.5	2.7	16.4	1.4 - 31.4	<1.6	1.6 - 7.5	>7.5
%Mollusca	18.6	15.5	30.5	4.2 - 34.3	<9.5	9.5 - 18.6	>18.6
Total no. Ephemeroptera	2.0	2.0	2.0	1.0 - 3.0	>2	2	1

QMCI values were significantly higher for the reference streams ($P<0.001$) with no overlap in reference and degraded interquartile ranges (Fig. 6.3, Table 6.7). Percent EPT was also significantly higher for reference streams ($P<0.001$), again with no overlap in the interquartile ranges of the two stream groups, although some overlap of maximum and minimum values. Both percent Chironomidae and percent Mollusca did not differ significantly between the two groups ($P=0.24$, $P=0.057$, respectively; Table 6.7). In both cases there was moderate overlap of interquartile ranges but at least one median was outside the interquartile range overlap (Fig. 6.3). Number of ephemeropteran taxa differed significantly between degraded and reference conditions, however ($P<0.05$; Table 6.7), although overlap was found between the reference minimum and the degraded interquartile range (Fig. 6.3). Overall BPMI values clearly separated reference sites from degraded sites with no overlap of interquartile ranges (Fig. 6.3).

Strong correlations between the BPMI and MCI ($r = 0.92$, $P<0.001$) and BPMI and QMCI ($r = 0.84$, $P<0.001$) show that all three methods assessed the health of Banks Peninsula streams (Fig. 6.4) in a very similar way.

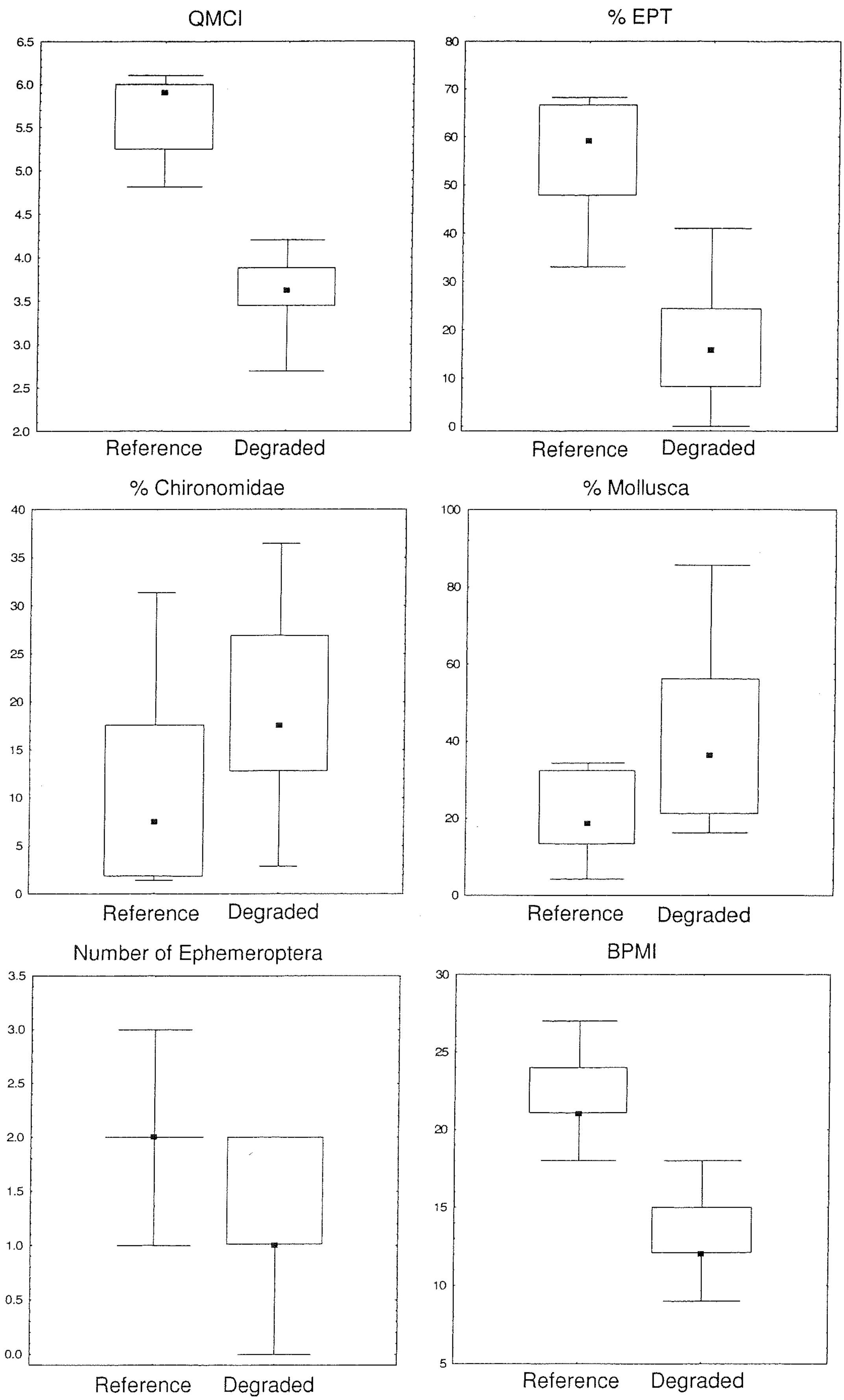


Fig. 6.3 Distributions of 5 core metrics and BPMI showing variability and sensitivity to degradation. Box represents 25th and 75th percentiles, dot represents median value and whiskers represent maximum and minimum values.

Table 6.7 Summary of significant differences (1-way ANOVA) between reference and degraded sites, for individual metrics and the combined index (BPMI). **Bold, P<0.05.**

	df	MS	F	p
QMCI	1	17.15	68.98	**<0.0001**
% EPT	1	5905.62	36.09	**<0.0001**
% Chironomidae	1	180.52	1.47	0.2448
% Mollusca	1	1701.78	4.31	0.0567
No. Ephemeroptera	1	2.38	6.00	**0.0280**
BPMI	1	339.51	33.61	**<0.0001**

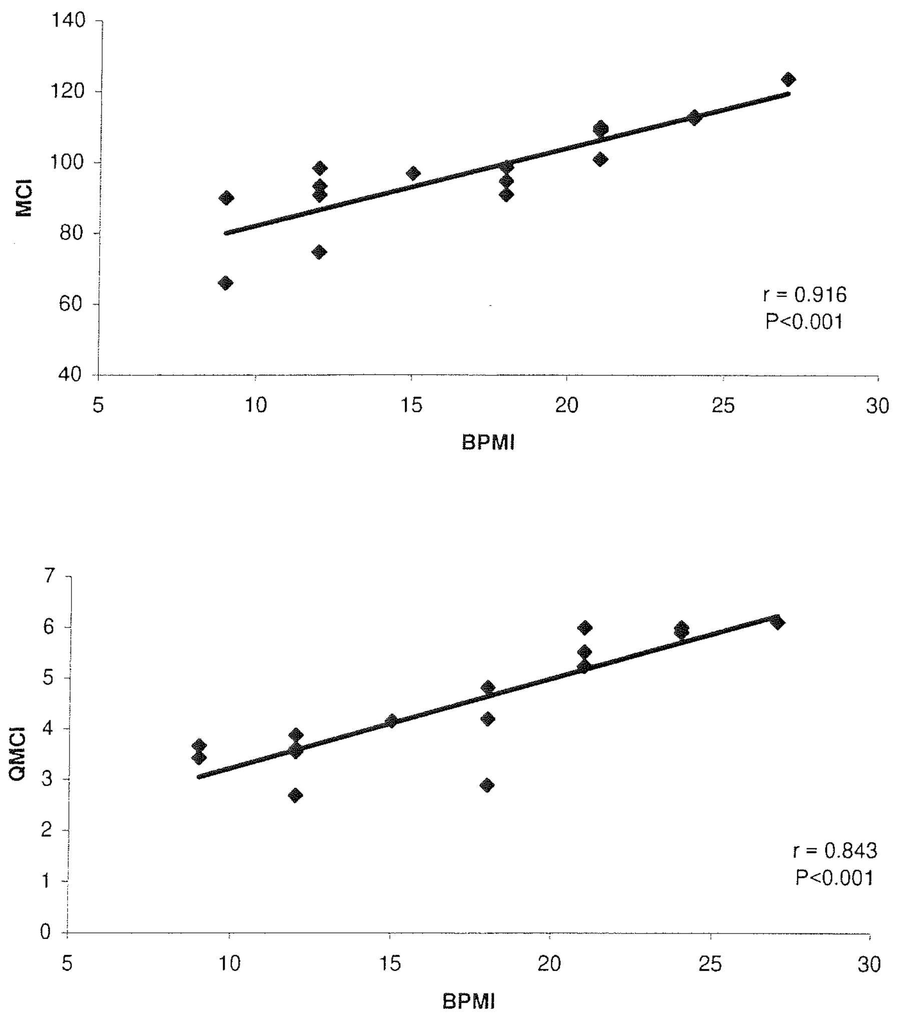


Fig. 6.4 Linear relationships between BPMI and MCI and QMCI.

6.4 DISCUSSION

The primary objective of this study was to select biological metrics that best discriminate reference sites from sites impaired by habitat disturbance and organic pollution in Banks Peninsula streams, and combine them into an index of biological integrity (BPMI). Five core metrics from three structural and functional categories were selected from an initial set of 29 for the BPMI. The 50th and 10th percentiles were selected as levels for defining metric scores (6, 3, 0) and scoring thresholds for the Banks Peninsula streams (Table 6.6). ANOVA and box-and-whisker graphs showed that QMCI, %EPT, No. Ephemeroptera and the BPMI were particularly important in discriminating reference and degraded sites, and strong correlations between the BPMI and MCI/QMCI showed that these three indices gave very similar assessments for the Banks Peninsula streams.

The 29 metrics initially tested were similar to those incorporated in some North American (Reynoldson et al. 1995, Barbour et al. 1996, Maxted et al. 2000) and New Zealand (Quinn et al. 1997, Collier et al. 1998) macroinvertebrate multivariate indices. Three of the selected metrics were measures of composition, which provide information on proportions of higher taxonomic groups in the total fauna. Percent EPT separated degraded and reference sites well, consistent with the findings of Collier et al. (1998). They found that % EPT was a useful indicator of biological quality in lowland Waikato streams, being relatively insensitive to the habitat from which the sample was taken, and little affected by sampling intensity. Percent Chironomidae was the least successful of the five core metrics in discriminating reference sites from degraded sites, possibly because of the small number of sites used, or because of differences in species composition within chironomid samples. Relative abundance of chironomids generally increases with increasing stream degradation (Chapter 3), and, Quinn et al. (1997) found that chironomid density was significantly higher in agriculturally impacted, Waikato hill-country streams. Harding et al. (1999) noted that chironomid species richness was significantly higher in downstream reaches of the Pomahaka River (Southland) where agricultural intensity was highest. Percent Mollusca was a relatively strong discriminator within the Banks Peninsula dataset and the increase in relative abundance of mollusc taxa at degraded sites is consistent with the results of several other New Zealand studies. For example, Scott et al. (1994) found that numbers of *Potamopyrgus*, *Physastra* and *Gyraulus* all tended to increase in developed pasture areas in 3 Southland streams, and Quinn et al. (1990a) reported that *Potamopyrgus* abundance was high in lowland rivers with developed catchments, throughout New Zealand. Similarly, Harding and Winterbourn (1995) found that molluscs were most prolific in pastorally developed streams near Hanmer and Hirsch (1958) found that a molluscan fauna was commonly present at moderately polluted streams throughout New Zealand.

One richness metric, the number of Ephemeroptera taxa, was selected as part of the core set for use in the BPMI. This metric was also included in the rapid bioassessment index for Oregon streams (Fore and Karr 1996), and on Banks Peninsula it distinguished disturbed sites from reference sites. Elsewhere in New Zealand, Harding et al. (1999) showed that the number of ephemeropteran taxa was significantly higher at less agriculturally impacted sites on the Pomahaka River, and Harding and Winterbourn (1995) found that more Ephemeroptera taxa occurred in beech forests than pastoral streams near Hanmer, North Canterbury. Furthermore, significantly lower numbers of ephemeropteran taxa were found at sites with >30% catchment development in a study of 88 New Zealand streams (Quinn and Hickey 1990a). Hirsch (1958) concluded that only a low degree of organic pollution was required before sensitive mayfly taxa were lost from New Zealand streams.

The sole tolerance metric selected for the BPMI was the Quantitative Macroinvertebrate Community Index. This commonly used biotic index assigns tolerance scores to taxa based on their perceived sensitivity to organic enrichment (Stark 1985). The QMCI was amongst the strongest discriminator of degraded from reference conditions (Table 6.7), consistent with the findings of several other New Zealand studies (Quinn and Hickey 1990a, Quinn et al. 1992, Scott et al. 1994, Quinn et al. 1997, Collier et al. 1998)

Finally, although number of taxa is a metric commonly used to infer stream condition (Plafkin et al. 1989, Quinn and Hickey 1990a, Barbour et al. 1996, Fore and Karr 1996, Collier et al. 1998, Maxted et al. 2000), I found it was a poor indicator of degradation (Table 6.3). An inherent problem with the taxon richness metric is the great variability in communities among relatively similar streams or habitats (Chapter 3) and therefore the samples taken from them. Furthermore, the samples taken will inevitably be only a proportion of the taxa actually present.

6.4.1 Aggregation of metrics into the Banks Peninsula Macroinvertebrate Index

The metrics that best discriminated reference sites from sites impaired by habitat disturbance and organic pollution were aggregated to produce the Banks Peninsula Macroinvertebrate Index (BPMI). Although, further testing is required before definitive threshold values can be established, I consider that streams with BPMI scores <15 can be considered to be in poor condition, those with scores ranging from 15-20 in moderate condition, those with scores ranging from 20-25 in good condition, and those >20 excellent.

While the summed BPMI and other similar indices give an overview of stream condition and can inform managers where action is needed, the exact nature of such an action (e.g., restoration, mitigation) must be determined by analysis of individual metrics and potential stressors (Barbour et al. 1996). Therefore, assessments should show both summed index scores and individual scores. For example, low Ephemeroptera numbers and high % Mollusca scores are common responses to high levels of sedimentation (Chapter 3), suggesting that management or mitigation to reduce silt inputs may be necessary.

Metric classifications such as the BPMI have rarely been developed in New Zealand, but they are relatively common in North America. One difference between many of the North American indices and the present index is that the former often include a macroinvertebrate feeding metric (Barbour et al. 1996, Fore and Karr 1996, Maxted et al. 2000). Such a component was not considered useful in this study because of the generalist feeding nature of many New Zealand invertebrate groups (Winterbourn 2000). For example, a majority of non-predatory stream invertebrates are usually classified as browsers, or collector-browsers, whereas the shredder and filter-feeding functional feeding groups are poorly represented (Harding et al. 1997). Additionally, the fact that some species span feeding groups (e.g. the facultative shredders/collector-browsers *Olinga* and *Austroperla*, Quinn et al. 1997) or change feeding modes as they get larger (e.g. Winterbourn (1974), found that small *Stenoperla* nymphs were detritorvous, whereas large ones were carnivores) suggests that functional feeding groups provide little reliable information for inclusion in New Zealand metric indices.

6.4.2 Application of BPMI in other regions

The BPMI was designed for use in Banks Peninsula streams and its application to other ecoregions in New Zealand may not be appropriate. Recent research in the Canterbury and Waikato regions has indicated that the grouping of streams according to similarities in source of flow, geology and landuse rather than ecoregion may be most appropriate for comparative purposes, especially for the selection of reference sites (Suren et al. 2000). Reference streams must be carefully selected for each area of interest (a potential problem for lowland Canterbury streams where 'reference' sites may not exist), and biological metrics that best discriminate reference sites from impaired sites chosen. Although the five metrics selected for the BPMI were strong discriminators, others may need to be used elsewhere. For example, relative abundance of Ephemeroptera appears to be a relatively strong discriminator of stream condition for the whole of the Canterbury region (Chapter 3). Percent Mollusca and percent Oligochaeta were identified as groups characteristic of sites associated with organic pollution in three Southland streams (Scott et al. 1994), indicating that they may be appropriate metrics to use in that region. Similarly, in the Taieri River, the role of browsers and shredders in the stream community is dependent on landuse (Townsend et al. 1997), indicating that functional feeding groups could be worth including in Otago.

6.4.3 Comparison of the BPMI, and the MCI and QMCI

The strong relationship between the BPMI and MCI, and BPMI and QMCI indicates that these three metrics evaluate the condition of sites, consistently. Furthermore, 12 of 16 sites were assigned to equivalent degradation categories by the BPMI and the MCI (see Chapter 4), and 15 of 16 sites to comparable BPMI and QMCI degradation categories. These strong relationships suggest that the extra effort required to produce a multimetric index did not result in an improved assessment of stream condition over that provided by the MCI or QMCI on Banks Peninsula streams. In fact, the QMCI discriminated reference from degraded sites more clearly than the BPMI (Fig. 6.3). However, it did provide insightful information, through the individual metric components, on what factors were likely to be causing the degradation, and where restoration or mitigation should be focussed.

Lastly, it needs to be noted that questions have been raised over the validity of adding groups of indices to form a single measure of stream health. Additivity assumes that the chosen metrics potentially vary in the same way and with similar magnitude in response to human activities, and that the indices are independent (Coysh and Norris 1999). In the present study, indices were calculated from the same raw data, and even though only one of a strongly correlated set of metrics was selected (to reduce redundancy), the final core group of metrics was still quite highly correlated (Table 6.4). If only indices that show a response to human activities are selected and they are not independent, the trends shown by one will automatically be shown by the others. Rather than providing a better overall assessment score, I suggest that the greatest potential value of a multimetric index is in the additional information provided by the individual metrics.

SUMMARY

1. **In the summer** of 1999-2000 Environment Canterbury undertook a wide-ranging macroinvertebrate and physico-chemical survey of 230 3rd and 4th order streams throughout the Canterbury region. Data were collected by kick-net and spot water sampling in association with habitat surveys. The information obtained fulfils some of the requirements for State of the Environment and Resource Management Act monitoring of rivers and streams. I was actively involved in all parts of the survey and was given the opportunity to use the data to develop into a thesis.

2. **A total of** 125 taxa (family, genus, species) were identified from the 230 sites. Aquatic insects formed the majority (90) with most belonging to the orders Diptera (25), Trichoptera (24), Ephemeroptera (11) and Plecoptera (10). Oligochaeta and the leptophelebid mayfly *Deleatidium* were the most frequently occurring taxa, being present at 208 and 207 of the sites, respectively.

3. **Invertebrate community composition** was dependent on where on an environmental gradient the site fell. Two overriding factors appeared to influence community composition; the physical condition of the stream, and the amount of anthropogenic development within the catchment. Faunas dominated by Ephemeroptera and Trichoptera, and with some Plecoptera present tended to occur in pristine, high altitude streams with low conductivity, well vegetated riparian zones, heterogeneous streambed substrates and periphyton consisting primarily of diatoms. Fauna dominated by Crustacea, Oligochaeta and Chironomidae occurred commonly at degraded lowland sites with high conductivity, little or no riparian vegetation, more homogeneous fine substrates and periphyton dominated by thick mats and filaments. Between these two extremes gradual change in faunas was found, with Trichoptera dominating intermediately disturbed sites. A striking decrease in the relative abundance of Ephemeroptera along an ecological gradient appeared to be associated with increasing intensity of landuse.

4. **Inconsistencies in the** ways the MCI, QMCI and SQMCI assess stream health in Canterbury are probably brought about by two factors. Firstly, presence-absence data used in calculating the MCI may not detect subtle differences in community structure that are picked up by the quantitative (QMCI) and semi-quantitative (SQMCI) variants of the index. Secondly, and probably more importantly, published degradation bands for the MCI, QMCI and SQMCI do not appear to be directly comparable in Canterbury.

5. **The development of** a quantitative MCI with low-level (order, class, phylum) identification (OQMCI) was found to be a potentially viable alternative to MCI and its derivatives when a low-cost, rapid assessment technique is needed, but expertise in identification is lacking.

6. **The MCI indicated** that Canterbury streams were generally more healthy if they were further inland, at higher altitudes, and in forested or unmodified catchments. Stream health was poorest at lowland sites with developed catchments, although 42 pastoral sites with MCI values >100 and taxonomic richness >25 (per 100 individuals sorted with a scan for rare taxa) indicated that healthy streams were attainable in agriculturally developed land.

7. **The development of** a multimetric approach for assessing the health of Banks Peninsula streams identified five biological metrics that best discriminated selected reference sites from sites impaired by habitat disturbance and organic pollution. QMCI, % EPT, % Chironomidae, % Mollusca and No. Ephemeroptera were combined into an index of biological integrity; the Banks Peninsula Macroinvertebrate Index (BPMI). Strong relationships between the BPMI and MCI and QMCI suggested that the extra effort required to produce a multimetric index did not result in improved assessment of stream condition. However, a multimetric index can provide additional information on the source of degradation to a stream and indicate where restoration or mitigation should be focussed.

ACKNOWLEDGEMENTS

A study of this nature is not possible without the help, support and encouragement of many people. Firstly, my sincere thanks go out to my supervisor Professor Mike Winterbourn for his guidance, time, editing skills and willingness to share his knowledge. I feel extremely lucky to be amongst the last students to have had Mike as a supervisor.

The data used in this thesis was kindly provided by Environment Canterbury. I am very grateful for this support, in particular I would like to thank Adrian Meredith, Malcolm Main and Ken Taylor. My thanks also go to Rochelle Lavender, Melissa Anthony, Darren Cottam, Michele Widdowson, Bruce Digby and Michael de Joux who unbeknown to them were my field and laboratory assistants.

I would like to thank Jon Harding for his advice and proof reading, Angus McIntosh for acting as co-supervisor and Alistair Suren for the QA/QC information provided. Thanks to the technical staff of the Zoology Department (Nick Etheridge, Bruce Lingard, Jo-Anne Burke, Rene Bishop, Franz Ditz and Victor Mencel) for help with field equipment and field assistance.

Finally, I am grateful to fellow students, friends and family. Thanks to Michele, Russell, Chris, Greg, Leanne, and FERG members for their advice and assistance. I am extremely grateful to my family and Jo who encouraged and supported me throughout this study, and my university career.

REFERENCES

Allan, J. D., D. L. Erikson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. *Freshwater Biology* 37: 149-161.

Angradi, T. R. 1999. Fine sediment and macroinvertebrate assemblages in Appalachian streams: a field experiment with biomonitoring applications. *Journal of the North American Benthological Society* 18: 49-66.

Armstrong, P. A. 1996. The effects of stream stability and flow variability on the production of aquatic invertebrates. Unpublished M.Sc thesis. University of Canterbury, Christchurch, New Zealand.

Barbour, M. T., and J. Gerritsen. 1996. Subsampling of benthic samples: a defence of the fixed-count method. *Journal of the North American Benthological Society* 15: 386-391.

Barbour, M. T., J. Gerritsen, G. E. Griffith, R. Frydenborg, E. McCarron, J. S. White, and M. L. Bastian. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. *Journal of the North American Benthological Society* 15: 185-211.

Barbour, M. T., J. B. Stribling, and J. R. Karr. 1995. *Multimetric approach for establishing biocriteria and measuring biological condition.* In: W. S. Davis and T. P. Simon, Biological assessment and criteria. Lewis Publishers, Boca Raton, Florida.

Biggs, B. F., M. J. Duncan, I. G. Jowett, J. M. Quinn, C. W. Hickey, R. J. Davies-Colley, and M. E. Close. 1990. Ecological characterisation, classification, and modelling of New Zealand rivers: an introduction and synthesis. *New Zealand Journal of Marine and Freshwater Research* 24: 277-304.

Biggs, B. J. F., C. Kilroy, and C. M. Mulcock. 1998. New Zealand Stream Health Monitoring and Assessment Kit. *Stream Monitoring Manual. Version 1.* NIWA Technical Report 40.

Bis, B., A. Zdanowicz, and M. Zalewski. 2000. Effects of catchment properties on hydrochemistry, habitat complexity and invertebrate community structure in a lowland river. *Hydrobiologia* 442/443: 369-387.

Boothroyd, I. 1999. Macroinvertebrate monitoring: alternate methods. In: The use of macroinvertebrates in water management. Ministry for the Environment.

Boothroyd, I., and J. Stark. 2000. *Use of invertebrates in monitoring.* In: K. J. Collier and M. J. Winterbourn, New Zealand stream invertebrates: ecology and implications for management. New Zealand Limnological Society, Christchurch, New Zealand.

Boothroyd, I. K. G. 2000. *Biogeography and biodiversity.* In: K. J. Collier and M. J. Winterbourn, New Zealand stream invertebrates: ecology and implications for management. New Zealand Limnological Society, Christchurch, New Zealand.

Brown, L. M. 1998. The effects of the Cobb River hydroelectiric power scheme on downstream macroinvertebrate communities. Unpublished M.Sc thesis. University of Canterbury, Christchurch, New Zealand.

Bunn, S. E., P. M. Davies, D. M. Kellaway, and I. P. Prosser. 1998. Influence of invasive macrophytes on channel morphology and hydrology in an open tropical lowland stream, and potential control by riparian shading. *Freshwater Biology* 39: 171-178.

Cao, Y., A. W. Bark, and W. P. Williams. 1996. Measuring the responses of macroinvertebrate communities to water pollution: a comparison of multivariate approaches, biotic and diversity indices. *Hydrobiologia* 314: 1-19.

Carter, J. L., S. V. Fend, and S. S. Kennelly. 1996. The relationships among three habitat scales and stream benthic invertebrate community structure. *Freshwater Biology* 35: 109-124.

CCC. 1980. A biological survey of rivers in the metropolitan Christchurch area and outlying districts; The Avon, Heathcote and Styx Rivers and their tributaries. Christchurch Drainage Board. Christchurch City Council Drainage and waste management.

Chapman, M. A., and M. H. Lewis. 1976. *An introduction to the freshwater Crustacea of New Zealand.* Collins, Auckland.

Charvet, S., B. Statzner, P. Usseglio-Polatera, and B. Dumont. 2000. Traits of benthic macroinvertebrates in semi-natural French streams: an initial application to biomonitoring in Europe. *Freshwater Biology* 43: 277-296.

Chessman, B. C. 1995. Rapid assessment of rivers using macroinvertebrates: a procedure based on habitat-specific sampling, family level identification and a biotic index. *Australian Journal of Ecology* 20: 122-129.

Chutter, F. M. 1969. The effects of silt and sand on the invertebrate fauna of streams and rivers. *Hydrobiologia* 34: 54-76.

Clenaghan, C., P. S. Giller, J. O'Halloran, and R. Hernan. 1998. Stream macroinvertebrate communities in a conifer-afforested catchment in Ireland: relationships to physico-chemical and biological factors. *Freshwater Biology* 40: 175-193.

Close, M. E., and R. J. Davies-Colley. 1990a. Baseflow water chemistry in New Zealand rivers. 1. Characterisation. *New Zealand Journal of Marine and Freshwater Research* 24: 319-341.

—. 1990b. Baseflow water chemistry in New Zealand rivers. 2. Influences of environmental factors. *New Zealand Journal of Marine and Freshwater Research* 24: 319-341.

Cobb, D. G., T. D. Galloway, and F. J. Flannagan. 1992. Effects of discharge and substrate stability on density and species composition of stream insects. *Canadian Journal of Fisheries and Aquatic Sciences* 49: 1788-1795.

Collier, K. J. 1989. Downstream impacts of pakihi development. Department of Conservation Science and Research Internal Report No. 42.

—. 1995. Environmental factors affecting the taxonomic composition of aquatic macroinvertebrate communities in lowland waterways of Northland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 29: 453-465.

Collier, K. J., and B. J. Smith. 1998. Dispersal of adult caddisflies (Trichoptera) into forests along three New Zealand streams. *Hydrobiologia* 361: 53-65.

Collier, K. J., K. J. Wilcock, and A. S. Meredith. 1998. Influence of substrate type and physico-chemical conditions on macroinvertebrate faunas and biotic indices of some lowland Waikato, New Zealand streams. *New Zealand Journal of Marine and Freshwater Research* 32: 1-19.

Cook, L. 1999. A Regional Profile; Canterbury. Statistics New Zealand. Catalogue No. 01.716.IN98.

Corn, P. S., and R. B. Bury. 1989. Logging in Western Oregon: a response of headwater habitats and stream amphibians. *Forest Ecology and Management* 29: 39-57.

Cottam, D. 1999. The importance of riparian cover and allochthonous inputs for the biological health of Christchurch waterways. Unpublished M.Sc thesis. University of Canterbury, Christchurch, New Zealand.

Courtemanch, D. L. 1996. Commentary on the subsampling procedures used for rapid bioassessments. *Journal of the North American Benthological Society* 15: 381-385.

Coysh, J., and R. Norris. 1999. Waikato Region AUSRIVAS/RIVPACS trial: Final Report SMF Contract No. 5091. Ministry for the Environment, New Zealand.

Cummins, K. W. 1974. Structure and function of stream ecosystems. *Bioscience* 24: 631-641.

Davies-Colley, R. J. 1997. Stream channels are narrower in pasture than in forest. *New Zealand Journal of Marine and Freshwater Research* 331: 599-608.

Davies-Colley, R. J., and M. E. Close. 1990. Water colour and clarity of New Zealand rivers under baseflow conditions. *New Zealand Journal of Marine and Freshwater Research* 24: 357-365.

Davies-Colley, R. J., and J. M. Quinn. 1998. Stream lighting in five regions of North Island, New Zealand: control by channel size and riparian vegetation. *New Zealand Journal of Marine and Freshwater Research* 32: 591-605.

de Lisle, J. F. 1969. *The climate and weather.* In: G. A. Knox, The natural history of Canterbury. A.H. and A.W. Reed, Wellington, New Zealand.

Death, R. G. 1991. Environmental stability: its effect on stream benthic communities. Unpublished Ph.D thesis. University of Canterbury, Christchurch, New Zealand.

—. 2000. *Invertebrate-substratum relationships.* In: K. J. Collier and M. J. Winterbourn, New Zealand stream invertebrates: ecology and implications for management. New Zealand Limnological Society, Christchurch, New Zealand.

Death, R. G., and M. J. Winterbourn. 1994. Environmental stability and community persistence: a multivariate perspective. *Journal of the North American Benthological Society* 13: 125-139.

Dewdney, M. D. 2000. The effect of pastoral clearings and a return to forest on two West Coast rivers: implications for reforestation and management. University of Canterbury, Christchurch, New Zealand.

Doledec, S., B. Statzner, and M. Bournard. 1999. Species traits for future biomonitoring across ecoregions: patterns along a human-impacted river. *Freshwater Biology* 42: 737-758.

Donald, R. 1995. Environment B.O.P natural environment regional monitoring network river ecology programme 1992-1995. Environment Report 95/16.

Dunning, K. J. 1998. Effects of exotic forestry on stream macroinvertebrates: the influence of scale in North Island, New Zealand streams. Unpublished M.Sc thesis. Massey University, Palmerston North.

Edwards, J. S., and P. Sugg. 1993. Arthropod fallout as a resource in the recolonization of Mount St. Helens. *Ecology* 74: 954-958.

Fitzharris, B. B., G. D. Mansergh, and J. M. Soons. 1992. *Basins and lowlands of the South Island.* In: J. M. Soons and M. J. Selby, Landforms of New Zealand. Longman Paul, Hong Kong.

Fore, L. S., and J. R. Karr. 1996. Assessing invertebrate responses to human activities: evaluating alternate approaches. *Journal of the North American Benthological Society* 15: 212-231.

Friberg, N., M. J. Winterbourn, K. A. Shearer, and S. E. Larson. 1997. Benthic communities of forest streams in the South Island, New Zealand: effects of forest type and location. *Archiv für Hydrobiologie* 138: 289-306.

Gage, M. 1969. *Rocks and landscape.* In: G. A. Knox, The natural history of Canterbury. A.H. and A.W. Reed, Wellington, New Zealand.

Gerritsen, J. 1995. Additive biological indices for resource management. *Journal of the North American Benthological Society* 14: 451-457.

Gray, L. J., and J. V. Ward. 1982. Effects of sediment releases from a reservoir on stream macroinvertebrates. *Hydrobiologia* 96: 177-184.

Harding, J. S. 1990. The effects of impoundment on three Westland river macroinvertebate communities. Unpublished M.Sc thesis. University of Canterbury, Christchurch, New Zealand.

—. 1994. Lotic ecoregions of New Zealand. Unpublished Ph.D thesis. University of Canterbury, Christchurch, New Zealand.

Harding, J. S., and M. J. Winterbourn. 1995. Effects of contrasting land use on physico-chemical conditions and benthic assemblages of streams in a Canterbury (South Island, New Zealand) river system. *New Zealand Journal of Marine and Freshwater Research* 29: 479-492.

Harding, J. S., M. J. Winterbourn, and F. McDiffett. 1997. Stream faunas and ecoregions in South Island, New Zealand: do they correspond? *Archiv für Hydrobiologie* 140: 289-307.

Harding, J. S., R. G. Young, J. W. Hayes, K. A. Shearer, and J. D. Stark. 1999. Changes in agricultural intensity and river health along a river continuum. *Freshwater Biology* 42: 345-357.

Hawkins, C. P., J. N. Hogue, L. M. Decker, and J. W. Feminella. 1997. Channel morphology, water temperature, and assemblage structure of stream insects. *Journal of the North American Benthological Society* 16: 728-749.

Haynes, A. 1999. The long term effect of forest logging on the macroinvertebrates in a Fijian stream. *Hydrobiologia* 405: 79-87.

Hirsch, A. 1958. Biological evaluation of organic pollution of New Zealand streams. *New Zealand Journal of Science* 1: 500-553.

Jacobsen, D., R. Schultz, and A. Encalada. 1997. Structure and diversity of stream invertebrate assemblages: the influence of temperature with altitude and latitude. *Freshwater Biology* 38: 247-261.

Johnston, W. B. 1969. *Modification of the natural environment by man.* In: G. A. Knox, The natural history of Canterbury. A.H. and A.W. Reed, Wellington, New Zealand.

Karr, J. R., and E. W. Chu. 1999. *Restoring life in running waters: better biological monitoring.* Island Press, Washington D.C.

Kutka, F. J., and C. Richards. 1996. Relating diatom assemblage structure to stream habitat quality. *Journal of the North American Benthological Society* 15: 469-480.

—. 1997. Short-term nutrient influences on algal assemblages in three rivers of the Minnesota river basin. *Journal of Freshwater Ecology* 12: 411-419.

Lenat, D. R., D. L. Penrose, and K. W. Eagleson. 1981. Variable effects of sediment addition on stream benthos. *Hydrobiologia* 79: 187-194.

MacGibbon, R. 2000. Managing waterways on farms: a guide to sustainable water and riparian management in rural New Zealand. Ministry for the Environment. MfE No. 353.

Maddock, I. 1999. The importance of physical habitat assessment for evaluating river health. *Freshwater Biology* 41: 373-391.

Maxted, J. R., M. T. Barbour, J. Gerritsen, V. Poretti, N. Primrose, A. Silvia, D. L. Penrose, and R. Renfrow. 2000. Assessment framework for mid-Atlantic coastal plain streams using benthic macroinvertebrates. *Journal of the North American Benthological Society* 19: 128-144.

McCune, B., and M. J. Mefford. 1999. *PC-ORD. Multivariate analyses of ecological data, Version 4.* MjM Software Design, Glenenden Beach, Oregon, USA.

McDowall, R. M. 1998. Once were wetlands. Fish and Game. New Zealand Fishing News, Auckland, New Zealand.

Miller, R. J., and R. M. Berry. 1999. From water quality to ecosystems: meeting the requirements of the Resouce Management Act. In: The use of macroinvertebrates in water management. Ministry for the Environment.

Moore, S. C. 1997. A photographic guide to the freshwater inverterbrates of New Zealand. Otago Regional Council, Dunedin.

—. 1999. Freshwater biomonitoring programme. 1996-1999 review. Otago Regional Council. Report No. 99-447.

Norris, R. H., and A. Georges. 1993. *Analysis and interpretation of benthic macroinvertebrate surveys.* In: D. M. Rosenberg and V. H. Resh, Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, London.

Pan, Y., R. J. Stevenson, B. H. Hill, A. T. Herlihy, and G. B. Collins. 1996. Using diatoms as indicators of ecological conditions in lotic systems: a regional assessment. *Journal of the North American Benthological Society* 15: 481-495.

Pielou, E. C. 1969. *An introduction to mathematical ecology.* Wiley, New York.

Plafkin, J. L., M. T. Barbour, K. D. Porter, S. K. Gross, and R. M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. U.S. Environmental Protection Agency, Office of Water Regulations and Standards.

Quinn, J. M., and A. B. Cooper. 1997. Land-water interactions at Whatawhata, New Zealand: introduction and synthesis. *New Zealand Journal of Marine and Freshwater Research* 31: 569-579.

Quinn, J. M., A. B. Cooper, R. J. Davies-Colley, and R. B. Williamson. 1997. Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research* 31: 579-597.

Quinn, J. M., and C. W. Hickey. 1990a. Characterisation and classification of benthic invertebrate communities in 88 New Zealand rivers in relation to environmental factors. *New Zealand Journal of Marine and Freshwater Research* 24: 387-410.

—. 1990b. Magnitude of effects of substrate particle size, recent flooding, and catchment development on benthic invertebrates in 88 New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research* 24: 411-428.

Quinn, J. M., G. L. Steele, C. W. Hickey, and M. L. Vickers. 1994. Upper thermal tolerances of twelve New Zealand stream invertebrate species. *New Zealand Journal of Marine and Freshwater Research* 28: 391-397.

Quinn, J. M., R. B. Williamson, R. K. Smith, and M. L. Vickers. 1992. Effects of riparian grazing and channelisation on streams in Southland, New Zealand. 2. Benthic invertebrates. *New Zealand Journal of Marine and Freshwater Research* 26: 259-273.

Resh, V. H., R. H. Norris, and M. T. Barbour. 1995a. Design and implementation of rapid assessment approaches for water resource monitoring using benthic macroinvertebrates. *Australian Journal of Ecology* 20: 108-121.

Reynoldson, T. B., R. C. Bailey, K. E. Day, and R. H. Norris. 1995. Biological guidelines for freshwater sediment based on BEnthic Assessment of SedimenT (the BEAST) using a multivariate approach for predicting biological state. *Australian Journal of Ecology* 20: 198-219.

Reynoldson, T. B., R. H. Norris, V. H. Resh, K. E. Day, and D. M. Rosenberg. 1997. The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. *Journal of the North American Benthological Society* 16: 833-852.

Richards, C., R. J. Haro, L. B. Johnson, and G. E. Host. 1997. Catchment and reach-scale properties as indicators of macroinvertebrate species traits. *Freshwater Biology* 37: 219-230.

Rounick, J. S., and M. J. Winterbourn. 1982. Benthic faunas of forested streams and suggestions for their management. *New Zealand Journal of Ecology* 5: 140-150.

Ryan, P. A. 1991. Environmental effects of sedimentation on New Zealand streams: a review. *New Zealand Journal of Marine and Freshwater Research* 25: 207-221.

Ryder, G. I. 1998. Southland Regional Council: Regional freshwater biological monitoring survey 1998. Ryder Consulting Ltd., Dunedin.

Scarsbrook, M. R., I. K. G. Boothroyd, and J. M. Quinn. 2000. New Zealand's National Water Quality Network: long-term trends in macroinvertebrate communities. *New Zealand Journal of Marine and Freshwater Research* 34: 289-302.

Scott, D., J. W. White, and D. S. Rhodes. 1994. Invertebrate fauna of three streams in relation to land use in Southland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 28: 277-290.

Shannon, C. E., and W. Weaver. 1949. *The mathematical theory of communication.* University of Illinois Press, Urbana.

Snelder, T., B. J. F. Biggs, U. Shankar, R. M. McDowall, T. Stephens, and I. Boothroyd. 1998. Development of a system of physically based habitat classification for water resources management in New Zealand rivers. NIWA Christchurch Consultancy Report CHC98/68.

Stansfield, B. 1999. Wellington Regional Council annual freshwater quaility report 1998/99.

Stark, J. D. 1985. A macroinvertebrate community index of water quailty for stony streams. *Water and Soil Miscellaneous Publication 87.* National Water and Soil Conservation Authority, Wellington.

—. 1993. Performance of the Macroinvertebrate Community Index: effects of sampling method, sample replication, water depth, current velocity, and substratum on index values. *New Zealand Journal of Marine and Freshwater Research* 27: 463-478.

—. 1998. SQMCI: a biotic index for freshwater macroinvertebrate coded-abundance data. *New Zealand Journal of Marine and Freshwater Research* 32: 55-66.

—. 1999. Biotic indices: Their role in water management in New Zealand. In: The use of macroinvertebrates in water management. Ministry for the Environment.

Stone, M. K., and J. B. Wallace. 1998. Long-term recovery of a mountain stream from clear-cut logging: the effects of forest succession on benthic invertebrate community structure. *Freshwater Biology* 39: 151-169.

Suckling, D. M. 1982. Organic wastewater effects on benthic invertebrates in the Manawatu River. *New Zealand Journal of Marine and Freshwater Research* 16: 263-270.

Suren, A. M. 1994. Macroinvertebrate communities of streams in western Nepal: effects of altitude and land use. *Freshwater Biology* 32: 323-336.

—. 1999. River ecoclassification protocol. Part I, Guidance notes: Part II, Survey Protocol. NIWA Client Report No. MFE00503.

Suren, A. M., T. Snelder, B. Biggs, M. Weatherhead, and D. Baird. 2000. Testing the value of a hierarchical river environment classification to constrain variance in macroinvertebrate communities. NIWA Client Report: CHC00/58.

Tate, M., and J. S. Heiny. 1995. The ordination of benthic invertebrate communities in the South Platte River basin in relation to environmental factors. *Freshwater Biology* 33: 439-454.

Thompson, R. M., and C. R. Townsend. 2000. *New Zealand's stream invertebrate communities: an international perspective.* In: K. J. Collier and M. J. Winterbourn, New Zealand stream invertebrates: ecology and implications for management. New Zealand Limnological Society, Christchurch, New Zealand.

Towns, D. R. 1981. Effects of artificial shading on periphyton and invertebrates in a New Zealand stream. *New Zealand Journal of Marine and Freshwater Research* 15: 185-192.

Townsend, C. R., C. J. Arbuckle, T. A. Crowl, and M. R. Scarsbrook. 1997. The relationship between land use and physicochemistry, food resources and macroinvertebrate communities in tributaries of the Taieri River, New Zealand: a hierarchically scaled approach. *Freshwater Biology* 37: 177-191.

TRC. 1999. Some statistics from the Taranaki Regional Council database (FWB) of freshwater macroinvertebrate surveys performed during the period from January 1980 to 31 December 1998. Taranaki Regional Council. TRC Technical Report 99-17.

Vant, B., A. Taylor, and B. Wilson. 2000. Land use and river condition in the Waikato region. Proceedings of Water 2000 Conference, March 2000, New Zealand Water and Wastes Association, Auckland.

Vinson, M. R., and C. P. Hawkins. 1996. Effects of sampling area and subsampling procedures on comparison of taxa richness among streams. *Journal of the North American Benthological Society* 15: 392-399.

Vucetich, C. G. 1969. *Soils of Canterbury.* In: G. A. Knox, The natural history of Canterbury. A.H. and A.W. Reed, Wellington, New Zealand.

Waters, T. F. 1995. Sediment in streams: source, biological effects, and control. *American Fisheries Society Monograph 7. American Fisheries Society, Bethesda, Maryland.*

Williams, D. D., and H. B. N. Hynes. 1976. The recolonization mechanisms of stream benthos. *Oikos* 27: 265-272.

Winterbourn, M. J. 1974. The life histories, trophic relations and production of *Stenoperla prasina* (Plecoptera) and *Deleatidium* sp. (Ephemeroptera) in a New Zealand river. *Freshwater Biology* 4: 507-524.

—. 1986. Effects of land development on benthic stream communities. *New Zealand Agricultural Science* 20: 115-118.

—. 1999. Recommendations of the New Zealand macroinvertebrate working group: Monitoring for sustainable river ecosystem management and the role of macroinvertebrates. In: The use of macroinvertebrates in water management. Ministry for the Environment.

—. 2000. *Feeding ecology.* In: K. J. Collier and M. J. Winterbourn, New Zealand stream invertebrates: ecology, and implications for management. New Zealand Limnological Society, Christchurch, New Zealand.

Winterbourn, M. J., and K. J. Collier. 1987. Distribution of benthic invertebrates in acid, brown water streams in the South Island of New Zealand. *Hydrobiologia* 153: 277-286.

Winterbourn, M. J., and K. L. D. Gregson. 1989. Guide to the aquatic insects of New Zealand. *Bulletin of the Entomological Society of New Zealand* 9, 80p.

Wohl, D. L., J. B. Wallace, and J. L. Meyer. 1995. Benthic macroinvertebrate community structure, function and production with respect to habitat type, reach and drainage in the southern Appalachians (U.S.A.). *Freshwater Biology* 34: 447-464.

APPENDIX ONE

QUALITY ASSURANCE FOR SORTING INVERTEBRATE SAMPLES: ASSESSING BETWEEN-OPERATOR VARIABILITY AND VALIDATING THE MODIFIED 100 FIXED COUNT METHOD FOR SORTING INVERTEBRATE SAMPLES

Modified from Suren et al. (2000).

INTRODUCTION

A key part of Environment Canterbury's and NIWA's 1999-2000 invertebrate survey programme was to develop an efficient sample processing protocol using a methodology that most accurately described the invertebrate communities at each site, yet was time efficient, and minimised between-operator variability. These three features were particularly important since the study involved sampling a large number of sites (230) within a short time-frame (c. 2 months). Further time constraints meant that the data needed to be available for analysis within about 2 months of collection. The six students employed by Environment Canterbury to collect and process all invertebrate samples were trained to a consistent standard by NIWA staff prior to the field collection phase of the study. Following this, each student processed the collected samples individually using Bogorov sorting trays as outlined by Winterbourn & Gregson (1989) and explained in more detail by Suren (1999) in the Protocol manual.

As part of standard Quality Assurance/Quality Control (QA/QC) checks employed for this study, concern was raised at two issues of the invertebrate processing method. The first major issue to address was the need to establish whether a fixed count of 100 invertebrates would be adequate to describe the communities in each stream. Although there are many differences in how invertebrate samples are processed, obtaining information on species richness and relative abundance of different taxa was a main objective. Values obtained for these metrics are related to sample size and sampling effort, such that an increase in sample size will result in a corresponding likelihood that rare taxa will be found. Rare taxa can have a significant effect on indices calculated, subsequently (Stark 1993). Two main methods are used to standardise invertebrate sample processing: fixed count methods, and sub-sampling.

Some of the inherent problems associated with different sample processing protocols are discussed by Courtemanch (1996), Barbour and Gerritsen (1996) and Vinson and Hawkins (1996). Fixed-count processing of samples (i.e., counting only a certain number of individuals) was criticised by Courtemanch (1996) as "lacking ecologically interpretive value" since the actual proportion of the community sampled is not known. Moreover, richness is affected by the number of organisms counted, and by the proportion of the community that is sampled. Barbour and Gerristen (1996), however, defend the use of fixed-count processing, arguing that it provides more consistent information than area-based sampling, in which the number of organisms sorted is variable. A compromise condition was put forward by Vinson and Hawkins (1996) who recommended a 2-phase sorting procedure that relies on both a fixed-count and the removal of rare species. In practice, the final choice of method is often dictated by a mixture of the precision required, cost and personal preference.

If a 2-phase, fixed count method is to be used, consideration needs to be made as to how many individuals should be counted. If too few individuals are counted, then the invertebrate communities may be inadequately characterised. Conversely, if too many individuals are counted, then the extra information that is gleaned adds little in the way of added accuracy or precision. Moreover, the number of invertebrates that need to be counted often depends upon species evenness. Samples where species evenness is high should require fewer animals to be counted to adequately characterise the community, since each taxon is equally likely to be found. Samples in which evenness is low require more individuals to be counted, since rare taxa are less likely to be encountered.

The first aim of the QA/QC study was to compare the effectiveness of the 2-phase fixed counting technique to determine whether different results would be obtained by counting the first 100, 200 and 300 individuals. A range of metrics that are commonly used for environmental assessment work were then calculated for the different data sets, and the results compared.

The second issue of concern was to establish the degree and magnitude of between-operator variability that may arise by having the samples processed by 6 different people. In this instance, invertebrate counts by different operators were compared to ensure that a consistent and accurate characterisation of the invertebrate communities was obtained.

METHODS

Field and laboratory methods

Invertebrate collection and sorting methods are described in Chapter 2. In summary, all samples were collected by kick sampling from 3 separate runs by moving methodically across each run. The substrate was disturbed to a depth of up to 10 cm within a 50 cm wide strip upstream of the net. All organic matter, invertebrates and fine inorganic sediments were washed into the net (0.5mm). Samples from each run were pooled, and all organic matter elutriated from heavier stones in the field. All samples were preserved with alcohol.

To quantify the effects of different fixed counts on the results, samples from 10 rivers were chosen and processed as follows. Organic material was separated from inorganic material by a final elutriation, and the remaining organic matter was subsampled as necessary to end up with a manageable quantity of material. This material was divided into three parts, each of which was placed in a Bogorov tray (Winterbourn and Gregson 1989). The first 100 invertebrates were identified and counted under various magnifications. After the first 100 animals were counted, the remaining contents of the tray were scanned for previously unrecorded taxa. Following this, the second and third subsamples were processed in a similar manner. In this way, data was collected that represented a 2-phase count of 100, 200 and 300 individuals (plus rare taxa from each count).

To quantify between-operator variability, samples from a new set of five rivers were chosen at random from the complete set of samples. They were processed as above, except that only the first 100 individuals in a sub-sample were counted. After scanning the rest of the subsample for rare taxa, all material was recombined. Another operator then processed this recombined sample. In this way, variability inherent in subsampling, counting, and identification was assessed.

Statistical analyses

Data from the 100, 200 and 300 fixed counts from each tray were analysed to give species lists and abundances of all invertebrates encountered. There data (relative abundance) were ordinated using DECORANA to determine whether ordinal scores differed between samples obtained with different fixed count methods. Multi-response permutation procedures (MRPP) were then used to test difference among invertebrate communities obtained by the three *a priori* defined fixed count methods. Following this, species evenness (Pielou's equitability calculation (Pielou 1969); diversity (*H'*) / ln(richness)) and diversity (Shannon-Weiner's *H*; (Shannon and Weaver 1949)), taxonomic richness, and the MCI and QMCI (Stark 1985) were calculated for each of the fixed counts. ANOVA was used to examine whether these commonly used metrics differed between samples sorted using different counts. Similar analyses were done on the data obtained by the different operators.

RESULTS

Validation of fixed counts

Forty nine taxa were collected from the 10 sites. The most common taxa were the leptophlebiid mayfly *Deleatidium*, followed by the snail *Potamopyrgus*, larval elmid beetles, and the caddisflies *Pycnocentrodes* and *Olinga*. No taxa were recorded from all 3 replicates from the 10 sites (i.e. in 30 samples), but *Pycnocentrodes*, *Deleatidium* and elmids were found in over 75% of them.

A significant difference was found between DECORANA scores of the 10 streams (F = 1366.5, P < 0.001; Fig. 1), highlighting the fact that they supported very different invertebrate communities. However, no significant difference in observed DCA score was found with respect to the numbers of invertebrates counted (F = 2.27, P = 0.148; Fig. 2). MRPP indicated no significant difference between invertebrate communities when 100, 200 or 300 counts were made (R = -0.064, P > 0.05).

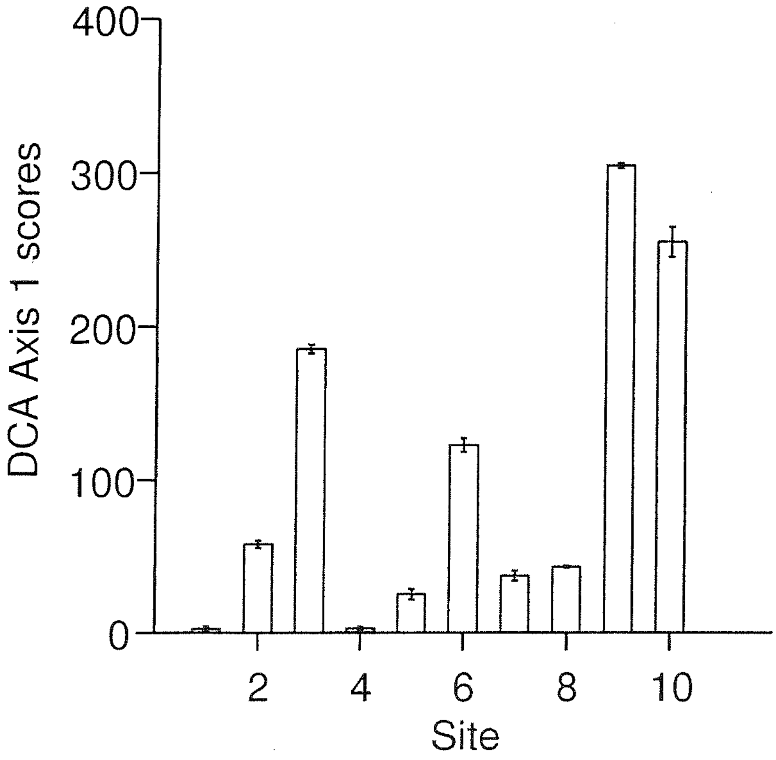


Fig. 1 Mean (±1SE, n=3) DCA scores for each of the 10 sites, derived from counts of 100, 200, and 300 invertebrates. Note how the error bars (±1SE: the variability associated with ordination scores derived from the different counting methods) are all very small.

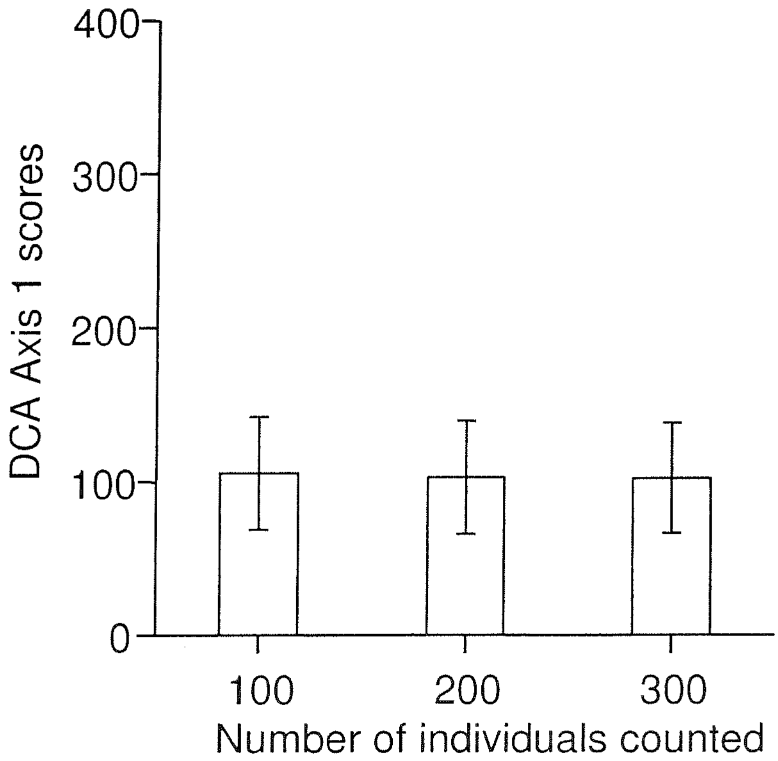


Fig. 2. Mean (±1SE) DCA axis 1 scores for the 10 rivers based on invertebrate communities assessed by the 100, 200 and 300 fixed count methods.

Similarly, no significant difference was found in either species evenness or diversity between counts of 100, 200 and 300 individuals for the 10 streams (F = 0.203 and 0.142, respectively, $P > 0.05$; Fig. 3). In addition, no significant difference was found between MCI scores or QMCI scores for each of the 10 sites when sorted by the different counts (F = 0.114 and 0.002 respectively, $P > 0.05$; Fig. 4). This was despite substantial differences in MCI and QMCI between the 10 sites (F = 9.80 and 14.98 respectively, $P < 0.005$; Fig 2). Mean taxonomic richness increased slightly as more invertebrates were counted (Fig. 5), but the differences were not significant (F = 2.762, P = 0.081).

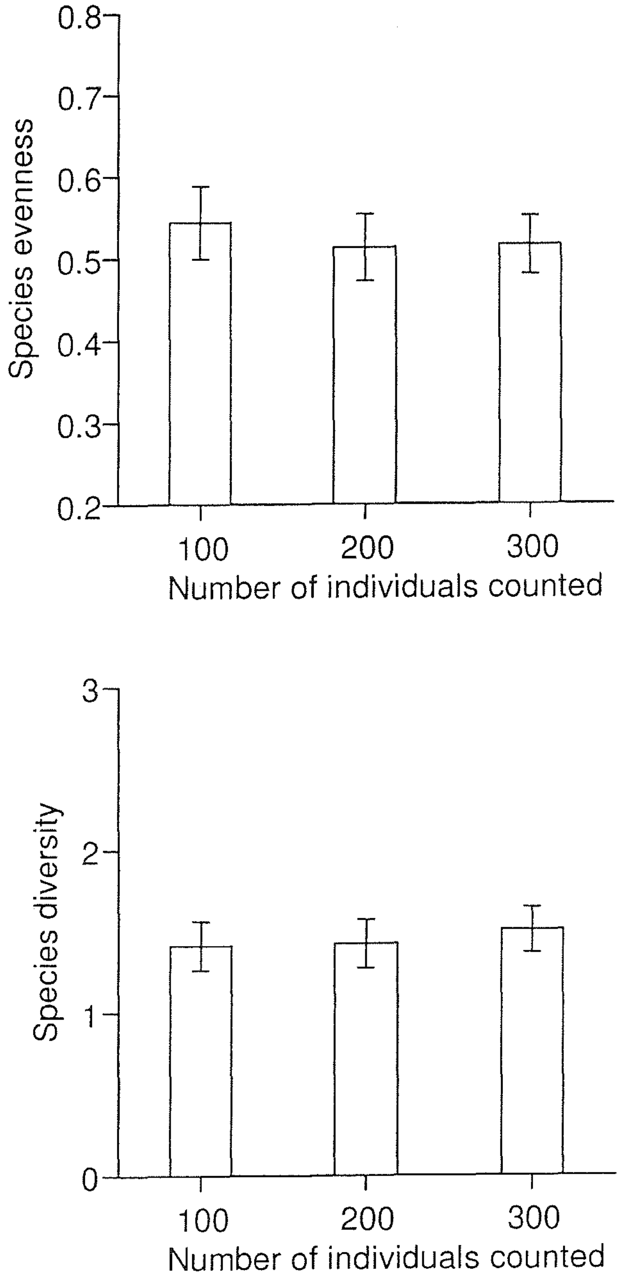


Fig. 3. Species evenness and species diversity of the 10 rivers based on invertebrate communities assessed by the 100, 200 and 300 fixed count methods (mean $\pm$ 1SE).

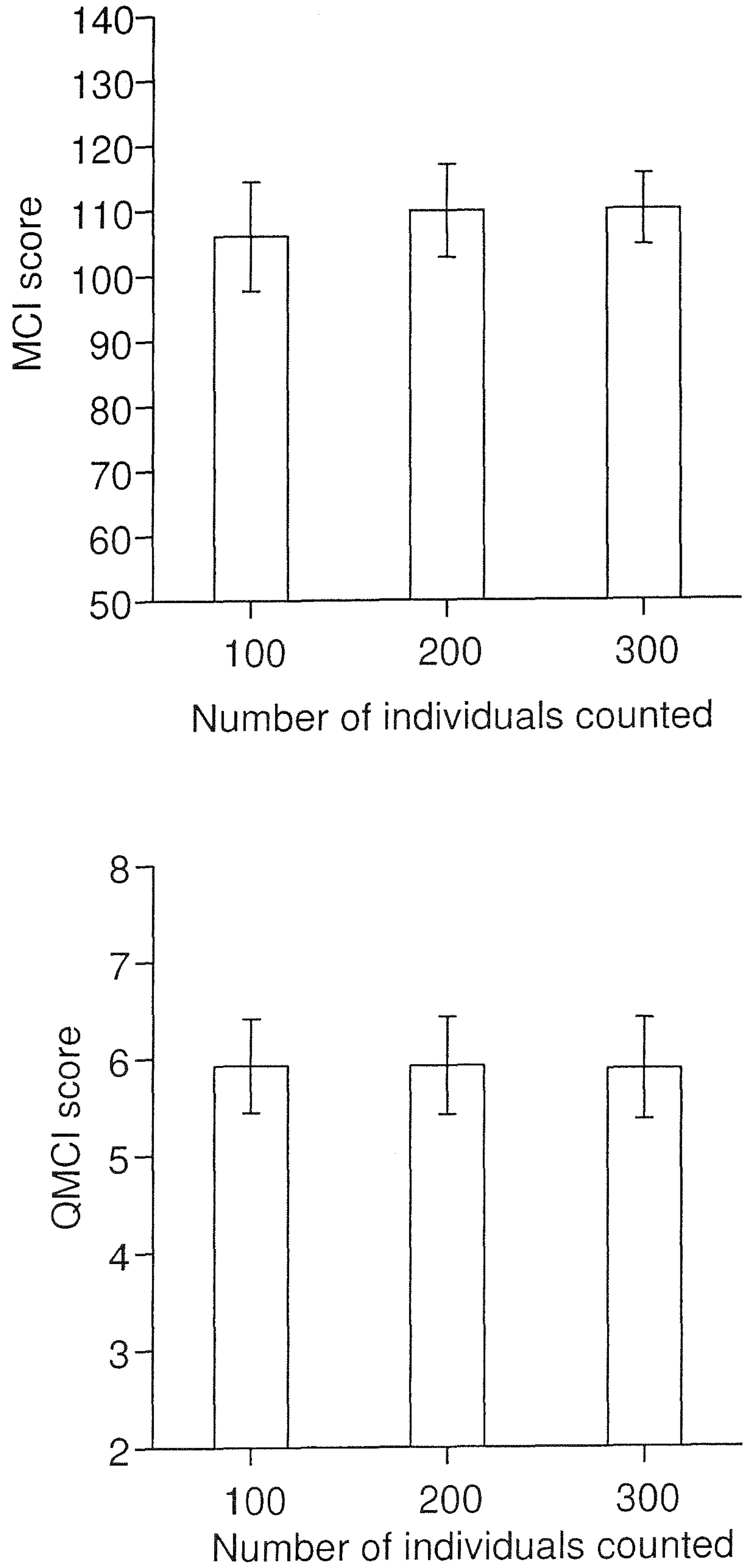


Fig. 4. Mean (±1SE) MCI and QMCI scores for the 10 rivers based on invertebrate communities as assessed by the 100, 200 and 300 fixed count methods.

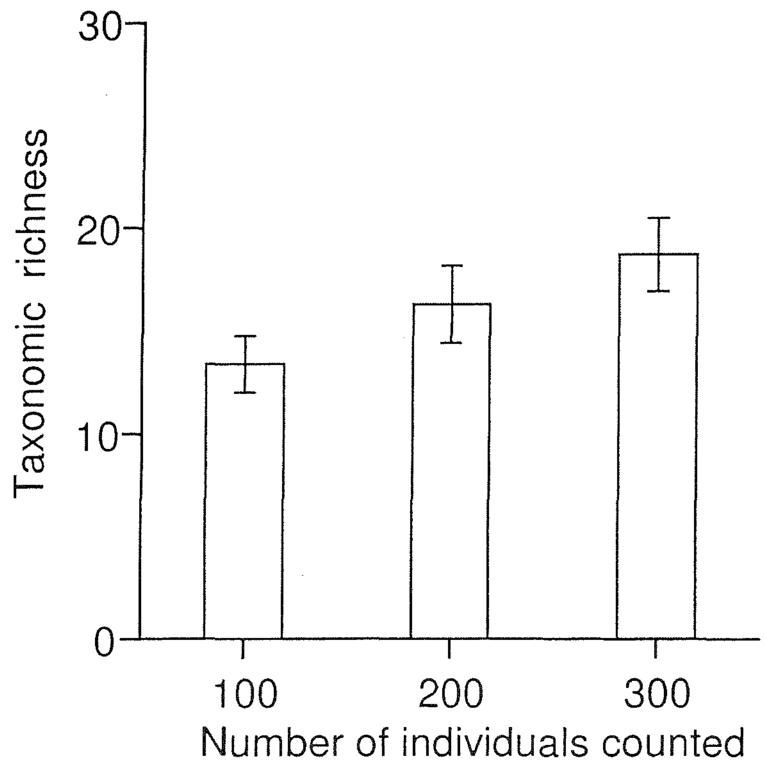


Fig. 5 Mean (±1SE) taxonomic richness as a function of the numbers of invertebrates counted in Bogorov trays from each of the 10 sites.

Assessment of between-sorter variability

Fifty four taxa were collected from the 5 sites and identified by the 5 operators. The most common taxa were Chironominae, *Pycnocentrodes*, Oligochaeta, *Deleatidium* and *Potamopyrgus*. Four taxa (*Potamopyrgus*, *Austrosimulium*, Orthocladiinae and Oligochaeta) were found in all samples by each of the 5 sorters.

The DCA axis 1 scores of the 5 streams differed significantly (F = 990.8, P < 0.001), indicating that they supported different invertebrate communities (Fig. 6). However, no significant difference in DCA axis 1 scores for the five sites was obtained by the different sorters (F = 0.0036, P > 0.05). This indicates that identification, sorting and counting by each sorter was very similar.

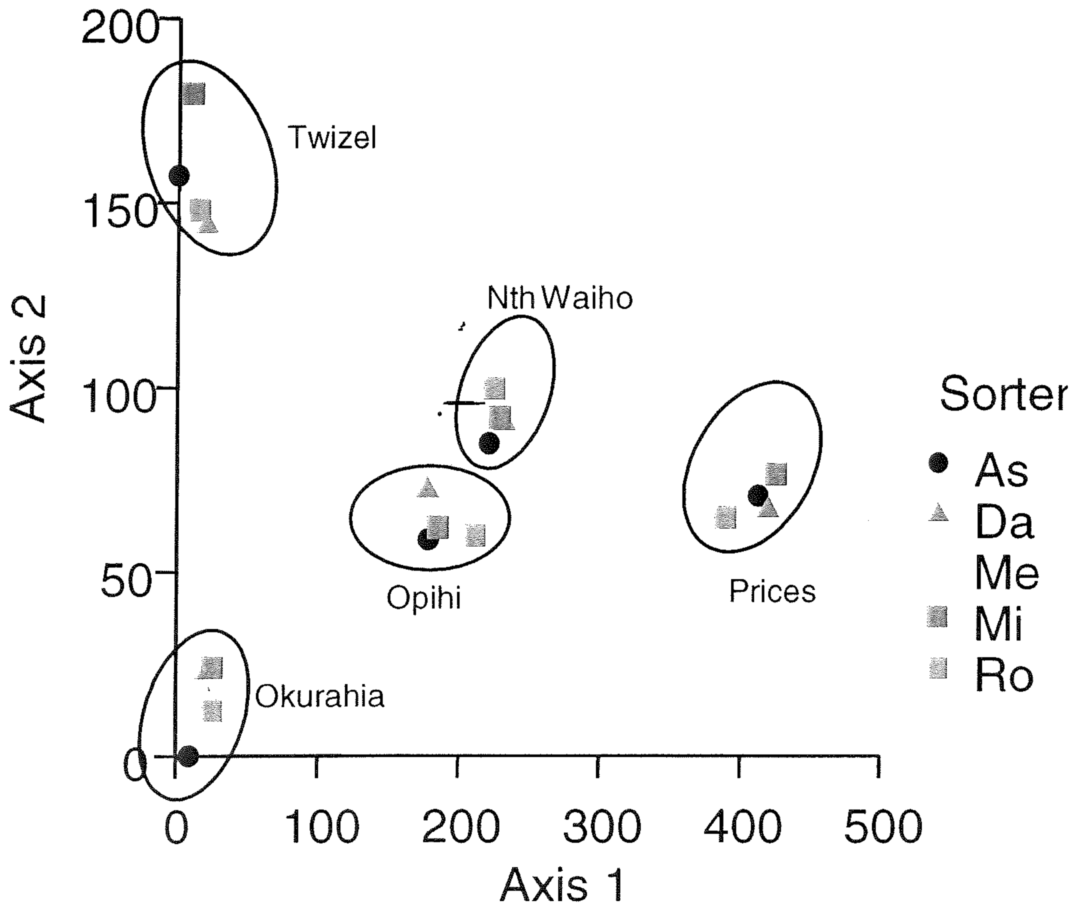
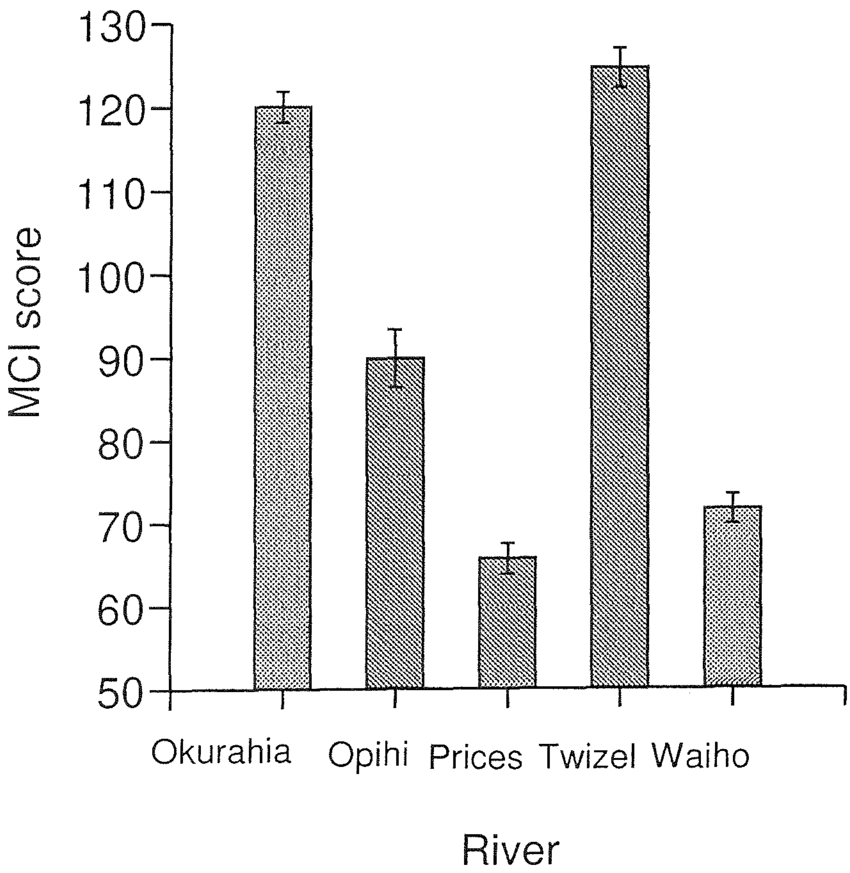


Fig. 6 Ordination (DCA) of the 5 stream communities based on the data obtained by 5 sorters.

MRPP analysis also indicated no significant difference between the invertebrate communities as assessed by the different operators (R = -0.121, P > 0.05), but highly significant differences among the five communities (R = 0.616, P < 0.001). In this case there was no problem, indicating that training and working practice was effective.

Furthermore, no significant differences in species evenness, species diversity or taxonomic richness of the 5 river communities were obtained by the different sample sorters (P<0.05). MCI and QMCI scores calculated for each stream for the five operators were also very similar (Fig. 7, F = 0.019 and 0.036 for MCI and QMCI respectively, P > 0.05).

A.


130
120
110
100
90
80
70
60
50
MCI score
Okurahia
Opihi
Prices
Twizel
Waiho
River

B.

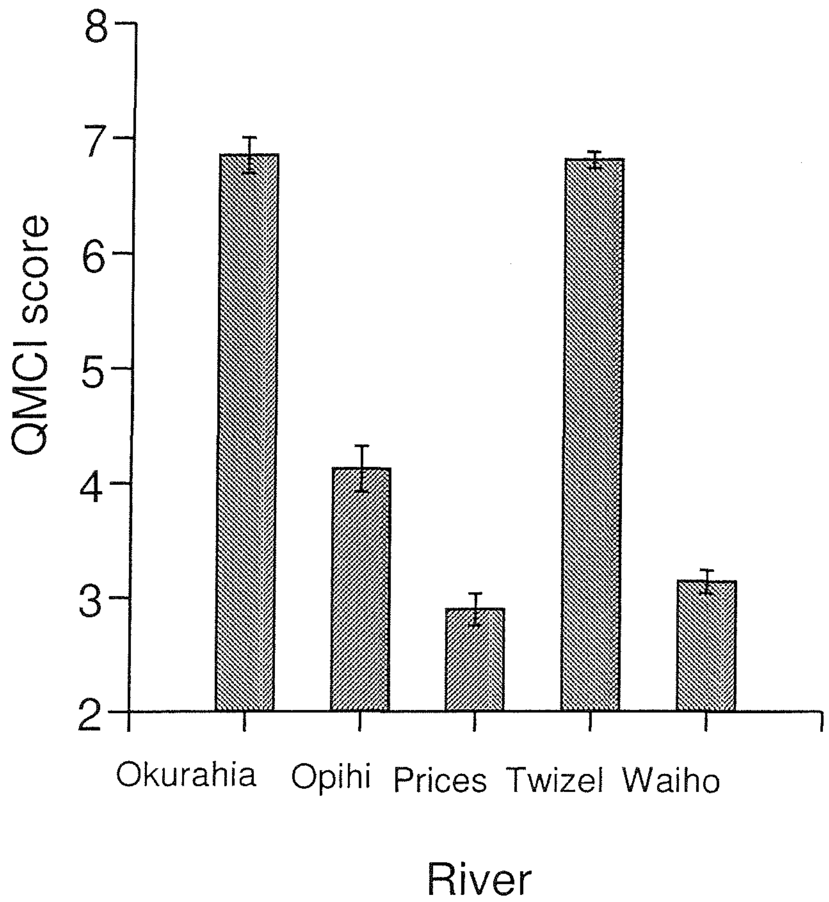


Fig. 7 Mean MCI (A) and QMCI (B) scores (± 1SE) for the 5 sites obtained by the 5 sample sorters. Note that the standard error bars are all very small.

DISCUSSION

The aims of this investigation were to assess whether similar results could be obtained from fixed counts of 100, 200 and 300 invertebrates, and to quantify between-operator variability in the processing of invertebrate samples.

Given the inherent time constraints associated with processing the samples, and the obvious trade-off between accuracy and time, it was apparent at the outset that some sort of fixed counting method would be required. The question was whether a fixed count of 100 invertebrates would be sufficient.

The results showed little difference in the commonly used metrics used to describe stream invertebrate communities when they were calculated from counts of 100, 200 or 300 individuals. The greatest difference that may become apparent is the increased species richness, which increased with sample size, although the difference was small (means of 14-19 per sample).

Overall, the results indicate that sorting a fixed count of 100 individuals, plus scanning the rest of the sub-sample for rare taxa produced consistent results that described the invertebrate communities, effectively. The results also showed that between-operator variability was low although small differences were found in the number of taxa identified by different operators. These differences had little effect on commonly used metrics such as species evenness, diversity, MCI and QMCI.

APPENDIX TWO
FIELD DATA COLLECTION SHEETS

SITE DETAILS

River Name ______________________________

Location ______________________________

*Grid reference (NZMS 260 series) Map No:*______________*Grid ref:* ______________

Date ______________________*Time* ____________

Observers ______________________

Institution ______________________

Photo Nos. ______________________*(attach to the data sheets)*

Weather ______________________________

1.0 BIOLOGICAL ASSESSMENT

1.1 Invertebrates

Collect a semi-quantitative kick-sample from three runs at each location: these are to be pooled. Remember to only collect samples from undisturbed areas of the stream; therefore always walk upstream to new sample locations and preferably do the kick sampling before any other work in the stream. All samples should be preserved in the field and kept separate for laboratory sorting. All invertebrates collected should be counted, and taxa identified to genera where possible, as per Stark (1993).

1.2 Periphyton

In each of the three runs, randomly select 5 stones, sediment or water plant samples and record the estimated % coverage of the exposed part of the stone by each type of periphyton.

Periphyton (on exposed surface)		Run 1					Run 2					Run 3				
		1	2	3	4	5	1	2	3	4	5	1	2	3	4	5
Thin mat/film:	green															
(under 0.5 mm thick)	light brown															
	Black/dark brown															
Medium mat:	green															
(0.5-3 mm thick)	light brown															
	Black/dark brown															
Thick mat:	green/light brown															
(over 3 mm thick)	black/dark brown															
Filaments, short	green															
(under 2 cm long)	brown/reddish															
Filaments, long	green															
(over 2 cm long)	brown/reddish															

2.0 Regional Features

2.1 Altitude at site ____________________________(m a.s.l)

2.2 Dominant catchment landuse

Land-use category	Tick dominant
Indigenous forest	
Exotic forest	
Tussock (not / lightly grazed)	
Tussock (modified and grazed)	
Pasture – sheep farming	
Pasture – dairy farming	
Pasture – deer farming	
Horticulture – crops	
Scrub land	

2.3 Catchment topography

1. Steep catchment > 20^{o}
2. Hilly catchment $10 - 20^{o}$
3. Rolling catchment $5 - 10^{o}$
4. Flat catchment < 5^{o}

3.0 Hydrology

3.1 Stream source

1. Spring-fed
2. Lake-fed
3. Mountain-fed (> 1000 m a.s.l.)
4. Foothills-fed (500-1000 m a.s.l.)
5. Rolling hill country (< 500 m a.s.l.)
6. Flat lowlands (wetland-fed)

3.2 Stream order at site ____________

4.0 Riparian Zone

4.1 Canopy cover over the streambed

	LB	RB	
Open	1	1:	0%
	2	2:	<25%
	3	3:	26-50%
	4	4:	51-75%
Closed	5	5:	>75%

4.2 Bank vegetation cover

LB	RB	
1	1	Over 80% of the streambank surfaces covered by vegetation.
2	2	50-79% of the streambank surfaces covered by vegetation.
3	3	25-49% of the streambank surfaces covered by vegetation.
4	4	Less than 25% of the streambank surfaces covered by vegetation.

4.3 Bank vegetation

For each bank along the 10 metre length of the site estimate the percentage (to the nearest 10%) covered by the listed vegetation types in a strip 5 metres wide parallel to the water's edge.
Note: the true left and true right are the left and right sides looking downstream.

	Native trees	Wet-land vegeta-tion	Tall tussock grass-land, not improved	Intro-duced trees (willow, poplar...)	Other intro-duced trees (conifers)	Scrub	Rock, gravels	Short tussock grass-land, improved	Pasture grasses and weeds	Bare ground, roads, build-ings	
%, true left:											100
%, true right											100
total % (L + R)											200

5.0 Bank Attributes/Channel Attributes

5.1 Bank stability

1. Bank stable. Little (< 5%) or no evidence of erosion or bank failure.
2. Moderately stable. Infrequent areas (5 – 30%) of erosion, often healed over.
3. Moderately unstable. Moderate frequency (30 – 50%) and size of eroded areas (without vegetation).
4. Unstable. Many eroded areas. "Raw" areas frequent (> 50%) along straight sections and banks.

5.2 Bank heterogeneity

1. Natural stream meander pattern, irregular sided banks, high bank heterogeneity from changes in shape, substrate or vegetation: natural banks.
2. Natural stream meander pattern, irregular sided banks, low heterogeneity from bank substrate or vegetation, banks natural to semi-natural.
3. Channelled stream, meander pattern greatly altered, few bends or sinuosity. High heterogeneity from bank substrate or vegetation.
4. Channelled stream, meander pattern greatly altered, few bends or sinuosity. Low heterogeneity from bank substrate or vegetation. Vertical or uniformly sloped banks, often reinforced.

5.3 Bank and channel roughness elements (trees, logs, rocks etc.)

1. Roughness elements large, stable, and common on banks and in channel.
2. Roughness elements common in channel but rare on banks.
3. Roughness elements on banks or in channel.
4. No roughness elements, channel smooth.

5.4 Channel heterogeneity

1. Great diversity of channel widths and depths forming a series of riffles, runs and pools; large variation in velocity throughout stream.
2. Little diversity in channel width, high diversity in stream depth, velocity still variable throughout stream.
3. Little diversity in channel width and depth, velocity within channel only slightly variable.
4. No change in channel width and depth, constant velocity throughout channel (or still water).

6.0. Sediments/Substrate

6.1 Macrophytes

1. Clumps of submerged macrophytes not covering the majority of the streambed (i.e., < 50%), and not greatly obstructing the flow patterns in the stream, OR an absence of plants in stony streams.
2. Clumps of emergent macrophytes, not choking the streambed or causing stagnation, OR submerged macrophytes covering the majority of the streambed (i.e., > 50%) but not obstructing the water flow.
3. Submerged macrophytes covering the majority of the streambed, and obstructing the flow in the channel, reducing water velocities in places.
4. Emergent macrophytes completely or largely choking the channel, causing areas of stagnation and excessive silt entrapment.

6.2 Bottom substrate available cover

1. Greater than 50% cover from bed material, or from submerged logs, undercut banks, macrophytes etc.
2. 30-50% cover from bed material or other stable habitat.
3. 10-30% cover from bed material or other stable habitat.
4. Less than 10% instream cover from bed material or other habitat. Lack of instream cover is obvious.

6.3 Siltation/embeddedness

1. Gravel, cobble and boulder particles are between 0 and 25% surrounded by fine sediment (i.e. clay, silt or sand).
2. Gravel, cobble and boulder particles are between 25 and 50% surrounded by fine sediment.
3. Gravel, cobble and boulder particles are between 50 and 75% surrounded by fine sediment.
4. Gravel, cobble and boulder particles are > 75% surrounded by fine sediment.

6.4 Packing of substrate material

1. Assorted sizes tightly packed.
2. Moderately packed.
3. Mostly a loose arrangement.
4. No packing evident; loose easily moved assortment.

6.5 Percentage of stable substrate

1. Stable materials 80-100%; bedrock or other immobile substrate often evident.
2. Stable materials 50-80%; bedrock or other immobile substrate usually evident.
3. Stable materials 20-50%; bedrock or other immobile areas rarely exposed in stream.
4. Stable materials 0-20%; bedrock or immobile areas not present.

6.6 Substrate heterogeneity

1. Heterogeneous substrate, many interstitial spaces, not filled with silt. Bed surface often rough, usually >5 size classes; no class >50% cover.
2. Slightly heterogeneous, some interstitial spaces slightly filled with silt, bed surface with only a few large rough elements. 3-4 size classes but no size class is >50%.
3. Mostly homogeneous, interstitial spaces rare; covered with silt, bed surface only slightly rough/undulating, no large rough elements, 2-3 size classes, but usually dominated by >50% of 1 class.
4. Homogeneous, interstitial spaces absent, bed surface uniform, little change in sizes evident, only 1-2 classes.

6.7 Periphyton cover

1. Cobbles and boulders without visible surface films, surfaces rough to the touch.
2. Cobbles and boulders covered with a thin film of tightly adhering algae, filamentous algae and algal mats absent or rare.
3. Cobbles and boulders covered with a thick film of algae, > 50% of streambed, filamentous algae and algal mats absent or rare.
4. Cobbles and boulders covered with a thick film of algae, > 50% of streambed, filamentous algae and algal mats common but < 50% of streambed covered.
5. Streambed cover dominated by filamentous algae or thick algal mats; cover of streambed > 50%.

6.8 Composition of the streambed

Estimate by eye the percentages (to the nearest 10%) of cover of different types of material making up the streambed.

Bedrock	Boulder	Large Cobbles	Small Cobbles	Gravel	Sand	Mud / silt	Man-made (concrete)	Woody debris	Water plants (rooted)
Size (cm)	(>25)	(12-25)	(6 – 12)	(0.2 – 6)					

7.0 Water Quality

7.1 Conductivity __________ μScm^{-1}

7.2 Water clarity (clarity tube reading) __________ cm

7.3 pH __________

7.4 Spot water temperature _________ ^{o}C

7.5 Flow velocity (three measurements of surface velocity in each run, taken by measuring the length of time for an object (such as an orange) to float a known distance)

Run 1

____

____

____

Run 2

____

____

____

Run 3

____

____

____

APPENDIX THREE
LIST OF SITE DETAILS AND SOME PHYSICAL AND WATER QUALITY MEASURES

Site No.	Name	Map #	Easting	Northing	Altitude (m a.s.l)	Temp[1]	Width[1]	Depth[1]	Conductivity[1] μS cm^{-1}
1	Cust River	M35	575	670	140	13.9	3.65	0.33	156.1
2	Eyre River	L35	420	670	250	13.25	7.75	0.34	97.3
3	Glentui River	M34	523	755	250	11.65	5.45	0.27	170.2
4	Maori Stream	M34	525	789	380	10.25	2.07	0.17	113.1
5	Tommy's Creek	M33	699	029	320	15.5	3.38	0.2	138.5
6	Cam River	M35	806	625	10	14.5	6.9	0.33	131.3
7	Kowai River	M34	843	866	80	14.75	3.63	0.2	218.0
8	Weka Creek	M34	878	970	100	14	4.52	0.29	359.5
9	Upper Eyre	L35	297	697	437	10.45	7.8	0.35	91.8
10	Coopers Creek West	L34	363	713	400	11.3	4.43	0.4	71.4
11	Lower Farm Stream	L34	131	001	500	9.75	3.57	0.26	59.2
12	Sudden Valley Stream	K33	073	025	600	12	4.6	0.32	63.2
13	Andrews Stream	L33	117	011	600	13.3	12.57	0.4	73.8
14	Broken River	K34	043	843	1100	7.5	4.01	0.33	62.9
15	Slovens Stream	L34	165	896	500	14.25	3.42	0.22	160.9
16	Little Kowai River	L35	233	652	400	13.95	4.06	0.3	76.3
17	Cust Main Drain	M35	799	595	10	17.75	8.1	0.36	174.3
18	Silverstream	M35	765	546	15	13.5	4.18	0.5	149.0
19	Kaiapoi River	M35	778	569	10	12.5	4.3	0.78	161.4
20	South Brook Stream	M35	777	647	20	11.5	2.28	0.39	138.2
21	North Brook Stream	M35	795	649	10	13.5	3.67	0.7	129.5
22	Cam River	M35	801	652	10	12.65	2.1	0.7	126.5
23	Waikuku Stream	M35	844	687	10	14.75	2.3	0.49	113.6
24	Taranaki Stream	M35	848	669	10	13.5	1.7	0.6	125.8
25	Ohoka Stream	M35	802	591	10	12.75	4.72	1.01	194.9
26	Kaiapoi River	M35	803	590	10	12.75	9.03	0.62	168.2
27	Ohoka Stream	M35	753	607	20	15.25	2.1	0.27	235.8
28	Saltwater Creek	M34	832	724	20	13	3.73	0.38	131.2
29	Waipara River	N34	922	865	40	15	25	0.57	283.0
30	Waipara River	M34	894	933	80	16.35	12	0.39	205.5
31	Waipara River	M34	830	938	100	17.25	40	0.41	197.1
32	Waipara River	M34	766	941	180	18	21.7	0.5	177.4
33	Birdlings Brook	M36	547	128	12	12.25	2.07	0.36	187.0
34	Silverstream	M36	614	260	15	14.5	3.47	0.19	229.9
35	Styx River	M35	775	493	18	13.5	3	0.43	124.8
36	Ashley River	L34	473	754	350	16	50	0.48	83.8
37	Ashley River	M34	845	700	10	17.3	60	0.3	80.1
38	Grey River	M34	684	818	200	12.5	6.13	0.34	92.5
39	Stony Creek	M34	766	748	150	11.75	1.62	0.13	137.3
40	Makerikeri River	M34	720	787	190	19.5	3.73	0.15	76.5
41	Grey River	M34	715	843	400	12.5	5.72	0.4	89.9
42	Kowai River	M34	797	805	100	13.5	4.63	0.29	113.9
43	Leader River	O32	315	391	90	17.15	6.13	0.31	244.0
44	Hawkswood Stream	O32	368	386	70	13.1	1.5	0.32	118.3
45	Conway River	O32	446	453	20	16.5	19.43	0.29	187.1
46	Limestone Stream	O32	477	441	20	15.4	5.38	0.25	139.8
47	Okarahia Stream	O32	473	517	150	14.5	2.97	0.23	136.3
48	Kaka Mutu Stream	O32	485	534	300	14.75	3.5	0.36	146.5
49	Oaro River	O32	513	545	10	14.75	5.63	0.4	150.5
50	Ote Makura Stream	O32	534	585	10	13.75	7.1	0.24	149.6
51	Kowhai River	O31	621	655	9	14	3.63	0.18	150.2
52	Kowhai River	O31	578	713	160	12.5	10	0.52	120.9
53	Lyell Creek	O31	658	698	10	14.55	2.1	0.42	224.5
54	Trib of Lyall Creek	O31	654	682	10	18	3.13	0.33	264.5
55	Dog Stm	N32	966	543	350	10.7	4.25	0.23	87.2

Site No.	Name	Map #	Easting	Northing	Altitude (m a.s.l)	Temp[1]	Width[1]	Depth[1]	Conductivity[1] $\mu S\ cm^{-1}$
56	Percival River	N32	977	528	350	15.5	3.7	0.35	91.2
57	Chatterton River	N32	946	528	340	15.75	6.73	0.22	61.1
58	Percival River	N32	940	504	308	16.25	8.5	0.4	83.2
59	Hanmer River	N32	927	477	300	15.25	6.4	0.11	101.8
60	Waitohi River	M33	715	168	340	16	7.8	0.46	119.0
61	Waitohi River	M33	883	132	210	16.5	14.67	0.35	153.8
62	Styx River	M35	817	490	10	14.5	10	0.84	124.4
63	Dudley Creek	M35	828	441	10	18.25	3.53	0.16	156.7
64	Ant Stream	L33	370	080	760	9	8.9	0.6	39.4
65	Esk River	L33	405	080	750	13.5	11.4	0.57	43.1
66	Lillburn River	L34	385	940	700	12.25	8.5	0.61	46.0
67	Cox River	L33	240	170	710	11.25	21.3	0.55	64.5
68	Upper Ashley River	L34	435	976	700	14	9	0.55	51.5
69	Boyle River	M32	594	545	570	12.75	23.6	0.43	94.7
70	Hope River	M32	650	465	440	14.5	11.6	0.46	90.2
101	Halswell River	M36	741	284	10	13.75	7	120	122.6
102	Kaituna River	M36	867	202	60	14.3	3.3	0.43	63.3
103	Selwyn River	M36	626	234	12	14.35	14.7	0.29	204.6
104	Irwell River	M36	563	218	28	14.75	7.03	0.37	164.9
105	Peraki Creek	N37	959	059	20	11.3	4.6	0.32	133.6
106	Okuti River	N36	945	135	35	12	5.2	0.4	131.0
107	Opuahou Stream	N36	968	183	80	13.9	3.02	0.4	142.2
108	Opara Stream	N36	128	216	15	11.6	6	0.38	176.5
109	Opara Stream	N36	101	201	150	10.95	3.35	0.43	152.3
110	Little Akaloa River	N36	087	256	20	13.45	4.65	0.24	203.0
111	Little Akaloa River	N36	075	241	100	10.5	3.65	0.22	117.0
112	Pigeon Bay Stream	N36	019	218	60	11.6	4.4	0.27	125.7
113	Pigeon Bay Stream	N36	017	235	20	11.9	3.77	0.19	141.1
114	Okana River	N36	936	154	5	12.1	4.6	0.22	123.4
115	Ackeron River	K35	957	556	360	14.25	5.7	0.45	78.5
116	Scamander Stream	K35	932	657	500	11.7	2.8	0.26	103.9
117	No name (Mt Hutt) stream	K35	997	406	390	16.25	1.83	0.33	76.7
118	Glenrock Riverr	K35	784	627	600	9.1	2.9	0.22	101.0
119	Redcliffe Steam	K35	874	573	435	10.95	2.4	0.13	63.9
120	North Ashburton River	K36	937	268	345	17.25	5.3	0.17	63.1
121	Charlie Stream	J35	606	569	680	14.35	4.6	0.24	51.4
122	Smite River	J35	647	519	770	14.05	3.9	0.24	48.5
123	Gentleman Smith Stream	J36	628	381	660	18.85	2.4	0.28	56.0
124	Bowyers River	K36	825	298	460	9.55	13	0.3	45.9
125	Moorhouse Stream	J36	612	205	530	13.5	3.2	0.2	67.8
126	Blue Duck Stream	K36	705	293	540	12.5	3.7	0.35	65.6
127	Hinds River	K37	963	892	85	16.5	13.1	0.37	87.4
128	Avon River	M35	805	422	0	13.25	14.55	0.47	177.7
129	Upper Avon River (UCSA)[2]	M35	763	425	5	9.575	4.7	0.43	178.9
130	Waimairi Stream	M35	770	428	10	13.75	3.2	0.17	170.6
131	Wairarapa Stream	M35	772	435	10	15	4.7	0.22	142.2
132	Bush Stream	J36	374	292	590	12.1	4.9	0.2	31.2
133	Scour Stream	J36	400	258	540	13.85	3.9	0.29	40.1
134	Forest Creek	J36	420	203	560	16.25	2.3	0.21	33.5
135	Owhetoro Stream	N36	954	267	20	12.5	2.6	0.24	125.1
136	Owhetoro Stream	N36	948	238	160	11.35	3.5	0.27	112.2
137	Hukahuka Turoa Stream	N36	937	178	40	11.85	4.3	0.26	127.5
138	Kaituna Stream	M36	834	164	20	15.55	2.87	0.63	132.7
139	Prices Stream	M36	854	134	20	15.4	2.45	0.53	142.1
140	Prices Stream	M36	874	158	60	12.5	4.52	0.22	116.9
141	Upper Heathcote River	M36	768	394	20	16	5.7	0.36	248.3
142	Upper Selwyn River	L35	206	494	300	11.25	19.6	0.45	88.2
143	Whitecliffs Rd Stream	L35	194	496	300	11.65	4.17	0.26	146.9
144	Whitecliffs Rd Stream	L35	207	477	280	12.95	4.8	0.32	74.1
145	Upper Selwyn River	L35	259	462	240	12.4	5.17	0.38	141.0

Site No.	Name	Map #	Easting	Northing	Altitude (m a.s.l)	Temp[1]	Width[1]	Depth[1]	Conductivity[1] $\mu S\ cm^{-1}$
146	Wainiwaniwa River	L35	285	475	220	15.5	6.9	0.21	128.8
147	Wainiwaniwa River	L35	268	507	240	12.5	3.4	0.15	182.5
148	Wainiwaniwa River	L35	279	520	260	12.9	4.3	0.21	113.2
149	Wainiwaniwa River	L35	276	534	280	15.1	4.7	0.21	140.3
150	Hawkins River	L35	317	544	280	14.3	5.1	0.33	91.8
151	Dog Brook Stream	N32	120	414	160	14.65	3.8	0.25	243.0
152	Mason River	N32	247	558	430	13.75	6.6	0.3	129.4
153	Conway River	O31	324	623	350	13.2	6.97	0.26	131.8
154	Charwell River	O31	398	652	480	14.25	6.2	0.39	110.0
155	Kahutara River	O31	469	691	180	13	4.7	0.35	149.6
156	Kahutara River	O31	546	665	150	15.95	6.6	0.38	177.0
157	Black Miller Stream	P31	804	865	10	10.5	3.4	0.31	123.2
158	Ohau River	P31	785	845	60	10	4.7	0.25	117.7
159	Blue Duck Stream	P31	732	808	20	13.4	2.85	0.2	359.0
160	Hapuka River	O31	691	773	60	15.05	7.4	0.29	212.1
161	Middle Creek	O31	642	711	30	13.95	2.63	0.44	151.0
162	Middle Creek	O31	658	706	10	14.75	6.4	0.47	155.9
163	Waiangarara River	O31	638	754	200	10.6	3.7	0.33	94.5
164	Lowry Drain	N33	045	244	160	14.15	1.5	0.36	100.8
165	School Stream	N33	977	263	170	13.75	4.2	0.2	297.5
166	Pahau River	N33	960	233	170	16.25	9.7	0.37	83.3
167	Dry Stream	N33	950	215	180	17.1	5.78	0.26	172.9
168	Hurunui River	N33	910	149	190	16	9.7	0.58	153.0
201	Otaio River	J39	555	315	110	15.55	3.8	0.133	148.3
202	Otaio River (SH1 bridge)	J39	653	274	5	14.65	2.4	0.103	143.1
203	North Opuha River	J37	360	10	650	10.5	7.3	0.46	26.8
204	Phantom River	J37	510	80	530	14	7.5	0.26	55.0
205	Hewson River	J36	551	89	480	14.5	11	0.39	51.7
206	Orari River	J37	500	20	540	14.5	8.4	0.88	48.4
207	Ribbonwood Creek	J37	354	913	400	13	5.4	0.33	53.6
208	Halls Stream	J38	329	765	320	12.95	4.2	0.19	71.8
209	Tengawai River	J38	349	671	290	12.8	9.4	0.52	75.3
210	Firewood Stream	I37	254	847	580	10	7.4	0.43	41.8
211	Little Opawa River	J38	385	625	320	16.55	3.3	0.25	123.5
212	Opawa River	J38	326	578	400	13.1	4.5	0.25	70.6
213	Rocky Gully Stream	J38	365	580	300	15.4	8.8	0.33	86.7
214	Raincliff Stream	J38	492	690	180	16.15	5.1	0.27	170.7
215	White Rock River	J39	424	454	220	13.95	10.5	0.32	89.1
216	Elder Stream	J39	470	374	185	14.6	8.7	0.24	76.5
217	Otaio Gorge	J39	446	290	260	14.25	8.7	0.25	90.0
218	Upper Makikihi River	J40	488	190	240	10.25	2.4	0.11	99.7
219	Gunns Bush Stream	J40	493	139	250	10	3.4	0.13	94.2
220	Waimate Creek	J40	486	105	220	10.25	4.1	0.22	79.1
221	Waikakahi Stream	J41	558	900	48	16.45	7.8	0.64	171.0
222	North Branch Waihao River	J40	375	200	520	13.75	13.8	0.29	42.5
223	South Waihao River	J40	357	106	300	13.95	7.23	0.3	72.1
224	Waihi Gorge	J37	615	879	310	11.25	9.1	0.5	60.3
225	Hae Hae Te Moana	J37	513	829	360	12.25	8.1	0.37	61.9
226	Te Moana	J38	676	736	95	14	11.3	0.44	87.7
227	Ohapi Creek	K38	741	698	50	11.35	2.8	0.52	73.5
228	Ohapi Creek	K38	750	695	48	11.95	3.7	0.48	66.6
229	Ohapi Creek	K38	812	618	5	12.75	6	0.53	90.3
230	North branch Ohapi Creek	K38	776	659	25	12.9	3.8	0.55	73.5
231	South Ohapi Creek	K38	763	645	20	13.55	2.8	0.58	81.5
232	Ohapi Creek	K38	762	644	20	14.25	5.4	0.63	91.3
233	Orakipaoa	K38	772	591	5	11.75	5.5	0.75	147.9
234	Ohapi Creek	K38	736	668	36	11.5	4.4	0.26	79.2
235	Old Orari Lagoon Outfall	K38	828	617	3	15.7	5.5	0.87	275.6
236	Setlement -Parke Road drain	K38	822	627	5	15.35	1.7	0.14	287.4
237	Rhodes Stream	K38	828	633	5	17	4.1	0.4	224.1

Site No.	Name	Map #	Easting	Northing	Altitude (m a.s.l)	Temp[1]	Width[1]	Depth[1]	Conductivity[1] $\mu S\ cm^{-1}$
238	Fitzgeralds Drain	K38	787	668	25	16.85	6.4	0.2	126.8
239	Pareora River	J39	667	334	10	15.6	33.9	0.52	127.1
240	Pareora River	J39	618	371	30	15.25	11	0.24	124.4
241	Pareora River	J39	581	403	70	15.25	12.8	0.37	122.5
242	Pareora River	J39	543	424	85	16.1	9.7	0.29	133.0
243	Pareora River	J39	487	463	160	15.1	12.2	0.46	99.2
244	Pareora River	J39	540	437	90	16.45	20.3	0.48	108.3
245	South Waihao River	J40	449	23	115	15.55	7.1	34	117.8
246	Waihao River	J40	471	2	85	16.85	11.4	27.3	65.1
247	Waihao River	J40	496	988	75	16.3	13.8	0.28	84.9
248	Whitney's Creek	J41	617	882	27	19.35	4	0.24	280.3
249	Waihao River	J40	643	15	4	18.1	10	0.36	88.1
250	Buchanans Creek	J40	634	17	5	12.95	2.8	0.77	135.3
251	Duck Stream	J38	260	693	500	12.95	3.2	0.2	76.5
252	Opihi River	I38	222	772	530	12.9	2.7	0.29	87.4
253	Opihi River	J38	366	767	295	13.6	11	0.58	61.1
254	South Opuha River	J37	310	903	560	11	11.2	0.5	27.2
255	South Opuha River	J37	373	882	420	12.75	11.1	0.44	29.0
256	North Opuha River	J37	396	928	430	13.75	16.5	0.25	39.7
257	Allandale Stream	J38	378	773	298	13	1.4	0.22	70.7
258	Coal Stream	J38	390	726	260	18.95	3.1	0.17	87.2
259	Opihi River	J38	457	690	180	17.6	11.3	0.53	76.2
260	Tengawai River	J38	595	608	70	17.4	17.7	0.29	112.1
261	Opihi River	J38	603	603	70	17.8	36.7	0.73	76.4
262	Rhodes Stream	K38	809	654	5	18.75	2.4	0.22	173.2
263	Petries Drain	K38	801	656	7	15.35	3.4	0.28	177.8
264	Coopers Creek	K37	720	866	176	14.35	6	29.3	102.8
265	Waihi River	K38	721	682	43	12.6	11.1	0.32	102.2
266	Kakahu River	J38	683	672	50	18.35	7.8	0.32	150.5
267	Temuka River	K38	726	600	8	15.65	19.5	0.4	108.1
268	Opihi River	K38	722	591	14	16.9	32.6	0.57	65.8
269	Opihi River	K38	769	582	4	17	53.3	0.76	75.8
270	Hook River	J40	631	131	7	12.25	6.6	0.23	145.2
271	Hook River	J40	532	153	119	12.45	10.1	0.48	84.1
272	Hook River	J40	493	161	200	10.5	6.8	0.25	70.5
273	Makikihi River	J40	582	200	50	14.55	11.2	0.27	116.9
274	Waikakahi Stream	J41	596	862	25	15.15	5.1	0.91	151.3
275	Elephant Hill Stream	J40	394	973	160	17.45	4.8	0.33	205.1
276	Penticotico Stream	I40	231	965	160	14.45	8.2	0.7	94.3
277	Hakataramea River	I40	198	136	260	15.15	26.8	0.61	60.6
278	Hakataramea River	I39	225	205	340	15.3	21.8	0.39	55.3
279	Marewhenua	I40	273	920	180	14.45	12.4	0.43	80.4
280	Marewhenua	I41	197	820	240	13.5	10.1	55.7	56.6
281	Otikiake River	I40	179	953	180	13.85	10.6	0.36	101.1
282	Awakino Stream	I40	61	49	380	17.95	5.8	0.3	45.1
283	Otemata	H40	879	188	270	15.7	23.3	0.47	32.1
284	Omarama Stream	H39	681	309	430	15.05	7.7	0.54	57.8
285	Omarama Stream	H39	620	185	620	16.65	7	0.14	37.7
286	Ahuriri River	H39	542	288	560	14.25	26.3	0.51	39.1
287	Quail Burn Stream	H39	655	355	455	19.3	6.8	22.7	34.2
288	Fork Stream	I37	25	865	750	10.6	12.8	0.38	32.4
289	Irishman Creek	I38	977	767	540	14.75	5.7	0.42	25.1
290	Mary Burn Stream	I38	960	669	517	16.25	5.1	0.23	58.5
291	Twizel River	H38	794	574	455	15.75	20.4	0.47	23.9
292	Spring Creek	H39	726	478	500	14.45	3.1	0.24	56.0

[1], values are average of 2 sampling occasions

2, University of Canterbury Students Association

APPENDIX FOUR
LIST OF INVERTEBRATE TAXA FOUND IN THE STUDY

INSECTA

Ephemeroptera
Ameletopsis
Atalophlebioides
Austroclima
Coloburiscus
Deleatidium
Mauiulus
Neozephlebia
Nesameletus
Oniscigaster
Rallidens
Zephlebia

Plecoptera
Acroperla
Austroperla
Megaleptoperla
Cristaperla
Spaniocerca
Spaniocercoides
Stenoperla
Zelandobius
Zelandoperla

Trichoptera
Aoteapsyche
Beraeoptera
Costachorema
Helicopsyche
Hudsonema
Hydrobiosella
Hydrobiosidae (indet.)
Hydrobiosis
Hydrochorema
Neurochorema
Oecetis
Oeconesus
Olinga
Oxyethira
Paroxyethira
Philorheithrus
Plectrocnemia
Polycentropodidae (indet.)
Polyplectropus
Psilochorema
Pycnocentria
Pycnocentrodes
Triplectides
Zelandopsyche

Diptera
Aphrophila
Austrosimulium
Ceratopogonidae
Culicidae
Diptera pupae (indet.)
Empididae
Ephydridae
Eriopterini
Hexatomini
Limoniinae
Mischoderus
Molophilus
Muscidae
Neocurupira
Nothodixa
Paradixa
Paralimnophila
Pelecorhynchidae
Peritheates
Psychodidae
Stratiomyidae
Tabanidae
Tanyderidae
Tipulidae pupae
Zelandotipula

Chironomidae
Chironomidae pupae
Chironominae
Diamesinae
Maoridiamesa
Orthocladiinae
Podonominae
Tanypodinae

Coleoptera
Antiporus
Berosus
Paracalliope
Huxelhydrus
Hydraenidae
Hydrophilidae
Liodessus
Orchymontia
Ptilodactylidae
Rhantus
Scirtidae

Lepidoptera
Hygraula

Megaloptera
Archichauliodes

Odonata
Xanthocnemis
Procordulia
Austrolestes
Hemicordulia

Mecoptera
Nannochoristidae

MOLLUSCA
Austropeplea
Gyraulus
Physa
Pisidium/Sphaerium
Potamopyrgus

CRUSTACEA
Cladocera
Copepoda
Cruregens
Isopoda
Ostracoda
Paracalliope
Paracorophium
Paraleptamphopus
Paranephrops
Paratya
Phreatogammarus

OLIGOCHAETA

NEMATODA

PLATYHELMINTHES
Cura
Neppia

HIRUDINEA
Glossiphoniidae

ACARINA

HEMIPTERA
Sigara
Anisops

APPENDIX FIVE
LIST OF 10 INVERTEBRATE METRIC SCORES FOR EACH SITE

Site No.	No. taxa[1]	MCI	QMCI	SQMCI	OQMCI	# EPT Taxa	%Dominant taxa	# Ephemeroptera	# Plecoptera	# Trichoptera
1	19	107.4	6.0	6.4	6.9	10	39.4	1	0	9
2	15	96.0	7.3	7.8	7.6	7	70.5	1	0	6
3	23	119.1	6.2	6.0	6.3	13	66.4	2	1	10
4	27	119.3	7.7	8.3	6.8	15	25.2	3	1	11
5	23	86.1	3.3	3.3	4.7	6	27.8	1	2	3
6	19	72.6	2.5	2.5	3.9	7	43.1	0	0	7
7	17	78.8	2.6	2.7	4.9	6	41.5	1	0	5
8	23	80.9	3.4	3.6	4.7	5	35.7	1	0	4
9	22	110.9	7.0	7.1	7.0	11	30.1	1	1	9
10	19	113.7	6.9	7.0	7.0	9	34.8	2	2	5
11	25	120.8	7.0	7.7	7.6	14	59.8	3	3	8
12	17	123.5	7.8	7.9	8.0	9	88.1	2	2	5
13	19	133.7	7.7	7.9	7.9	12	60.7	2	3	7
14	27	111.9	7.2	7.7	7.4	14	54.7	2	3	9
15	19	84.2	3.3	3.5	4.0	7	40.4	1	0	6
16	15	129.3	7.2	7.7	7.5	11	62.7	1	1	9
17	21	96.2	6.1	6.2	7.0	11	32.5	1	0	10
18	18	103.3	5.6	5.8	7.0	11	32.4	1	0	10
19	17	78.8	4.6	4.9	4.6	5	78.1	1	0	4
20	24	89.2	4.4	4.2	4.0	11	76.8	1	0	10
21	11	78.2	3.8	4.0	3.7	3	76.5	0	0	3
22	20	92.0	4.1	4.2	4.4	9	40.4	1	0	8
23	21	101.0	5.4	5.3	5.7	11	30.7	1	0	10
24	18	70.0	4.1	4.3	4.0	6	43.8	1	0	5
25	19	95.8	6.1	5.7	6.0	9	41.7	1	0	8
26	21	94.3	5.9	5.4	5.5	10	35.1	1	0	9
27	14	72.9	2.5	2.6	2.7	6	40.7	1	0	5
28	22	95.5	4.6	4.3	4.8	11	57.3	1	0	10
29	26	86.9	3.8	3.9	5.0	8	19.1	1	0	7
30	17	83.5	4.9	5.2	6.1	6	35.1	1	0	5
31	21	81.9	4.7	4.9	5.5	7	27.0	1	0	6
32	21	93.3	5.4	5.2	5.5	9	36.2	1	0	8
33	19	94.7	5.7	6.3	6.5	9	28.8	1	0	8
34	18	107.8	5.2	4.7	5.9	9	33.3	1	0	8
35	17	96.5	5.5	5.8	6.8	9	48.4	1	0	8
36	16	116.3	6.8	6.8	7.2	9	43.0	1	0	8
37	19	95.8	7.3	7.8	7.7	9	73.5	2	0	7
38	23	112.2	6.1	6.5	6.5	13	49.3	3	1	9
39	15	60.0	2.3	2.3	4.6	3	46.6	0	0	3
40	15	84.0	2.6	2.5	5.7	5	42.2	1	0	4
41	25	116.8	6.2	6.6	6.6	12	37.2	3	1	8
42	21	93.3	6.5	6.6	7.1	9	50.2	1	0	8
43	15	98.7	6.7	6.8	6.9	7	43.6	1	0	6
44	24	90.0	3.5	3.0	5.3	6	48.5	1	0	5
45	11	118.2	7.5	7.9	7.7	7	74.9	1	0	6
46	31	114.8	6.8	7.6	7.4	17	62.7	5	2	10
47	21	119.0	6.8	6.9	7.1	14	37.3	3	1	10
48	25	116.0	7.1	7.3	7.1	14	41.6	2	2	10
49	28	106.4	6.5	7.4	7.3	15	60.7	3	1	11
50	19	104.2	4.7	4.3	4.6	10	65.9	2	1	7
51	15	113.3	7.7	7.9	7.9	8	84.3	1	1	6
52	14	124.3	7.7	7.9	8.0	10	86.7	1	1	8
53	22	85.5	3.5	3.4	3.8	9	22.8	1	0	8
54	21	84.8	3.1	2.9	3.3	7	32.8	1	0	6
55	31	109.0	6.1	6.6	6.0	15	28.2	2	3	10
56	24	95.8	5.2	5.6	5.6	11	31.3	1	1	9

Site No.	No. taxa[1]	MCI	QMCI	SQMCI	OQMCI	# EPT Taxa	%Dominant taxa	# Ephemeroptera	# Plecoptera	# Trichoptera
57	25	109.6	7.0	7.6	7.2	13	59.9	3	1	9
58	21	102.9	6.5	6.5	7.2	11	49.8	1	1	9
59	19	113.7	7.2	7.5	7.4	12	63.4	2	1	9
60	28	102.9	5.2	5.2	5.9	12	36.8	3	0	9
61	26	109.2	6.4	6.3	6.6	10	52.9	1	1	8
62	21	75.2	2.6	2.9	3.8	5	20.2	0	0	5
63	12	65.0	3.0	2.8	2.6	0	43.8	0	0	0
64	22	135.5	7.7	7.9	7.8	14	67.7	3	4	7
65	21	123.8	7.3	7.4	7.4	12	43.9	2	2	8
66	20	115.0	6.9	7.0	7.4	12	50.2	2	2	8
67	18	130.0	7.7	7.9	8.0	13	81.9	2	3	8
68	20	120.0	7.4	7.9	7.6	11	53.8	2	4	5
69	21	110.5	6.8	7.7	7.4	10	56.4	2	1	7
70	23	107.0	5.9	6.1	7.0	13	43.3	2	1	10
101	14	65.7	3.4	3.4	4.0	4	45.4	0	0	4
102	22	123.6	6.1	6.4	6.0	13	32.9	2	0	11
103	20	94.0	4.8	4.9	6.2	10	52.5	1	0	9
104	23	88.7	3.1	2.8	5.4	9	27.1	1	0	8
105	18	93.3	3.6	3.9	4.2	9	50.7	1	0	8
106	24	112.5	6.0	6.0	6.4	11	32.7	3	0	8
107	30	98.7	4.8	4.8	5.8	13	34.2	1	0	12
108	20	91.0	2.9	3.1	3.8	8	22.9	2	0	6
109	22	109.1	5.5	5.2	4.9	10	33.0	2	0	8
110	26	90.8	3.6	3.6	4.2	10	17.5	1	0	9
111	16	110.0	6.0	6.3	5.0	6	34.3	2	0	4
112	23	94.8	4.2	4.1	5.2	13	22.8	2	0	11
113	22	90.0	3.4	3.1	4.3	11	22.6	1	0	10
114	26	96.9	4.2	4.0	4.3	13	55.1	2	0	11
115	22	100.9	4.9	5.0	5.2	8	41.4	1	1	6
116	24	97.5	4.6	4.1	5.9	12	25.6	2	0	10
117	30	82.7	2.9	2.3	3.7	7	27.3	0	0	7
118	18	121.1	6.1	6.0	6.6	9	26.7	3	0	6
119	18	103.3	6.2	5.8	7.0	6	48.3	1	1	4
120	23	101.7	6.3	7.4	7.1	10	58.8	2	1	7
121	19	94.7	6.8	7.6	7.1	8	63.2	1	1	6
122	15	117.3	7.4	7.8	7.9	8	78.7	1	1	6
123	22	108.2	5.0	4.9	5.3	11	44.7	2	0	9
124	29	113.8	6.9	7.7	7.2	14	50.7	1	2	11
125	20	91.0	5.9	6.2	6.8	9	47.0	1	0	8
126	21	101.0	6.1	6.0	6.3	10	19.3	2	0	8
127	17	111.8	6.5	7.0	6.7	9	34.0	2	0	7
128	15	64.0	3.6	3.9	3.5	1	65.9	0	0	1
129	15	64.0	3.3	3.8	3.1	2	63.8	0	0	2
130	15	66.7	3.3	3.8	3.1	2	66.7	0	0	2
131	13	78.5	3.1	3.8	3.3	2	51.6	1	0	1
132	23	104.3	6.1	7.5	7.0	12	53.5	1	3	8
133	24	121.7	6.8	7.1	7.5	13	45.6	2	1	10
134	16	118.8	7.6	7.9	7.9	11	79.8	1	1	9
135	24	98.3	3.9	3.9	3.9	12	67.1	1	0	11
136	23	98.3	6.5	5.4	5.8	9	24.0	1	1	7
137	23	113.0	5.9	5.6	6.1	11	22.1	2	0	9
138	10	66.0	3.7	3.9	3.2	0	79.4	0	0	0
139	19	74.7	2.7	2.5	3.2	4	34.2	1	0	3
140	20	101.0	5.2	5.0	5.7	11	21.3	2	0	9
141	10	62.0	2.0	1.9	2.8	0	49.3	0	0	0
142	22	119.1	6.9	6.8	6.7	10	19.1	2	1	7
143	14	81.4	3.4	3.9	3.1	5	63.3	1	0	4
144	15	73.3	3.5	3.9	3.2	5	74.9	0	0	5
145	24	110.8	5.9	5.4	5.5	12	40.3	2	1	9
146	10	66.0	2.6	3.1	4.3	1	44.6	0	0	1

Site No.	No. taxa[1]	MCI	QMCI	SQMCI	OQMCI	# EPT Taxa	%Dominant taxa	# Ephemeroptera	# Plecoptera	# Trichoptera
147	15	61.3	3.0	3.1	3.4	4	29.0	0	0	4
148	19	85.3	3.1	3.3	3.9	5	42.1	1	0	4
149	16	82.5	3.1	3.3	3.6	6	31.3	1	0	5
150	22	90.9	2.6	2.9	3.1	9	46.8	1	0	8
151	29	89.7	4.5	5.7	6.1	6	26.8	1	0	5
152	19	95.8	5.8	6.4	6.1	8	25.1	2	0	6
153	16	102.5	5.0	4.8	6.2	6	31.5	1	0	5
154	23	105.2	6.9	7.6	7.3	11	58.4	1	0	10
155	23	110.4	7.3	7.8	7.4	12	61.4	2	1	9
156	15	101.3	7.5	7.8	7.8	8	79.4	1	0	7
157	26	119.2	7.1	7.0	6.9	11	34.1	3	3	5
158	15	126.7	8.8	8.7	7.4	8	47.4	2	1	5
159	22	95.5	2.8	2.4	4.5	10	35.7	1	1	8
160	20	93.0	6.5	7.5	7.2	6	64.4	1	0	5
161	18	96.7	4.2	4.1	5.0	9	42.6	1	1	7
162	22	84.5	4.0	4.0	4.9	9	23.4	1	0	8
163	19	131.6	7.7	7.9	7.3	11	56.5	2	3	6
164	16	81.3	3.9	4.0	4.0	7	58.1	1	0	6
165	18	80.0	2.9	2.9	3.9	7	31.9	1	0	6
166	10	102.0	6.6	6.3	6.6	6	62.4	1	0	5
167	21	81.0	3.4	3.2	4.5	7	25.4	1	0	6
168	24	108.3	5.8	5.7	6.4	12	40.2	2	1	9
201	21	103.8	5.2	6.6	6.7	9	28.8	2	1	6
202	36	94.4	4.9	4.7	5.9	12	29.3	1	1	10
203	30	120.7	5.9	5.9	7.2	18	34.1	4	2	12
204	22	111.8	6.2	6.3	7.1	11	33.5	1	1	9
205	24	111.7	6.0	6.5	6.8	12	31.3	1	1	10
206	24	103.3	6.3	6.2	7.0	12	34.4	1	1	10
207	19	108.4	6.0	6.2	6.7	8	33.6	1	1	6
208	17	101.2	6.5	7.5	7.0	8	58.9	1	0	7
209	27	125.2	6.9	7.0	6.9	15	39.6	3	1	11
210	26	121.5	6.8	6.8	7.3	15	47.4	3	3	9
211	28	88.6	3.4	2.7	4.1	8	26.4	1	0	7
212	27	116.3	6.5	6.7	6.7	14	23.3	3	2	9
213	25	111.2	5.4	5.4	6.2	13	43.9	2	1	10
214	27	79.3	3.3	3.7	4.1	8	25.4	1	0	7
215	25	117.6	6.4	6.8	6.4	13	26.1	2	1	10
216	26	120.8	6.8	6.9	7.0	14	33.5	3	2	9
217	34	124.7	6.5	6.8	6.6	19	27.2	6	3	10
218	39	114.9	7.0	6.7	6.6	19	20.3	4	4	11
219	40	117.5	7.1	6.5	6.8	21	15.7	4	6	11
220	29	111.7	5.6	5.7	5.8	14	25.0	3	3	8
221	28	85.0	3.7	3.2	3.8	10	42.4	1	0	9
222	34	120.0	6.4	7.0	6.9	19	26.4	7	1	11
223	38	104.2	4.5	4.3	4.9	19	30.7	5	1	13
224	27	125.2	7.1	7.3	6.8	16	32.3	4	2	10
225	22	117.3	6.4	6.3	6.7	12	36.8	3	1	8
226	24	117.5	6.8	7.7	7.4	13	54.4	3	1	9
227	18	78.9	4.0	4.7	4.2	5	58.6	1	0	4
228	24	107.5	5.1	4.6	5.3	11	24.4	2	0	9
229	25	96.0	3.9	3.2	4.5	12	24.3	1	0	11
230	26	101.5	5.0	4.7	5.1	13	28.3	3	0	10
231	20	90.0	3.5	3.2	3.6	9	31.4	1	0	8
232	24	98.3	4.1	3.8	4.1	12	25.5	2	0	10
233	25	86.4	4.1	4.8	4.2	11	51.1	1	0	10
234	25	88.0	5.1	5.9	5.2	10	34.1	1	0	9
235	21	68.6	3.7	3.9	3.4	5	76.9	0	0	5
236	14	62.9	3.1	3.0	3.6	2	33.0	0	0	2
237	25	85.6	2.9	2.8	3.3	9	30.5	1	0	8
238	27	101.5	5.5	5.2	5.7	11	35.2	1	0	10

Site No.	No. taxa[1]	MCI	QMCI	SQMCI	OQMCI	# EPT Taxa	%Dominant taxa	# Ephemeroptera	# Plecoptera	# Trichoptera
239	28	86.4	4.0	3.9	4.3	9	26.6	1	0	8
240	26	92.3	4.0	4.0	4.8	8	17.2	2	0	6
241	27	88.9	4.9	4.6	6.1	9	30.4	1	0	8
242	23	93.9	5.5	5.2	6.4	10	29.7	1	0	9
243	25	116.8	6.5	6.9	7.1	13	49.6	3	1	9
244	31	100.6	4.9	4.9	5.3	13	18.1	2	0	11
245	24	82.5	3.4	3.8	4.0	8	52.8	1	0	7
246	20	67.0	2.5	2.5	4.4	3	26.2	0	0	3
247	22	75.5	3.4	3.8	4.2	5	51.6	1	0	4
248	21	70.5	3.2	3.0	3.3	3	31.2	0	0	3
249	19	71.6	4.2	4.4	4.8	5	37.7	0	0	5
250	21	101.0	5.2	5.7	5.4	11	28.6	1	0	10
251	24	116.7	6.7	6.8	7.1	11	41.2	1	1	9
252	27	96.3	3.9	3.4	4.2	9	31.1	2	0	7
253	24	110.8	7.3	7.8	7.6	11	69.4	2	2	7
254	27	118.5	6.8	7.6	7.4	14	56.7	3	3	8
255	23	111.3	6.1	6.6	7.2	13	33.7	3	0	10
256	21	125.7	6.8	7.0	7.3	12	48.4	4	1	7
257	30	102.7	5.4	5.6	5.2	14	19.9	2	0	12
258	14	94.3	5.6	6.0	5.9	6	31.7	1	0	5
259	21	114.3	6.0	6.5	6.5	12	31.3	2	1	9
260	23	102.6	6.1	6.1	6.9	11	36.8	1	0	10
261	19	107.4	6.1	6.4	7.2	11	43.7	1	0	10
262	26	85.4	3.5	3.4	4.0	9	47.8	1	0	8
263	23	91.3	3.7	4.3	4.9	9	22.8	1	0	8
264	25	96.8	5.1	4.6	5.8	12	23.4	2	0	10
265	30	108.0	5.7	5.7	6.1	15	30.2	3	1	11
266	27	82.2	2.7	3.2	3.9	9	31.9	0	0	9
267	26	83.1	3.4	3.5	3.5	8	49.5	1	0	7
268	29	100.7	5.3	5.1	6.7	13	57.4	2	0	11
269	28	88.6	5.1	5.2	5.8	11	33.9	1	0	10
270	30	96.0	5.1	4.9	5.4	12	44.4	1	0	11
271	27	111.1	6.9	6.7	7.1	11	49.8	2	1	8
272	32	120.6	7.4	7.5	7.2	17	43.5	4	3	10
273	31	96.8	4.3	4.2	4.6	13	35.6	2	1	10
274	24	81.7	3.4	3.2	3.4	8	30.8	0	0	8
275	23	85.2	2.1	2.1	3.5	9	44.2	1	0	8
276	29	98.6	4.7	4.7	4.8	11	49.6	1	0	10
277	28	92.9	3.8	4.4	4.6	11	25.0	1	0	10
278	32	110.0	5.2	4.9	5.8	16	29.2	3	1	12
279	22	108.2	5.7	5.9	5.7	9	66.5	3	0	6
280	32	103.1	5.1	5.3	5.7	14	38.5	4	0	10
281	20	115.0	6.4	6.3	6.3	11	51.6	2	1	8
282	24	106.7	5.3	5.1	6.3	10	26.2	2	1	7
283	24	109.2	5.7	6.1	6.7	12	23.9	2	1	9
284	32	96.3	4.8	5.0	5.8	12	30.4	2	0	10
285	28	112.1	5.8	6.4	6.5	14	23.8	3	1	10
286	25	108.0	7.0	7.7	7.5	15	59.2	4	1	10
287	24	95.0	3.4	3.5	4.9	10	23.6	2	0	8
288	27	117.8	6.7	7.6	7.4	16	54.7	4	4	8
289	32	119.4	6.0	6.3	6.5	18	23.5	5	0	13
290	27	102.2	5.2	5.0	5.7	13	28.3	2	0	11
291	25	116.8	6.8	6.8	7.1	14	39.8	3	1	10
292	29	101.4	4.7	4.7	5.3	14	34.5	2	0	12

[1], combined taxon richness from the two sampling periods

APPENDIX SIX

RELATIONSHIPS BETWEEN QMCI AND SQMCI, AND THE ENVIRONMENTAL AND PHYSICO-CHEMICAL VARIABLES SELECTED IN CHAPTER FIVE

The purpose of this appendix is to show that relationships between QMCI and SQMCI scores and environmental and physico-chemical variables were very similar to those between MCI and these variables. Thus, only MCI data were presented in Chapter 5.

Quantitative Macroinvertebrate Community Index (QMCI)

The QMCI is similar to the MCI except that it uses quantitative abundance data. This implies that it is more sensitive to picking up subtle changes in macroinvertebrate community composition (Stark 1998).

The QMCI frequency distribution for all samples collected is shown in Chapter 4; Fig. 4.5. QMCI values ranged from 2.0 to 8.8. The median QMCI value was 5.4 and the average was 5.3. The sites with the cleanest water were the Cox River (QMCI = 7.7, Site # 67), Sudden Valley Stream (QMCI = 7.8, Site # 12) and Ohau Stream (QMCI = 8.8, Site # 158). The sites designated most degraded were the Upper Heathcote River (QMCI = 2.0, Site # 141), Elephant Hill Stream (QMCI = 2.1, Site # 275) and Stony Creek at Te Puke Road (QMCI = 2.3, Site # 39). The clean water sites were found in indigenous forest, had low accumulations of sediment and a heterogeneous substrate composition. They were generally found at high altitudes with steep topography (>20^0). The most degraded sites occurred in pasture or exotic forest catchments, generally at lower altitudes. *Deleatidium* dominated the majority of the "clean" sites, whereas *Potamopyrgus* dominated most of the sites considered to be "highly degraded".

As with the MCI, higher quality QMCI sites were generally further inland at higher altitudes, and more degraded sites were closer to populated areas. QMCI and altitude were significantly correlated (r = 0.47, P<0.001, Fig. 1a), although a reasonable number of sites at lower altitudes had high QMCI values. Of note is Ohau Stream (Site # 158), which had the highest QMCI value (8.8), but was at an altitude of only 60 m a.s.l. Ohau stream is set in steep catchment of indigenous forest, has low accumulations of sediment and a heterogeneous substrate composition, indicating that factors associated with altitude (e.g., landuse intensity is generally higher at low altitudes), rather than altitude *per se* affects stream health. Sites at altitudes between 0 and 600 m a.s.l. incorporated the full range of QMCI values.

Streams in higher altitude bands generally had higher QMCI values (Fig. 1b). Streams in the 0-200 and 200-500 m a.s.l bands covered the full range of taxon richness and QMCI values, indicating that healthy streams are attainable at low altitudes, where anthropogenic development is frequently greatest.

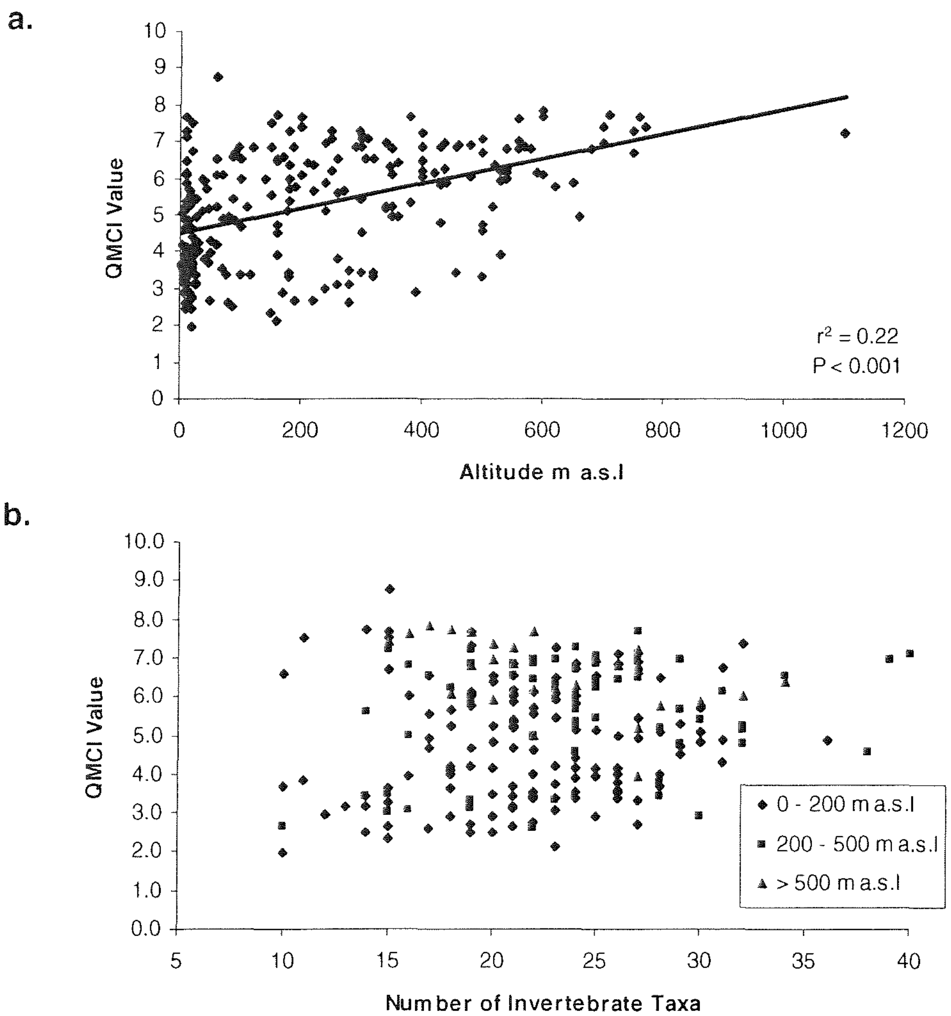


Fig. 1 Correlations between QMCI values and (a) altitude, and (b) number of invertebrate taxa. Altitude band graduations are shown for all sites in b.

Lowest QMCI scores occurred where urban/city landuse dominated the catchment (Fig. 2). A notable exception was the Styx River at Styx Mill Reserve (Site # 35), which had a QMCI of 5.5 and taxon richness of 17. This was 1.9 QMCI units higher than the next best rated stream within an urban/city catchment. Streams with agriculture as their main catchment landuse had a large spread of data points indicating that their "health" was highly variable. Diverse QMCI scores also occurred in streams with tussock catchments, whereas those in primarily forested catchments generally had higher QMCI values. However, some forested streams had low QMCI values, notably Makerikeri (Site # 40) and Stony Creek (Site # 39) whose QMCI values were less than three (Fig. 2). Gunns Bush Stream (Site # 219), an indigenously forested site, had the highest taxon richness.

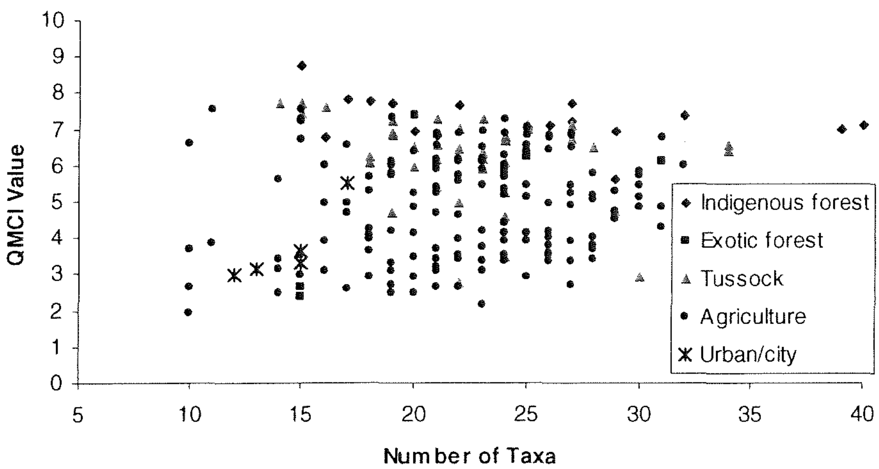


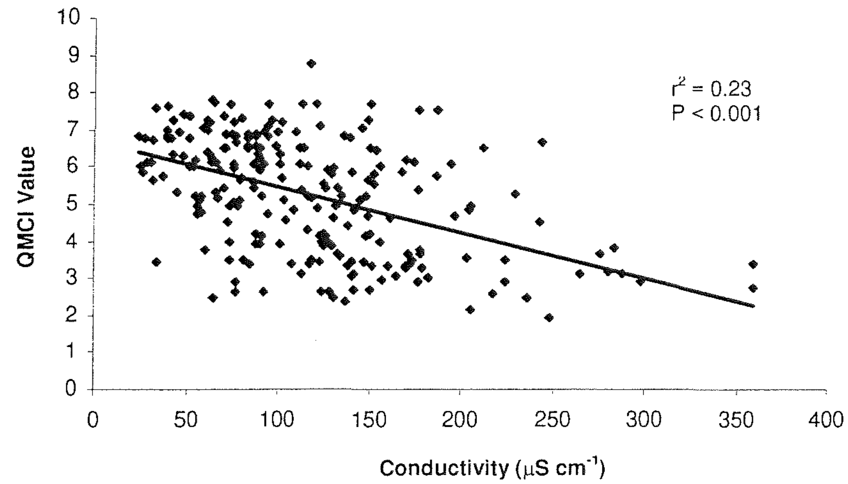
Fig. 2 Relationship between QMCI values and taxon richness. Dominant catchment landuse of all sites is shown.

Associations with environmental factors

Conductivity was the measured variable most strongly correlated with QMCI ($r = -0.49$, $P<0.001$, Fig. 3a), high conductivity being associated with low QMCI values. However, neither clarity nor pH was significantly correlated with QMCI. Temperature was significantly and negatively correlated with QMCI ($r = -0.30$, $P<0.001$, Fig. 3b) although some of the warmer streams had relatively high QMCI scores, and some of the cooler ones had low scores.

Of the other environmental factors, bank vegetation was most strongly correlated with QMCI (r = 0.57, P<0.001, Fig. 4a) suggesting that as the amount of bank vegetation and indigenous flora increase, the ratio of sensitive to insensitive taxa increases. At some sites, however, high vegetation scores were not associated

a.



b.

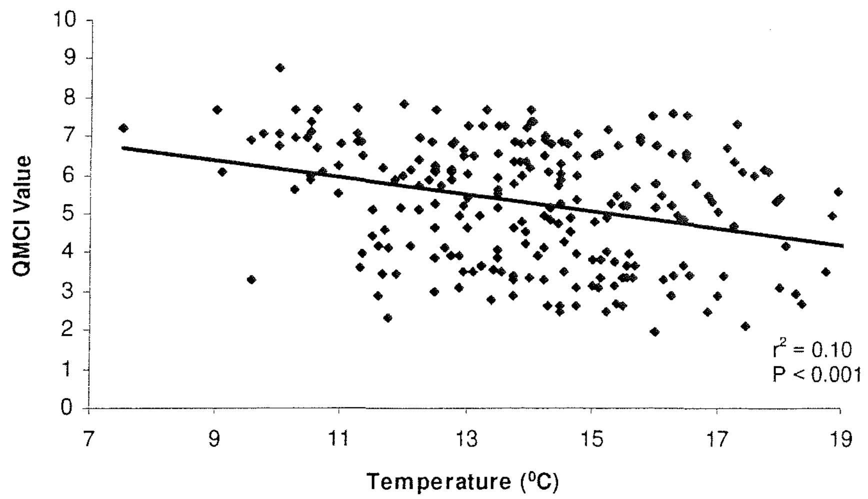


Fig. 3 Correlations between QMCI values and (a) conductivity, and (b) temperature. Conductivity and temperature values plotted are averages for the two sampling periods (November 1999 and January 2000).

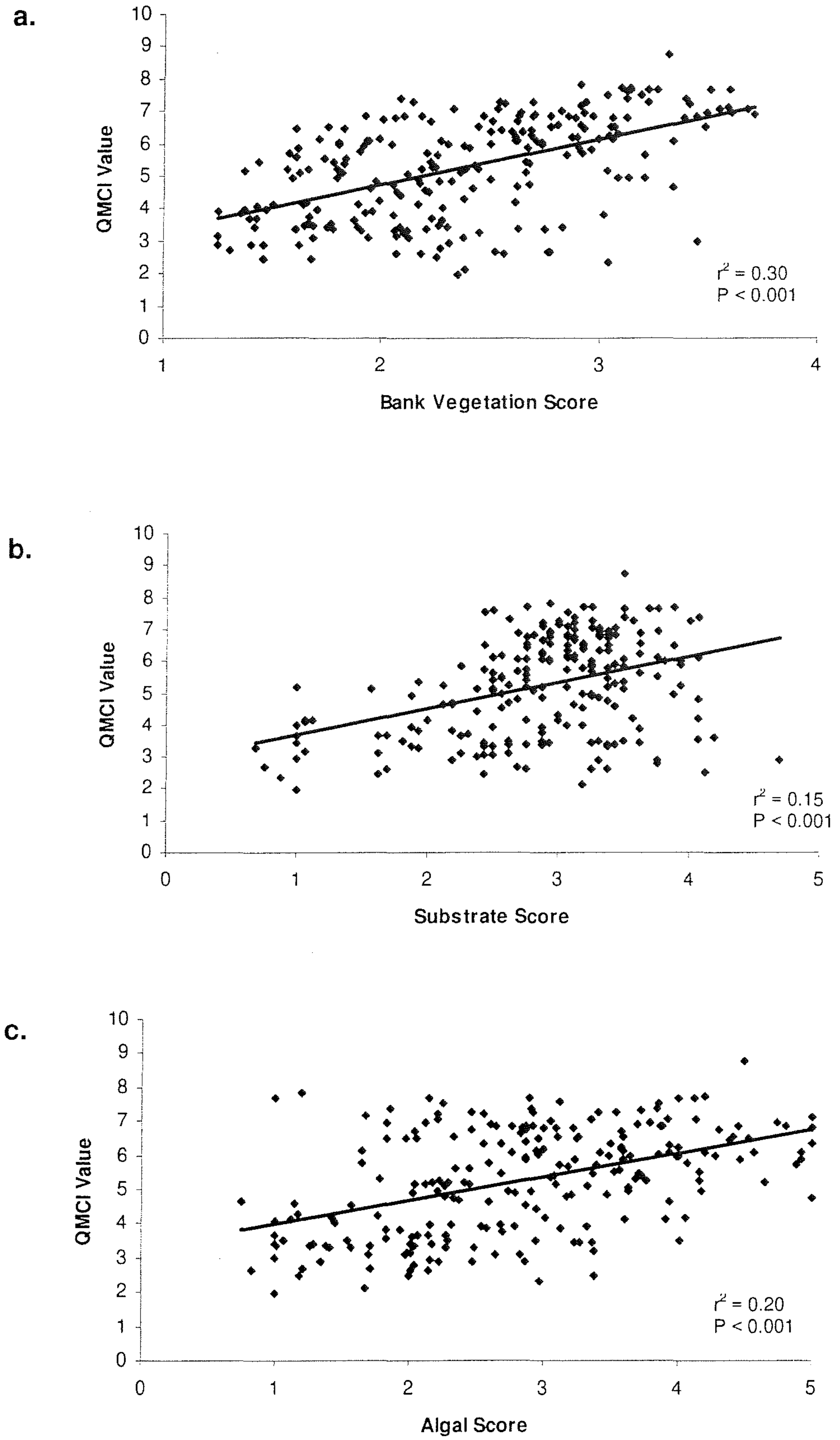


Fig. 4 Relationship between QMCI values and (a) bank vegetation scores, (b) substrate scores and (c) algal scores. Details of how scores were calculated are given in Chapter 2.

with high QMCI values. Substrate scores were significantly and positively correlated with QMCI values (r = 0.40, P<0.001, Fig. 4b), but when scores were above 2-3 (gravel, small cobbles, large cobbles, boulders) a wide scatter of points was found. Algal scores were also correlated significantly with QMCI values (r = 0.45, P<0.001, Fig. 4c) as was habitat assessment (r = 0.53, P<0.001, Fig. 5). Two notable algal score outliers were Kowai River (Site # 52) and Sudden Valley Stream (Site # 12) which had low algal scores and high QMCI values. This is possibly an artifact of the algal scoring system, which gives low scores to sites with no algae present. These two high country streams occur in locations with propensities for high flow disturbance, which could have scoured algae off the substrate, thus giving low algal scores.

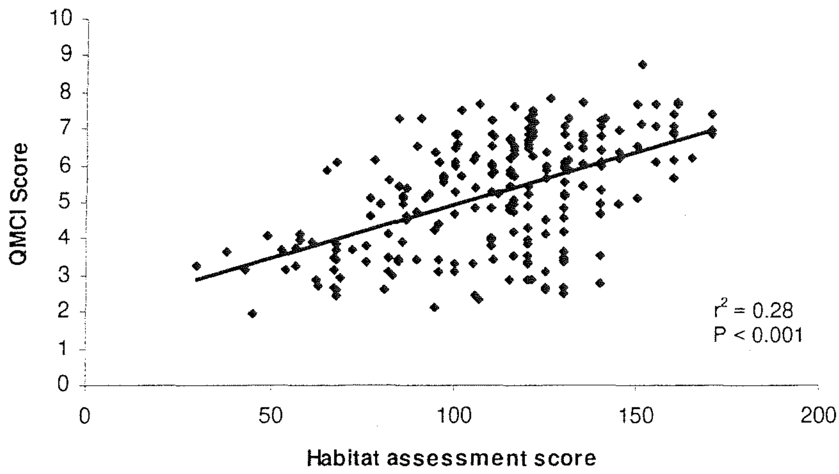


Fig. 5 Relationship between QMCI Values and habitat assessment scores. The calculation of habitat assessment scores is explained in Chapter 2.

Semi-Quantitative Macroinvertebrate Community Index

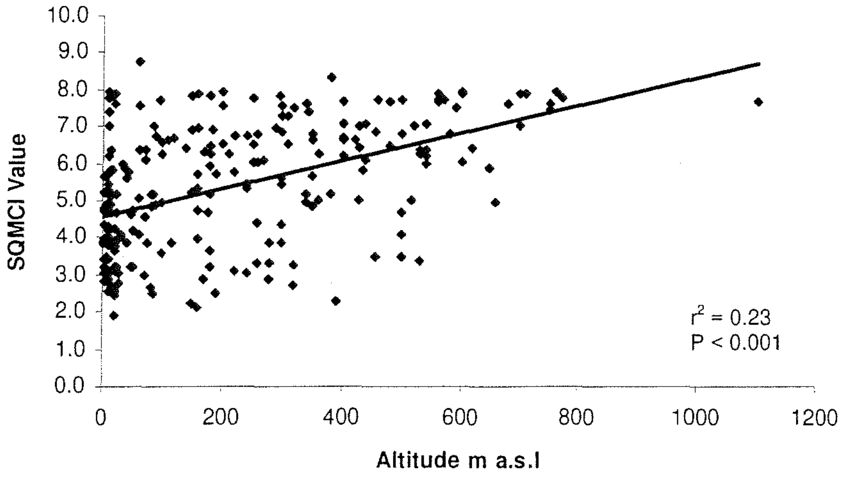
The SQMCI is similar to the QMCI except that coded abundances are substituted for actual abundances. The SQMCI was designed to give an indication of stream health similar to QMCI, but with reduced sample processing effort (Stark 1998).

The SQMCI frequency distribution for all samples is shown in Chapter 4, Figure 4.7. Values ranged from 1.9 to 8.7, with a median of 5.4 and an average of 5.4. The majority of "clean" sites were again dominated by *Deleatidium*, whereas *Potomopyrgus* dominated the sites classified as "severely degraded". The sites with the highest SQMCI values were Waiangarara (SQMCI = 7.9, Site # 163), Maori Stream (SQMCI = 8.3, Site # 4) and Ohau Stream (SQMCI = 8.7, Site # 158). The most severely degraded sites were the Upper Heathcote River (SQMCI = 1.9, Site # 141), Elephant Hill Stream (SQMCI = 2.1, Site # 275) and Stony Creek (SQMCI = 2.3, Site # 39). The high scoring sites were generally at higher altitudes and in catchments dominated by indigenous forest. The low scoring sites were generally at lower altitudes, and in agriculturally dominated catchments.

As with the MCI and QMCI, higher SQMCI values generally occurred further inland and lower values were found close to populated areas. Thus, a significant positive correlation was found between SQMCI and altitude (r = 0.49, P<0.001, Figs. 6a and 6b). Nevertheless, high SQMCI values were found in streams at all altitudes.

Forest-dominated catchments generally had the highest SQMCI values (Fig. 7), and the two forested sites with unusually low SQMCI scores are the same ones that had low QMCI scores. Sites in primarily tussock and agricultural catchments had a wide range of SQMCI values. Urban/city catchment sites had low SQMCI values and low taxon richness (Fig. 7), with the exception of the Styx River at Styx Mill Reserve.

a.



b.

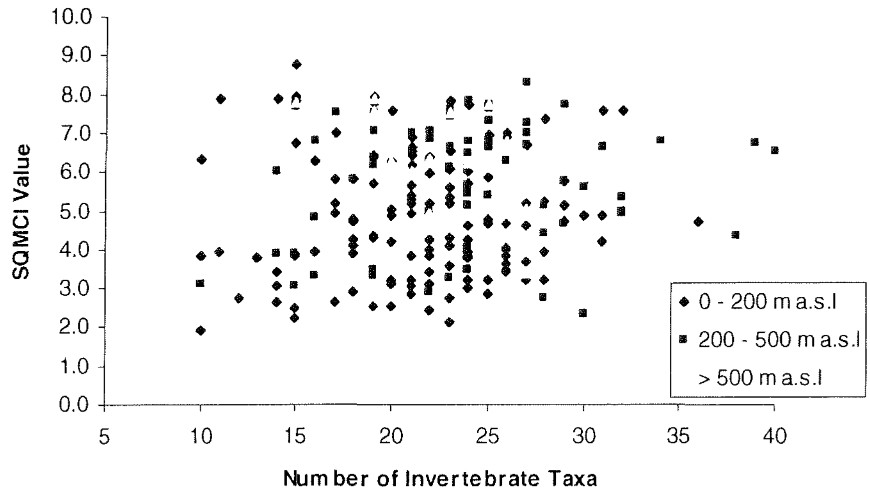


Fig. 6 Correlations between SQMCI values, and (a) altitude, and (b) number of taxa. Altitude band affiliations are shown for all sites.

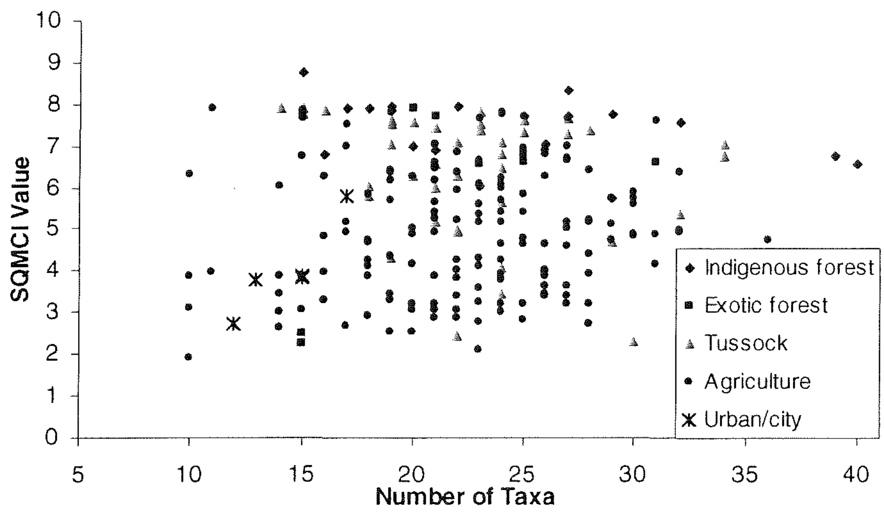


Fig. 7 Relationship between SQMCI values and taxon richness. Dominant catchment landuse is shown for all sites.

Associations with environmental factors

SQMCI scores were most strongly correlated with conductivity ($r = -0.49$, $P<0.001$, Fig. 7a), which was highest at sites with low scores. Neither clarity, nor pH were significantly correlated with SQMCI ($P<0.001$), but an increase in temperature was associated with a decrease in SQMCI ($r = -0.28$, $P<0.001$, Fig. 7b).

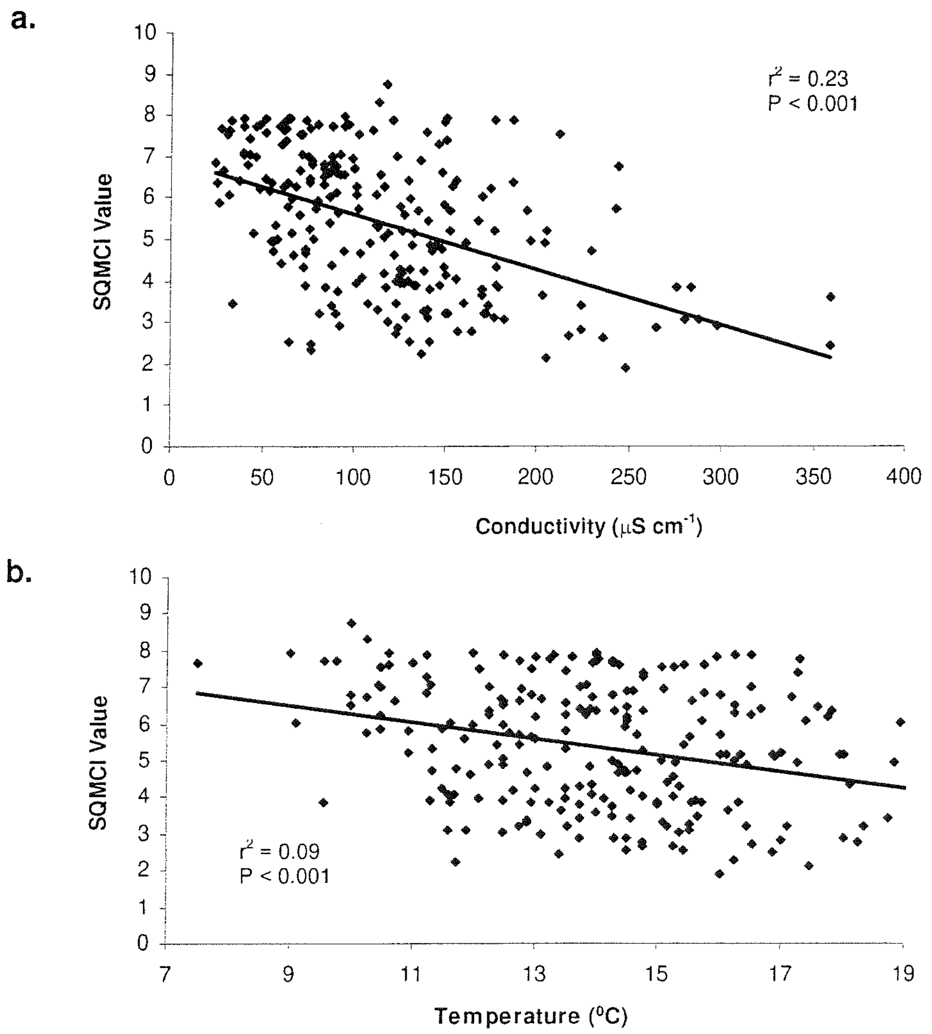
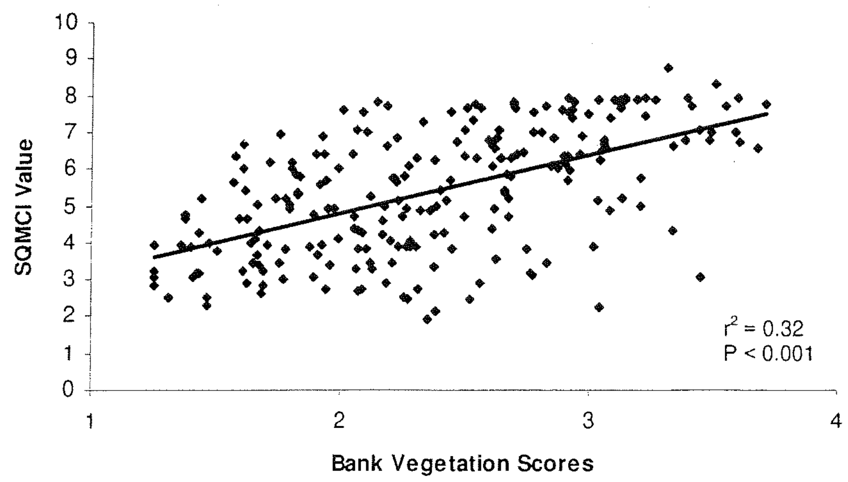


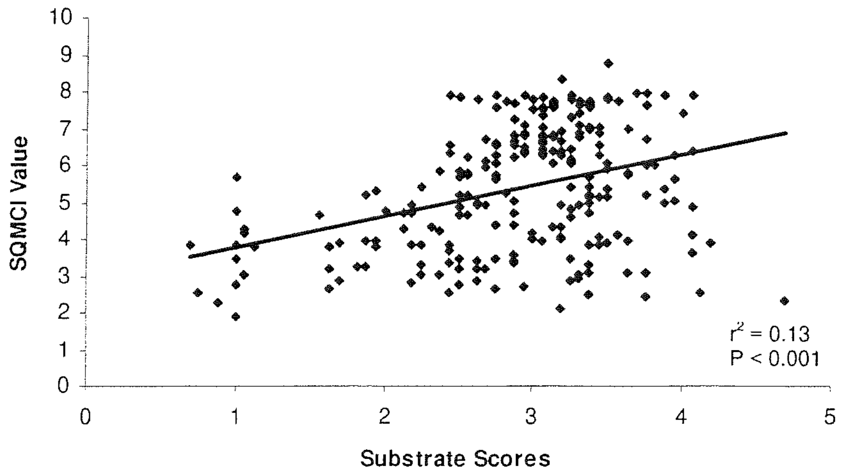
Fig. 7 Correlations between SQMCI values and (a) conductivity ($\mu S\ cm^{-1}$), and (b) temperature (^{0}C). Conductivity and temperature values are averages for the two sampling periods (November 1999 and January 2000).

Bank vegetation scores were strongly and positively correlated with SQMCI scores ($r = 0.58$, $P<0.001$, Fig. 8a) as were substrate scores ($r = 0.37$, $P<0.001$, Fig. 8b). However, when substrate score exceeded about 2.5, a wide scatter of SQMCI values was found. Algal scores were also correlated positively with SQMCI values ($r = 0.41$, $P<0.001$, Fig. 3.23c) although "clean" sites were observed across the whole range of algal scores. The two outliers that had low algal scores but high SQMCI values were again the Kowai River (SQMCI = 7.9, Site # 51) and Sudden Valley Stream (SQMCI = 7.9, Site # 12).

a.



b.



c.

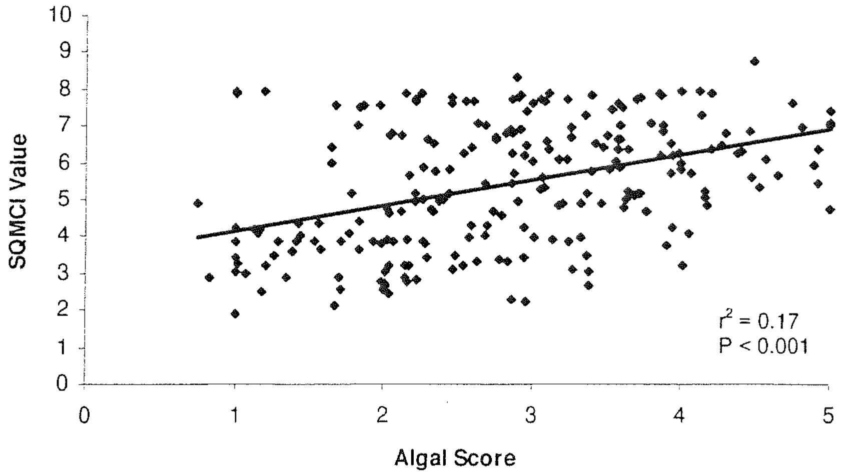


Fig. 8 Relationships between SQMCI values and (a) bank vegetation scores, (b) substrate scores, and (c) algal scores. See Chapter 2 for details of how scores were calculated.

Lastly, habitat assessment was significantly correlated with SQMCI (r = 0.51, P<0.001, Fig. 9), although as with substrate scores a wide scatter of SQMCI values can be seen on Figure 8 above a habitat score of about 100.

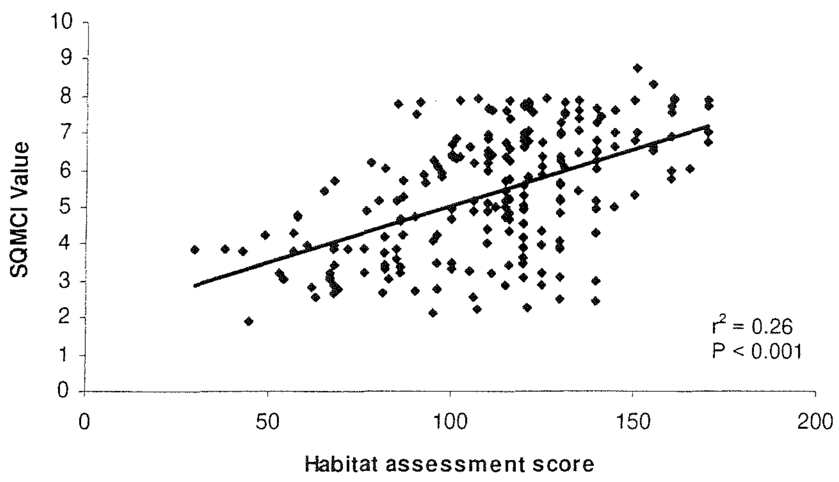


Fig. 8 Relationship between SQMCI and habitat assessment score. Habitat assessment calculation is explained in Chapter 2.