# Benthic communities of the Whanganui River catchment: the effects of land use and geology

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# ABSTRACT

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The response of macroinvertebrate communities and freshwater mussels to variation associated with land use and geology was investigated in small headwaters of the Whanganui River catchment in the North Island of New Zealand. Conductivity and water clarity were higher in streams with soft tertiary/quaternary sedimentary geology, independent of land use. These soft geology catchments, when forested, had distinctive community structure often involving high relative abundances of Ephemeroptera. Pastoral agriculture resulted in a lower diversity and abundance of pollution sensitive taxa. Impacts of pastoral land use were accentuated by soft sedimentary geology culminating in low diversity and abundance in pasture streams with soft geology. Multivariate analysis showed that community structure varied significantly depending on the combination of land use and geology type present, with land use as the most significant factor. However, variations in geology may mask the effects of land use between catchments.

In the main Whanganui River, taxonomic diversity and numbers of pollution sensitive taxa decreased downstream. This was correlated with reduced periphyton biomass and increased suspensoids. Estimates of macroinvertebrate community structure differed between artificial substrate and kick samples collected from the same sites.

Shell morphology of the freshwater mussel *Hyridella menziesi* was not correlated with water chemistry. One site near the northern boundary of the Whanganui River catchment contained mussels with distinct shell morphology. A possible explanation involving stream capture by tectonic movements is considered. Shell erosion was correlated with channel width suggesting shell erosion is greater in larger waterways. Lack of demarcation in shell growth annuli meant accurate estimates of mussel age were not possible. Poor demarcation is likely to result from non seasonal patterns of environmental variation.

# Chapter One

# General introduction

# INTRODUCTION

The character of streams and rivers reflects the integration of physical and biological processes occurring in the catchment (Johnson and Gage, 1997). Most of the Whanganui River catchment (Fig. 1.1) was once heavily forested but much of this land has been cleared for pastoral agriculture. Some of these areas have been retired allowing the native vegetation to re-grow. The land is young in geological terms consisting predominantly of Pliocene-Pleistocene marine sediments or volcanics which are younger than most of the North Island. Various unconsolidated rock types combined with low gradient provide suitable soft bottom habitats for the New Zealand freshwater mussel *Hyridella menziesi*. Many ecological features of this species are well documented in lakes but little is known of the populations present in the Whanganui River, or other lotic habitats.

The landscape influences it's water bodies through multiple pathways and mechanisms, operating at different spatial scales (Allan and Johnson, 1997). Large scale factors influencing stream ecosystem function such as geology and land use (Winterbourn, 1986) have an important influence on the physicochemical characteristics of streams (Richards *et al.*, 1997). Macroinvertebrate community structure is closely linked to these characteristics making them appropriate indicators for investigating environmental change (Stark, 1993). Changes in land use practices such as those related to agricultural development are predicted to alter habitat characteristics, invertebrate community composition and water quality (Lenat, 1984; Corkum, 1990; Quinn and Hickey, 1990b). Catchment geology also affects the structure of aquatic ecosystems over a range of spatial scales through its influence on water chemistry (Maasdam and Smith, 1994), catchment vegetation, flow variability and substrate characteristics (Cummins *et al.*, 1984; Richards *et al.*, 1997). Catchments with a combination of pastoral agriculture and soft sedimentary geology can have distinctive community structure such as low abundance and diversity (Quinn and Hickey, 1990a; Davies-Colley and Stroud, 1995). The substrate of many streams in the Whanganui River catchment are dominated by erodable Tertiary aged sedimentary rock. Little is known about the basic ecology of invertebrate communities in these mudstone streams, and the influence of land use upon them.

*Hyridella menziesi* is abundant throughout many lakes in both the North Island and South Island. However, little is known about this species in the Whanganui River catchment. Anecdotal evidence concerning the distribution and abundance of *Hyridella menziesi* in the Whanganui River catchment suggest that the species has declined since the turn of the century. Clearly, more information concerning their response to environmental variation will be beneficial to understanding reasons for their decline.

Increased knowledge of ecological conditions that occur in the main river and its tributaries will provide greater understanding of environmental and ecological characteristics of the Whanganui River. Chapter Two investigates the effects of pastoral land use and geology type on macroinvertebrate communities in headwater streams, in particular those from catchments with soft sedimentary geology. Longitudinal changes in macroinvertebrate communities along the main Whanganui River in response to tributary inputs are explored in Chapter Three. The use of both artificial substrate and kick sampling were used to assess these changes and their respective abilities to do so were compared. In Chapter Four, morphological and spatial characteristics of the New Zealand freshwater mussel *Hyridella menziesi* are investigated in relation to physicochemistry and habitat from six sites in the Whanganui river and its tributaries.

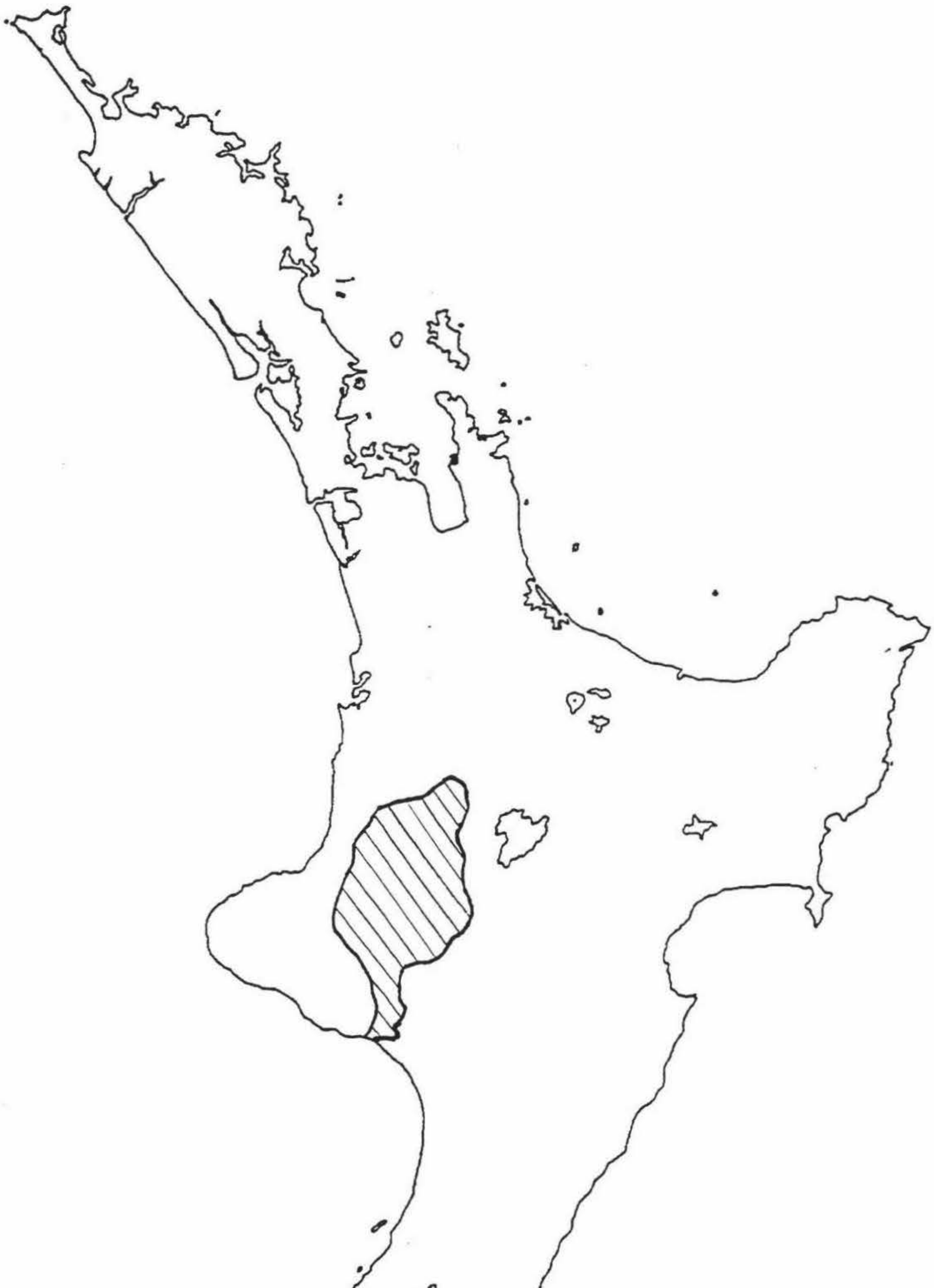

Figure 1.1. Map showing the position of the Whanganui River catchment in the North Island of New Zealand.

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# Chapter Two

# Effects of land use and geology on lotic macroinvertebrate communities in the Whanganui River

## ABSTRACT

Benthic macroinvertebrate communities were sampled during 1996 and 1997 in streams from the Whanganui River catchment varying in geology and land use. Mudstone streams had higher conductivity and lower water clarity, independent of land use. The mayfly species *Coloburiscus humeralis, Zephlebia dentata* and *Deleatidium* spp. as well as *Pycnocentria funerea*, a caddisfly were characteristically abundant in forested streams with soft sedimentary geology. Pastoral agriculture resulted in lower diversity and abundance of pollution sensitive taxa. Impacts of pastoral land use were accentuated by soft sedimentary geology culminating in low diversity and abundance in mudstone pasture streams, with greater dominance of the gastropod *Potamopyrgus antipodarium*. Multivariate analysis showed that community structure varied significantly depending on the combination of land use and geology type present.

**Keywords**: benthic macroinvertebrate communities, mudstone, geology, pastoral agriculture, native forest, water clarity, relative abundance, diversity.

## INTRODUCTION

Human alteration of landscapes has widespread implications for the well being of aquatic ecosystems (Naiman *et al*., 1995). Land use such as pastoral farming, inevitably has a major effect on the physicochemical characteristics of streams (Richards *et al.*, 1997). As well as land use characteristics, the dominant surficial geology type can account for a large proportion of the variation in macroinvertebrate distribution patterns (Richards *et al.*, 1996).

Changes in land use practices such as those related to agricultural development are predicted to alter habitat characteristics, invertebrate community composition and water quality (Lenat, 1984; Corkum, 1990; Quinn and Hickey, 1990a). Changes in invertebrate community structure in response to land development has been well documented in both New Zealand (e.g., Allan, 1959; Winterbourn, 1986; Quinn and Hickey, 1990b; Smith *et al.*, 1993; Quinn *et al*., 1997; Townsend *et al.*, 1997) and overseas (Dance and Hynes, 1980; Reed *et al.*, 1994; Richard and Host, 1994; Ventura and Harper, 1996).

Changes in community structure resulting from agricultural development include decreases in diversity and taxonomic richness , increases in invertebrate biomass, and reductions in the number of Ephemeroptera, Trichoptera and Plecoptera. A dominant feature of changing land use is the alteration or loss of riparian vegetation which then results in increases in light, nutrient and sediment levels, and alterations of flow and temperature regimes (Cowie, 1985; Winterbourn, 1986; Dons, 1987; Fahey and Rowe, 1992; Williamson *et al.*, 1992; Prat and Ward, 1994; Quinn *et al.,* 1994). Catchment vegetation types such as pine and native forest often have higher amounts of particulate organic matter than that of pasture (e.g., Friberg *et al.*, 1997). Increases in CPOM can alter community energetics and subsequently invertebrate functional feeding groups allowing for greater abundance of facultative shredders (e.g., Harding and Winterbourn, 1995).

The underlying geology of a catchment plays a key role in structuring aquatic ecosystems at both macro and micro habitat level through its influence on water quality, catchment vegetation, flow variability and substrate characteristics. Geology may also have a major influence on aquatic ecosystems through the regulation of dissolved ion input (Cummins *et al.,* 1984; Lay and Ward, 1987; Close and Davies-Colley, 1990). Areas containing volcanic rocks and tephra in New Zealand often have higher phosphorus concentrations (Currie and Gilliland, 1980; Timperley, 1983), granite and limestone can increase nitrate levels (Lay and Ward, 1987), and bicarbonate ions created from the dissolution of limestone bedrock can increase alkalinity (Stumn and Morgan, 1981). Geology may also influence the composition of catchment vegetation and the rate of organic matter turn-over, both of which can influence stream ecosystem function (Miller, 1968; Lay and Ward, 1987). Large scale geological characteristics also influence stream hydrology and thermal characteristics (Wiley *et al.,* 1997). For example streams supplied predominantly by surficial runoff rather than ground water have more variable flow and temperature regimes (Haro and Wiley, 1992; Richards *et al.*, 1997). Flow variation may affect aquatic communities directly (Statzner *et al.,* 1988; Quinn and Hickey, 1990b; Poff and Allan, 1995), or indirectly in combination with gradient, bed stability and the size distribution of substrate (Brussven, 1984; Winterbourn, 1986; Jowett and Richardson, 1990).

It is evident from the above studies that geology and land use both play important roles in shaping invertebrate community structure. Despite this, the focus of research concerning the effects of land use on water quality and invertebrate community structure in New Zealand (Quinn *et al.,* 1992; Harding and Winterbourn, 1995) has been focused primarily in cobble bottom streams with hard geology. The substrate of many streams in the Whanganui River catchment and others in the central North Island are dominated by erodable Tertiary aged sedimentary rock. Little is known about the basic ecology of invertebrate communities in these mudstone streams. These types of catchments however are among those most highly influenced by agricultural activity (Maasdam and Smith, 1994). To successfully manage these streams it is necessary to know how impacts of land use on water quality and invertebrate community structure may differ with catchment geology. In this study I survey invertebrate communities and associated measures of water quality in 48 streams in the Whanganui River catchment that exhibit different geological characteristics, and compare the influence of land use on those communities.

## STUDY SITES

All study sites were third to fifth order streams draining into the Whanganui River in the central North Island of New Zealand. Tertiary sandstone, mudstone, and siltstone are the most common geology types in the catchment. Streams around Mount Ruapehu flow through bedrock of volcanic origin composed of Andesites. In the northern catchment, streams flow through Jurassic sandstone and siltstone with Ignimbrite substrates in many headwaters. Near the city of Wanganui, geology is dominated by less stable Quaternary sands and silts. Sites were chosen based on land use and surface geology. Streams were therefore grouped as having either harder Quaternary Andesite and Jurassic substrates or softer mudstone substrates of Tertiary/Quaternary age, and land use types as predominantly pasture or native vegetation. Most of the agricultural activity consisted of lower intensity sheep/beef grazing. Native vegetation consisted of mixed broadleaf/podocarp forest with areas of beech and native grassland in some of the catchments situated on the volcanic plateau of Mount Ruapehu. Aborted attempts to farm many marginal areas before and during the 1940's meant that some of the lower altitude forests are of relatively young age. Stream geology was determined using a combination of topographical maps, geological maps, and the substrate type found when sampling. Land use/geology classes will be referred to as mudstone pasture or forest for substrates formed by Tertiary/Quaternary sediments and hardstone pasture or forest for Andesitic/Jurassic substrates. Study sites ranged in altitude between < 20 m and 840 m a.s.l.

## METHODS

*Sampling protocol*

Sampling was conducted on two occasions. Between March 25 and April 28, 1996, 33 sites differing in geology and land use (Fig. 2.1) and February, 1997, 30 sites all in catchments of sand/silt/mudstone geology's were sampled (Fig. 2.2). Fifteen of these sites had also been sampled during the 1996 sampling period.

On both occasions sampling of macroinvertebrates was conducted by collecting four replicate 250 μm mesh, $0.11m^2$ surber samples which were preserved in 70% alcohol. Samples were sieved through a 250 μm Endecott sieve and animals identified and enumerated with a 40× microscope using available keys (McFarlane, 1951; Winterbourn, 1973; Chapman and Lewis, 1976; Cowley, 1978; Winterbourn and Gregson, 1989; Towns and Peters, 1996). If taxa could not be identified to species they were assigned to apparent morphospecies.

Concurrent with each invertebrate collection a range of physicochemical parameters were measured (Table 2.1 and 2.2). Conductivity and temperature were measured using an Orion 122 conductivity meter and pH with an Orion 250A pH meter. A black disk was used to measure water clarity (Davies-Colley, 1988). Velocity was measured using a Kempton A.OTT current meter. Between 3 and 10 measures of depth and velocity were made in a transect at a right angle across the stream. Large rivers or those with greater channel variability required more measurements. Substrate size composition was assessed visually as a proportion and converted to a substrate index following Jowett *et al.* (1991). Stability was assessed using the bottom component of the Pfankuch index (Pfankuch, 1975). Channel slope was determined as the distance between three 20 m contours on a 1:50 000 topographical map and converted to degrees. The percentage of forest cover was calculated by tracing onto acetate sheets the catchment from topographical maps. Areas of forest were then cut, weighed and converting to a proportion of the total weight.

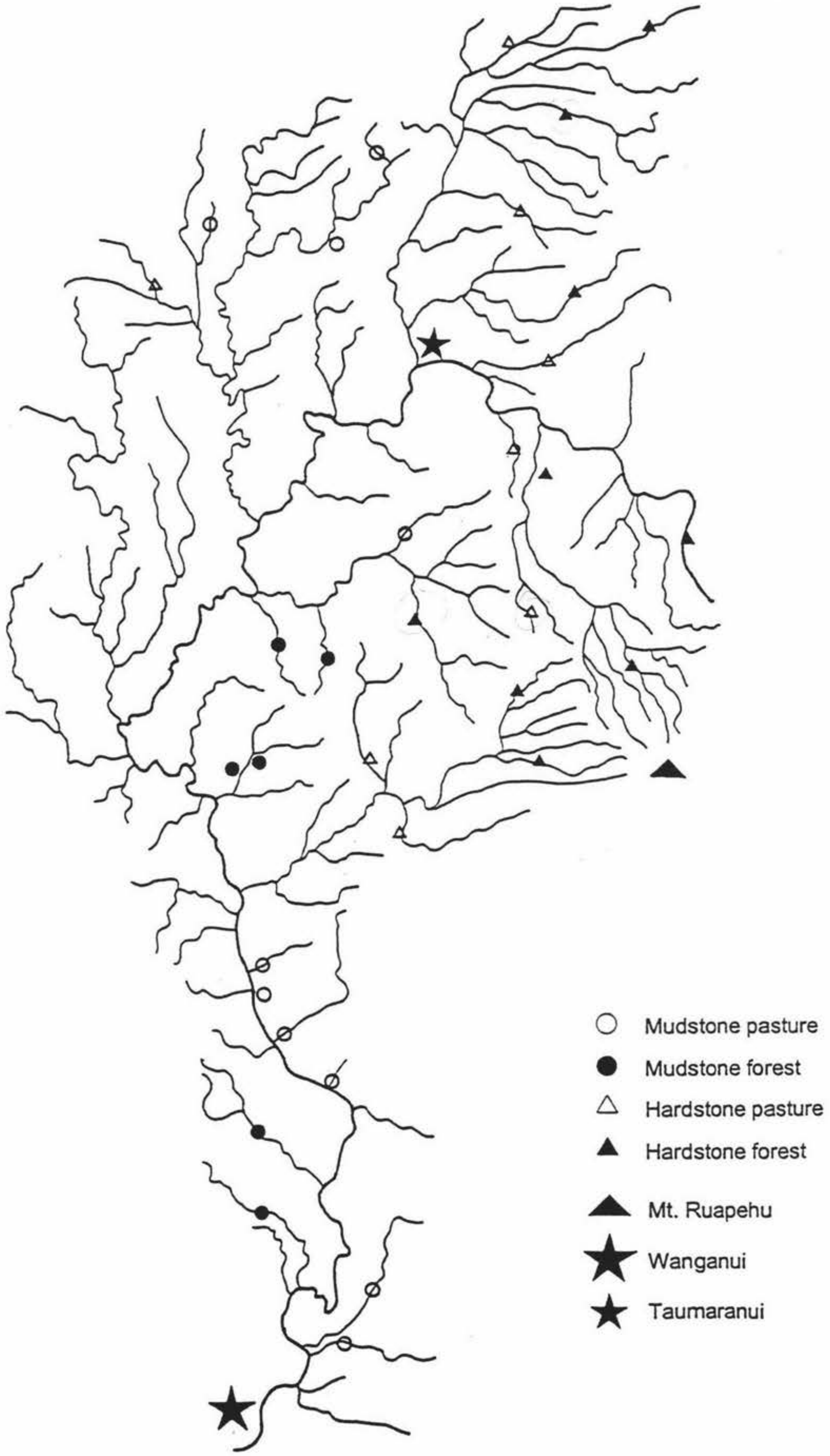


Figure 2.1. Location of the 33 streams sampled between March 25 and April 28, 1996, showing the geology/land use classes and major population centres.

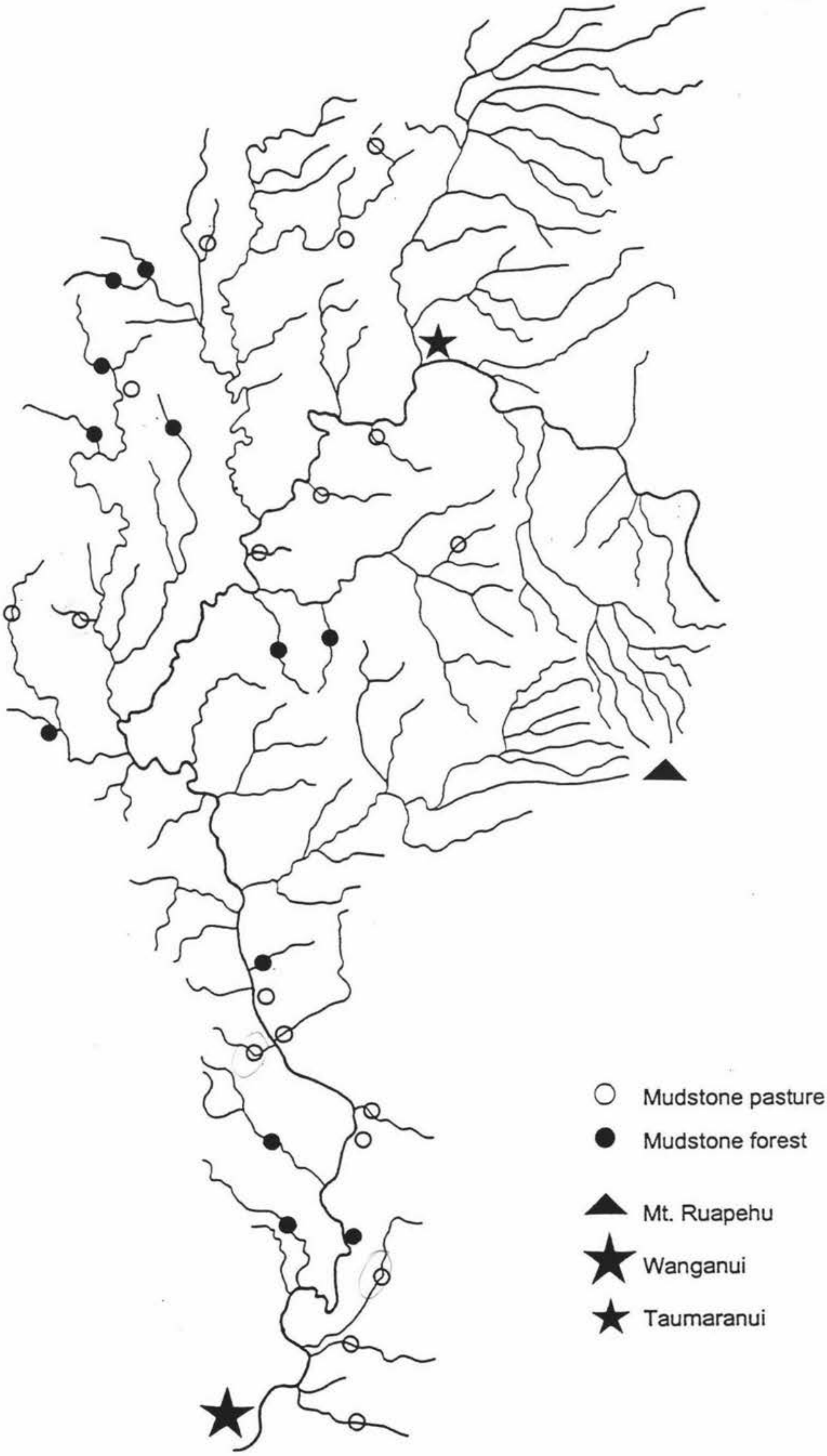


Figure 2.2. Location of the 30 mudstone streams sampled February, 1997, showing the land use classes and major population centres.

Table 2.1. Physical and chemical characteristics recorded from 33 sterams in the Whanganui River catchment, between March and April, 1996, which have mudstone or hardstone substrates and drain pasture or indigenous forest.

| Sites | Width (m) | Mean depth (cm) | Mean velocity (cm $s^{-1}$) | Slope (degrees) | Stability | Conductivity ($\mu Scm^{-2}$) | Black disk (m) | pH | Substrate size index | Altitude (metres) | Map reference 1:50 000 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *Mudstone Pasture* | | | | | | | | | | | |
| Kawautahi | 6.3 | 20 | 52 | 0.41 | 31 | 93.3 | 0.53 | 6.87 | 0.049 | 240 | S19 055390 |
| Makirikiri | 2 | 17 | 78 | 0.28 | 45 | 299 | 0.6 | 7.51 | 0.029 | 40 | S22 934474 |
| Motuaruhe | 7.1 | 16 | 59 | 0.37 | 31 | 200 | 0.23 | 7.52 | 0.059 | 40 | S21 956754 |
| Ohura | 5 | 34 | 34 | 0.92 | 45 | 141 | 0.8 | 7.19 | 0.029 | 240 | S18 028770 |
| Otahu | 2.5 | 12 | 26 | 0.37 | 40 | 81 | 1.6 | 7.03 | 0.040 | 180 | S18 979688 |
| Upokongaru | 3 | 72 | 51 | 0.18 | 50 | 393 | 0.35 | 7.65 | 0.029 | 40 | S22 935473 |
| Hapurua | 3.1 | 43 | 45 | 0.28 | 35 | 83 | 0.57 | 7.01 | 0.033 | 180 | R18 833670 |
| Whatauma | 7.1 | 50 | 69 | 1.83 | 28 | 255 | 0.11 | 7.6 | 0.045 | 40 | S21 920780 |
| Kaukore | 6.4 | 17 | 60 | 1.31 | 28 | 154 | 0.7 | n.a. | 0.060 | 60 | R21 862898 |
| Waimarino.Pipiriki | 1.8 | 21 | 72 | 9.09 | 30 | 136 | 0.17 | 7.3 | 0.050 | 60 | R21 864861 |
| *Mudstone Forest* | | | | | | | | | | | |
| Crocadile.Creek | 4.3 | 22 | 56 | 1.83 | 26 | 95 | 1.4 | 7.78 | 0.061 | 180 | R20 876130 |
| Kaiwhakauka | 18.6 | 35 | 52 | 1.22 | 20 | 92 | 1.9 | 7.02 | 0.061 | 140 | R19 899271 |
| Morinui | 11.2 | 25 | 53 | 0.61 | 22 | 115 | 1.7 | 6.95 | 0.062 | 200 | S19 953260 |
| Ahuahu | 2.3 | 54 | 40 | 0.46 | 45 | 147 | 1.2 | 7.48 | 0.035 | 120 | R21 860703 |
| Kaurapaoa | 4.3 | 37 | 71 | 0.2 | 28 | 211 | 1.05 | 7.47 | 0.056 | 80 | S21 893606 |
| Whirinaki | 2.7 | 14 | 55 | 2.62 | 27 | 102 | 1.7 | 7.2 | 0.042 | 140 | R20 852115 |
| *Hardstone Pasture* | | | | | | | | | | | |
| Kaitieke | 10.5 | 32 | 71 | 0.61 | 19 | 114 | 0.85 | 7.01 | 0.058 | 180 | S19 060131 |
| Kakahi | 6 | 15 | 68 | 1.22 | 22 | 86 | 1.65 | 7.3 | 0.058 | 280 | S19 016549 |
| Mangakara | 3 | 64 | 79 | 3.66 | 34 | 94 | 1.95 | 7.23 | 0.049 | 180 | R18 769658 |
| Mangakahu | 7.1 | 40 | 78 | 0.46 | 24 | 85 | 1.25 | 7.18 | 0.056 | 200 | S18 134720 |
| Orautohu | 11.1 | 36 | 87 | 0.92 | 25 | 87 | 1.35 | 7.68 | 0.053 | 310 | S20 025054 |
| Pungapunga | 10.4 | 15 | 96 | 0.41 | 28 | 90 | 2.45 | 6.98 | 0.048 | 240 | S18 200552 |
| Ruatiti | 13.1 | 54 | 81 | 0.23 | 22 | 78 | 1.1 | 7.15 | 0.057 | 240 | S20 999080 |
| Waimiha | 19.5 | 61 | 90 | 0.23 | 27 | 60 | 2.2 | 6.93 | 0.047 | 140 | S17 152869 |
| *Hardstone Forest* | | | | | | | | | | | |
| Makatote | 16.4 | 35 | 68 | 1.83 | 18 | 74 | 7.3 | 6.86 | 0.065 | 740 | S20 168128 |
| Maramataha | 8.8 | 24 | 47 | 0.37 | 26 | 52 | 1.7 | 6.8 | 0.049 | 240 | S17 528266 |
| Pumice.Quarry | 2.7 | 43 | 46 | 4.57 | 20 | 45 | 3.2 | 7.01 | 0.048 | 380 | S19 180420 |
| Upper.Retaruke | 22.7 | 21 | 55 | 0.37 | 22 | 107 | 2.15 | 6.75 | 0.061 | 200 | S19 440320 |
| Ongarue | 11.4 | 36 | 62 | 0.61 | 30 | 55 | 3.25 | 6.8 | 0.050 | 340 | S17 245885 |
| Taringamotu | 7.2 | 26 | 95 | 0.76 | 27 | 73 | 2.5 | 6.93 | 0.052 | 340 | S18 224616 |
| Waimarino.Ruapehu | 10.1 | 36 | 31 | 0.73 | 23 | 55 | 3.7 | 7.05 | 0.050 | 740 | S20 173180 |
| Whakapapanui | 23.8 | 50 | 118 | 1.73 | 32 | 120 | 9.7 | 7.24 | 0.052 | 830 | S19 270254 |
| Whanganui | 7.5 | 63 | 56 | 1.83 | 26 | 94 | 8.3 | 7.71 | 0.050 | 680 | T19 337353 |

Table 2.2. Physical and chemical characteristics recorded from 30 streams in the Whanganui River catchment, between the 14 and 26 February, 1997, which have mudstone substrates and drain pasture or indigenous forest.

| Site | Width (m) | Mean depth (cm) | Mean velocity (cm $s^{-1}$) | Slope (degrees) | Stability | Conductivity ($\mu Scm^{-2}$) | Black disk (m)) | pH | Substrate size index | Chorophyll *a* ($\mu g$ $cm^{-2}$) | Altitude (m) | Map reference 1:50 000 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Pasture* | | | | | | | | | | | | |
| Ohura | 2.3 | 14.0 | 129.3 | 0.92 | 45 | 235 | 1.1 | 7.4 | 0.029 | 0.00035 | 240 | S18 028770 |
| Matarawa | 1.9 | 0.3 | 69.0 | 0.25 | 43 | 610 | 0.7 | 7.3 | 0.040 | 0.00530 | 10 | S22 923 393 |
| Makirikiri | 2.0 | 17.7 | 106.7 | 0.28 | 45 | 213 | 1.3 | 7.8 | 0.029 | 0.00048 | 40 | S22 934474 |
| Whangamomona | 3.6 | 0.3 | 198.7 | 0.10 | 33 | 124 | 0.3 | 7.5 | 0.044 | 0.00004 | 160 | R19 608207 |
| Te Maire | 8.1 | 9.3 | 116.3 | 0.72 | 29 | 269 | 0.5 | 7.3 | 0.349 | 0.00150 | 155 | S19 998478 |
| Operiki | 2.4 | 12.3 | 95.0 | 1.99 | 21 | 450 | 2.3 | 7.2 | 0.061 | 0.00081 | 90 | S21 971 696 |
| Kokakonui | 4.4 | 11.7 | 98.3 | 0.97 | 32 | 289 | 1.3 | 7.5 | 0.060 | 0.00201 | 150 | S19 958416 |
| Motuaruhe | 7.1 | 16.3 | 133.8 | 0.37 | 31 | 303 | 2.4 | 7.3 | 0.059 | 0.00195 | 40 | S21 956754 |
| Otahu | 2.5 | 18.3 | 61.0 | 0.37 | 40 | 109 | 1.9 | 7.1 | 0.040 | 0.00376 | 180 | S18 979688 |
| Whataumu | 4.2 | 15.0 | 107.0 | 1.83 | 28 | 395 | 0.6 | 7.5 | 0.045 | 0.00023 | 40 | S21 920780 |
| Waimarino | 2.1 | 10.3 | 58.7 | 9.09 | 30 | 269 | 0.5 | 7.4 | 0.050 | 0.00300 | 60 | R21 864861 |
| Kauwautahi | 4.8 | 18.7 | 139.0 | 0.41 | 31 | 172 | 1.5 | 7.3 | 0.050 | 0.00095 | 240 | S19 055390 |
| Tauwhata | 3.0 | 10.3 | 108.0 | 0.85 | 39 | 161 | 1.7 | 7.5 | 0.050 | 0.00185 | 140 | R19 882348 |
| Hapurua | 2.1 | 10.3 | 58.7 | 0.28 | 35 | 137 | 0.5 | 7.5 | 0.033 | 0.00143 | 180 | R18 833670 |
| Upokongaru | 3.0 | 72.0 | 61.0 | 0.18 | 50 | 261 | 1.1 | 7.2 | 0.029 | 0.00020 | 40 | S22 935473 |
| Paparata | 1.8 | 11.0 | 85.3 | 1.79 | 34 | 166 | 3.3 | 7.4 | 0.047 | 0.00136 | 200 | R19 743464 |
| Otaupare | 7.6 | 20.3 | 46.0 | 0.90 | 24 | 243 | 3.5 | 7.1 | 0.064 | 0.00108 | 50 | R21 882808 |
| *Forest* | | | | | | | | | | | | |
| Mangakara East | 3.0 | 11.3 | 115.3 | 3.66 | 34 | 104 | 3.1 | 7.2 | 0.049 | 0.00085 | 180 | R18 769658 |
| Heao | 2.9 | 9.8 | 129.8 | 0.25 | 46 | 160 | 0.8 | 7.2 | 0.032 | 0.00088 | 170 | R19 793475 |
| Kaukore | 6.4 | 16.7 | 129.3 | 1.31 | 28 | 193 | 0.3 | 7.3 | 0.060 | 0.00291 | 60 | R21 862898 |
| Ahuahu | 2.3 | 23.7 | 90.3 | 0.46 | 45 | 147 | 1.1 | 7.4 | 0.035 | 0.00014 | 120 | R21 860708 |
| Morinui | 11.2 | 27.2 | 166.8 | 0.61 | 22 | 203 | 2.1 | 7.8 | 0.062 | 0.00161 | 200 | S19 953260 |
| Karapaoa | 4.3 | 37.4 | 161.7 | 0.20 | 26 | 180 | 1.3 | 8.1 | 0.056 | 0.00306 | 80 | S21 893606 |
| Pitangi | 2.5 | 14.0 | 75.3 | 0.89 | 32 | 300 | 2.0 | 7.9 | 1.416 | 0.00119 | 20 | S22 960582 |
| Tangarakau | 14.5 | 24.0 | 183.3 | 0.34 | 41 | 153 | 1.7 | 7.1 | 0.047 | 0.00030 | 180 | R19 733470 |
| Mangare | 3.2 | 0.5 | 102.5 | 0.4 | 40 | 94 | 0.5 | 7.3 | 0.049 | 0.00003 | 160 | R19 602276 |
| Mangakara West | 6.3 | 14.8 | 157.5 | 1.15 | 30 | 76 | 2.1 | 7.2 | 0.045 | 0.00066 | 290 | R18 763664 |
| Matauru | 4.8 | 25.0 | 157.0 | 1.15 | 27 | 99 | 1.8 | 7.1 | 0.057 | 0.00055 | 155 | R19 707457 |
| Kaiwhakauka | 18.6 | 29.7 | 159.7 | 1.22 | 20 | 198 | 2.6 | 7.3 | 0.061 | 0.00088 | 140 | R19 899271 |
| Morgans | 1.3 | 6.7 | 102.7 | 2.98 | 17 | 180 | 3.3 | 8.3 | 0.079 | 0.00007 | 180 | R19 689460 |

Algal biomass was assessed (in 1997 only) by extracting photosynthetic pigments (chlorophyll *a* and phaeophytin ) in 90% acetone, for 24 hours at 5 °C, from five small cobbles (mean circumference = 10 cm) collected on each sampling occasion. Wavelength absorbtion was measured on a Jenway 6105 Spectrophotometer. Total pigment concentration was calculated using the method of Moss (1967 a, b) and corrected for stone surface area (Graham *et al.*, 1988).

*Data analysis*

Data were $\log_{10}(x+1)$ transformed prior to analysis where appropriate to remove heterogeneity of variance. Differences in physicochemical and biological measures among site classes were analysed with one-way analysis of variance (ANOVA) using the SYSTAT statistical package (SYSTAT, 1996). Correlations between the physicochemical variables and six indices of community structure were calculated. These indices were the Simpson's index of eveness (Simpson, 1949), Berger-Parker dominance index (Berger and Parker, 1970), Margelefs index (Clifford and Stevenson, 1975), EPT (Lenat, 1988), MCI (Stark, 1985) and QMCI (Stark, 1993).

Simpson's index of equability or eveness is given by:

$$D = \sum \frac{(n_i - 1)}{(N(N-1))}$$

and the Berger-Parker dominance index by: $D = N_{max}/N$ where $n_i$ = number of individuals in the *i*-th species, N = total number of individuals, S = species number and $N_{max}$ = number of individuals in the most abundant species. Indices that provided an indication of diversity were taxa richness, and Margalefs index by: D = (S-1)/1n/ N where S = species number and N = total number of individuals. EPT scores consisted of the number of Ephemeroptera, Plecoptera and Trichoptera taxa present. The MCI weights taxa according to their tolerance of organic enrichment. Taxa that are characteristic of higher water quality score more highly than those commonly found in polluted conditions (Stark, 1993). The MCI is given by:

$$MCI = \frac{site \quad score}{no. \quad of \quad scoring \quad taxa} \times 20$$

where the site score is the sum of individual taxon scores.

The QMCI (Stark, 1993) is given by:

$$QMCI = \sum_{i=1}^{i=S} \frac{(n_i \times a_i)}{N}$$

where $S$ = the total number of taxa in the sample, $n_i$ is the number of individuals in the $i$-th scoring taxon, $a_i$ is the score for the $i$-th taxon, and $N$ is the total number of individuals in the sample.

Community structure was analysed with detrended correspondence analysis (DECORANA), cluster analysis (using 2W/(A+B) and the group average), and two-way indicator species analysis (TWINSPAN) using the PC-ORD statistical package (McCune and Mefford, 1995). Pseudo-species cut levels used were 1, 10, 100 and 1000. Multi-response permutation procedure (MRPP), a non parametric discriminant analysis was carried out also using the PC-ORD statistical package to assess if there were significant differences in overall community structure between classes.

## RESULTS

Between year effects

*Physicochemical characteristics*

At sites which were re-sampled in 1997, conductivity and water clarity were the only two environmental variables to change significantly (Table 2.3) and both were higher than the previous year (Fig. 2.3). However, both variables were correlated between years (Table 2.3).

*Community structure*

Of the biological indices only the Berger-Parker Dominance index differed significantly between 1996 and 1997 (Table 2.3). Relative abundances of the major taxonomic groups did not differ statistically between years. Overall there was no significant difference in community structure between 1996 and 1997 (MRPP $r = 0.001$, $P = 0.37$). TWINSPAN (Fig. 2.4) indicated close similarities in community structure of most but not all site pairs between years.

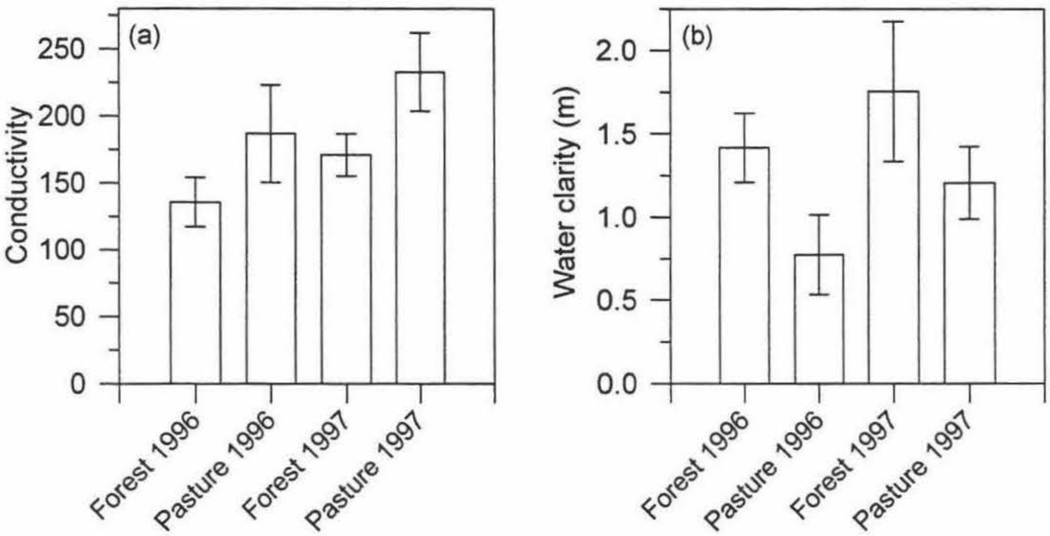


Figure 2.3. Mean (± 1 S.E.) (a) water clarity and (b) conductivity (*μScm*$^{-2}$) measured in 15 streams in the Whanganui River catchment, between March and April, 1996, and February, 1997, which have mudstone substrates and drain pasture or indigenous forest.

## Land use and geology effects 1996

### *Physicochemical characteristics*

The size of streams varied with mean depth and width ranging from 15 cm to 64 cm and 1.8 m to 23.8 m, respectively. Water clarity was highest in the hardstone forest class with low levels occurring in the mudstone pasture class (Fig. 2.5a). Conductivity displayed the reverse pattern with higher conductivity in mudstone streams (Fig. 2.5b).

Mudstone pasture streams had the lowest channel stability and smallest substrate size. Hardstone streams were wider with faster current velocity. Neither pH nor depth differed significantly among land use/geology classes (Table 2.4). Catchment forest cover was positively correlated with black disk visibility and substrate size and negatively correlated with stability and conductivity (Table 2.5).

### *Community structure*

Of the 112 taxa encountered, 104 were found across all four land use/geology classes. In mudstone pasture streams the gastropod *Potamopyrgus antipodarium* was numerically dominant and on average accounted for 45% of the individuals present (Fig. 2.6).

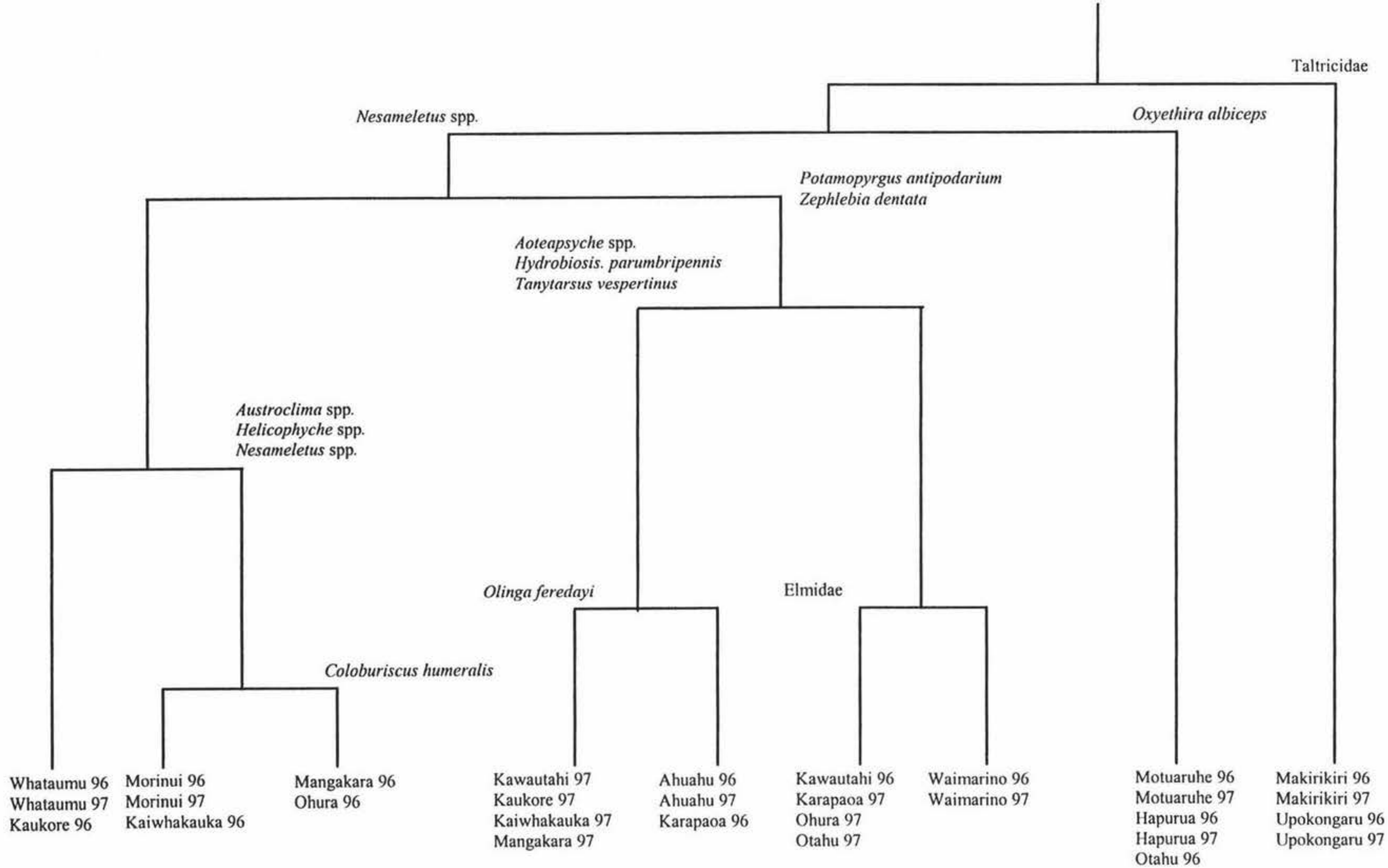


Figure 2.4. TWINSPAN classification of 15 mudstone streams sampled between March and April, 1996, and February, 1997.

Table 2.3. Paired t-tests and Pearson correlations between environmental and biotic measurements for 15 streams sampled in March to April, 1996 and February, 1997, which have mudstone substrates and drain pasture or indigenous forest.

| Variable | *t* statistic | *P* value | *r* value | *P* value |
|---|---|---|---|---|
| Conductivity | 2.95 | 0.01 | 0.67 | 0.01 |
| Velocity | -1.31 | 0.21 | 0.29 | 0.29 |
| Water clarity | 2.71 | 0.02 | 0.73 | 0.00 |
| Total abundance | 1.16 | 0.27 | 0.12 | 0.68 |
| pH | 1.19 | 0.25 | 0.14 | 0.61 |
| Taxa richness | 0.69 | 0.50 | 0.81 | 0.00 |
| Margalefs index | -0.77 | 0.46 | 0.58 | 0.02 |
| Simpson's index | -1.27 | 0.23 | 0.34 | 0.22 |
| Berger-Parker index | 2.63 | 0.02 | 0.48 | 0.07 |
| MCI | 0.29 | 0.77 | 0.38 | 0.17 |
| QMCI | 1.12 | 0.28 | 0.48 | 0.07 |
| EPT score | 0.28 | 0.78 | -0.34 | 0.22 |

Ephemeroptera dominated in the mudstone forest streams. In contrast, hardstone streams had a greater relative abundance of Trichoptera (36%) and Coleoptera (21%) than those containing soft substrate (12% and 8%, respectively). Proportions of Coleoptera and Diptera were higher and Ephemeroptera and Trichoptera lower as a result of agricultural development.

Diversity as assessed by taxonomic richness and Margalefs index was lowest in the mudstone pasture class and highest in the mudstone forest class with similar levels of diversity in the two hardstone groups (Fig. 2.7a). While abundance did not differ significantly among groups hardstone pasture streams clearly had the most individuals with similar abundance in the other three groups (2.7b). Neither Simpson's or Berger-Parker indices differed significantly among classes (Table 2.4).

MCI scores were highest in those classes with forested catchments whereas QMCI values did not differ significantly (Fig. 2.8b) (Table 2.2) although it revealed a similar trend to the MCI. The number of Ephemeroptera, Plecoptera and Trichoptera (EPT) was greatest in mudstone forested streams (Fig. 2.8a). MCI values were correlated positively with water clarity, catchment forest cover, substrate size, and negatively with conductivity, stability and pH while QMCI was correlated negatively with conductivity and positively with slope (Table 2.5).

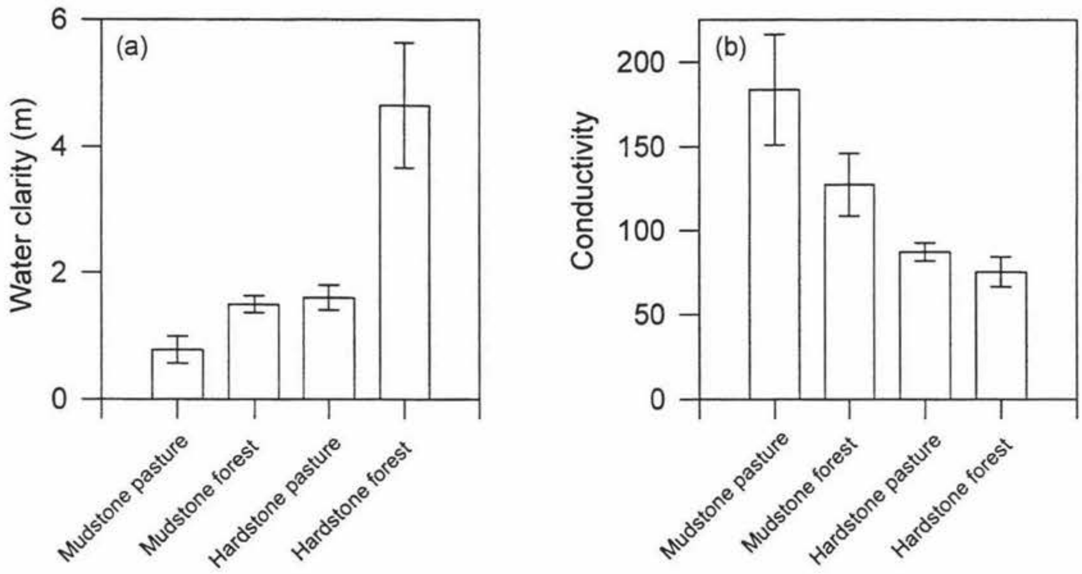


Figure 2.5. Mean (± 1 S.E.) (a) water clarity and (b) conductivity ($\mu Scm^{-2}$) measured in 33 streams in the Whanganui River catchment, between March and April, 1996, which have mudstone or hardstone substrates and drain pasture or indigenous forest.

Table 2.4. One-way analysis of variance tests between environmental and biotic measurements for 33 streams in the Whanganui River catchment, between March and April, 1996, which have mudstone or hardstone substrates and drain pasture or indigenous forest.

| Variable | *F* Ratio | D.F. | *P* value | Variable | *F* Ratio | D.F. | *P* value |
|---|---|---|---|---|---|---|---|
| Physical | | | | Biological | | | |
| Conductivity | 7.91 | 3,29 | 0.00 | | | | |
| Water clarity | 5.67 | 3,29 | 0.00 | Total abundance | 1.76 | 3,29 | 0.18 |
| Stability | 6.3 | 3,29 | 0.00 | Taxa richness | 3.2 | 3,29 | 0.04 |
| Substrate size | 3.63 | 3,29 | 0.02 | Margalefs index | 3.02 | 3,29 | 0.05 |
| Forest cover | 8.45 | 3,29 | 0.00 | Simpson's index | 1.33 | 3,29 | 0.29 |
| Width | 4.98 | 3,29 | 0.01 | Berger-Parker index | 2.34 | 3,29 | 0.09 |
| Depth | 0.84 | 3,29 | 0.48 | MCI | 7.86 | 3,29 | 0.00 |
| Velocity | 3.52 | 3,29 | 0.03 | QMCI | 2.16 | 3,29 | 0.11 |
| pH | 1.99 | 3,29 | 0.14 | EPT score | 2.34 | 3,29 | 0.09 |
| Slope | 0.44 | 3,29 | 0.73 | Axis 1 scores | 6.0 | 3,29 | 0.00 |
| Altitude | 14.7 | 3,29 | 0.00 | Axis 2 scores | 4.13 | 3,29 | 0.02 |

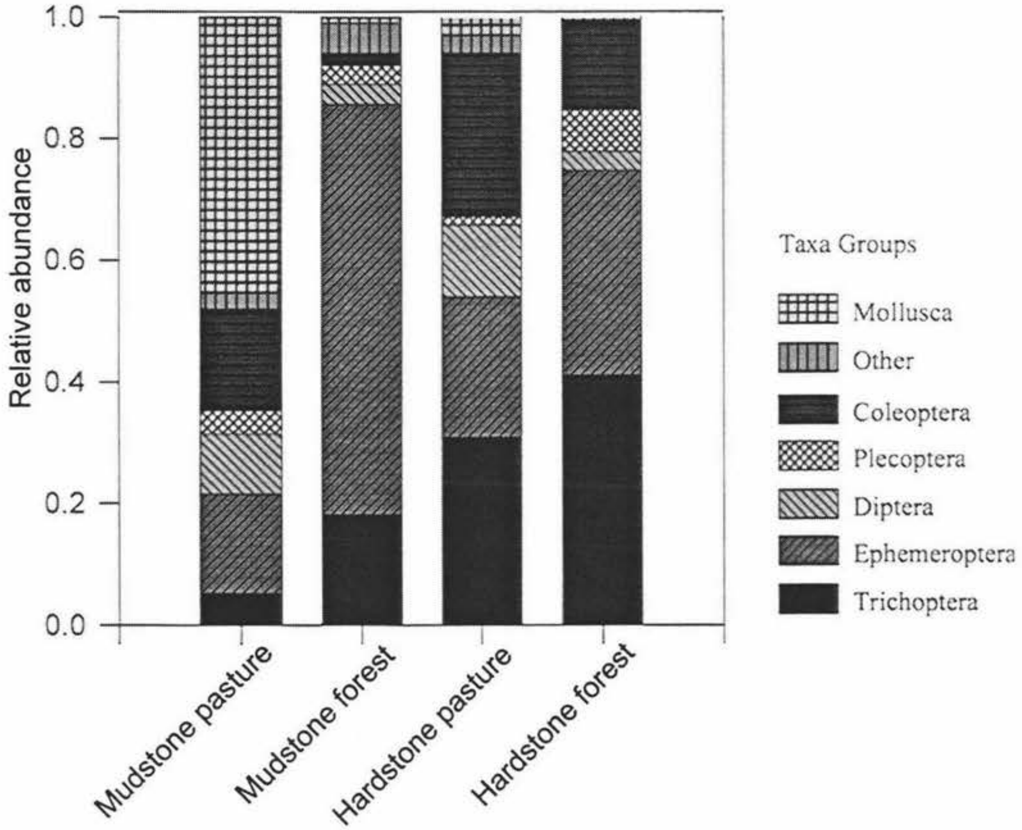


Figure 2.6. Mean relative abundance of individuals from seven taxonomic groups measured in 33 streams in the Whanganui River catchment, between March and April, 1996, which have mudstone or hardstone substrates and drain pasture or indigenous forest.

In the ordination analysis land use classes (of both geology types) were segregated on axis 1 (Fig. 2.9) with geology separated on axis 2, particularly between forested streams. The percentage of variance explained by axis 1 and 2 was 32% and 11%, respectively. Increasing axis 1 scores were indicative of sites with less pollution sensitive taxa particularly in the mudstone pasture class. *P. antipodarium* was strongly associated with these high axis 1 scores while *Deleatidium* spp. commonly occurred at sites that scored low on axis 1. Mudstone substrate streams were located higher on axis 2 with *Austroclima* spp., *Pycnocentria* spp. and *Zephlebia dentata* commonly occurring here as well. Elmid beetles preferred streams low on axis 2. A number of variables were significantly correlated with the first DECORANA axis. These included positive correlations with conductivity, stability, and negative correlations with water clarity, slope, substrate size and catchment forest cover. Velocity was negatively correlated with the second DECORANA axis (Table 2.5).

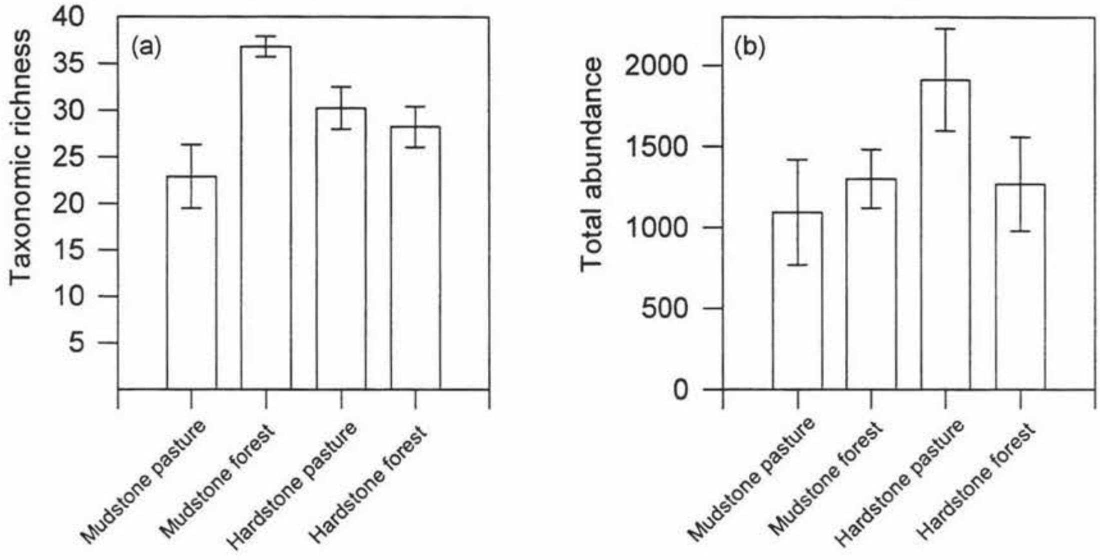


Figure 2.7. Mean (± 1 S.E.) (a) taxonomic richness and (b) total abundance measured in 33 streams in the Whanganui River catchment, between March and April, 1996, which have mudstone or hardstone substrates and drain pasture or indigenous forest.

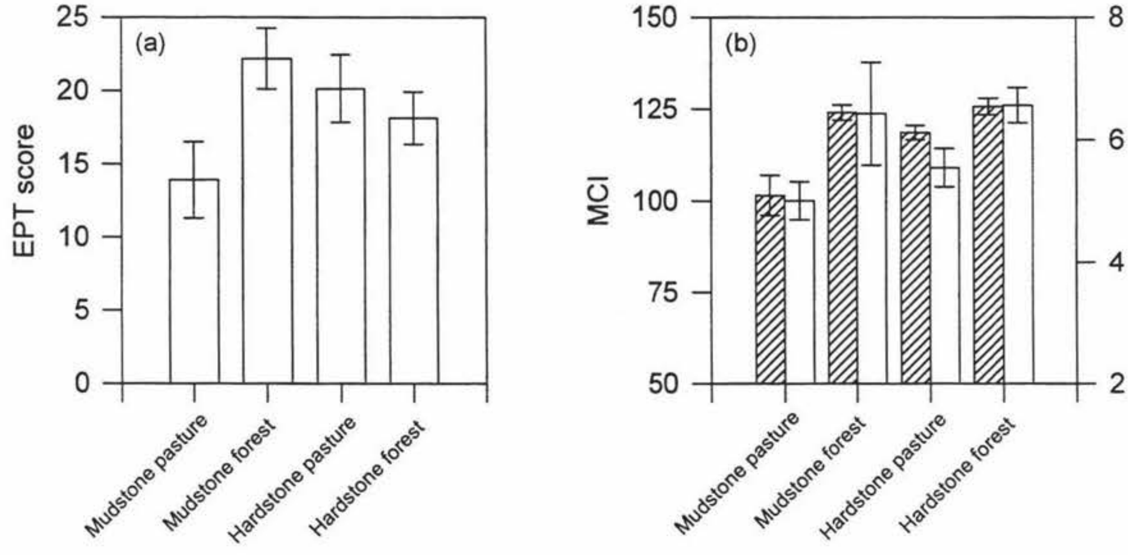


Figure 2.8. Mean (± 1 S.E.) (a) EPT scores and (b) MCI (diagonal hatching) / QMCI (no hatching) measured in 33 streams in the Whanganui River catchment, between March and April, 1996, which have mudstone or hardstone substrates and drain pasture or indigenous forest.

Table 2.5. Pearson correlations between environmental and biotic measurements for 33 streams in the Whanganui River catchment, between March and April, 1996. * P < 0.05, ** P < 0.01, *** P < 0.001.

| | Water clarity | Conductivity | pH | Substrate size | Stability | Width | Depth | Velocity | Forest cover % | Slope (degrees) | Altitude |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Conductivity | **-0.55 | - | - | - | - | - | - | - | - | - | - |
| pH | -0.28 | *0.37 | - | - | - | - | - | - | - | - | - |
| Substrate size | *0.4 | *-0.40 | -0.23 | - | - | - | - | - | - | - | - |
| Stability | *-0.36 | ***0.52 | 0.35 | ***-0.84 | - | - | - | - | - | - | - |
| Width | **0.5 | -0.33 | *-0.38 | ***0.58 | ***-0.59 | - | - | - | - | - | - |
| Depth | 0.05 | 0.03 | 0.19 | -0.13 | 0.05 | 0.18 | - | - | - | - | - |
| Velocity | 0.09 | 0.1 | 0.1 | 0.34 | -0.23 | 0.35 | 0.15 | - | - | - | - |
| Forest cover % | -0.39* | **-0.45 | -0.34 | *0.43 | *-0.42 | 0.32 | 0.17 | 0.04 | - | - | - |
| Slope | -0.04 | -0.20 | 0.14 | 0.23 | -0.26 | -0.17 | 0.01 | 0.08 | 0.16 | - | - |
| Altitude | ***0.58 | ***-0.6 | **-0.54 | 0.13 | *-0.36 | *0.39 | 0.17 | 0.07 | 0.26 | 0.21 | - |
| Taxa richness | 0.26 | ***-0.56 | -0.25 | **0.47 | **0.46 | 0.16 | -0.09 | 0.35 | *0.41 | -0.05 | 0.07 |
| Total abundance | 0.2 | -0.27 | -0.25 | *0.39 | *-0.36 | 0.14 | *-0.44 | 0.16 | 0.07 | 0.08 | 0.31 |
| Margalefs index | *0.36 | ***-0.59 | -0.22 | **0.53 | **-0.53 | 0.25 | 0.07 | -0.2 | **0.46 | 0.18 | 0.09 |
| Simpson's index | -0.18 | 0.28 | -0.06 | -0.28 | *0.35 | -0.01 | -0.1 | 0.02 | -0.2 | -0.34 | -0.07 |
| Berger-Parker index | -0.27 | *0.36 | 0.04 | -0.34 | *0.42 | -0.06 | -0.03 | 0.04 | -0.27 | -0.30 | 0.19 |
| MCI | ***0.59 | ***-0.70 | *-0.37 | **0.46 | ***-0.49 | 0.32 | 0.03 | -0.17 | **0.48 | 0.22 | 0.32 |
| QMCI | 0.28 | **-0.49 | -0.18 | 0.22 | -0.26 | 0.11 | -0.06 | -0.22 | 0.28 | ***0.54 | 0.26 |
| EPT score | 0.28 | ***-0.62 | -0.27 | ***0.49 | **-0.45 | 0.2 | -0.07 | -0.13 | 0.32 | 0.32 | 0.09 |
| Axis 1 scores | **-0.38 | **0.5 | 0.26 | ***-0.64 | ***0.62 | -0.33 | 0.06 | -0.05 | **-0.48 | -0.6 | -0.27 |
| Axis 2 scores | 0.02 | 0.14 | 0.27 | -0.24 | 0.2 | -0.25 | 0.21 | *-0.36 | 0.13 | -0.04 | -0.07 |

TWINSPAN analysis formed five groups (Fig. 2.10) comprising of two silt bottom streams (E) and four main groups (A, B, C and D). Group A and D consist of streams with predominantly soft substrates while B and C consisted of those with mainly hard substrates. There were significant differences among communities in the four land use/geology classes when analysed using MRPP ($r = 0.078$, $P = 3\times10^{-6}$). An average distance (Sorenson) of 0.87 within the mudstone pasture class indicated that community structure varied most among streams within this class whereas average distance within the mudstone forest, hardstone pasture and hardstone forest classes were 0.57, 0.67 and 0.65, respectively. This is also evident in the DECORANA analysis (Fig. 2.9).

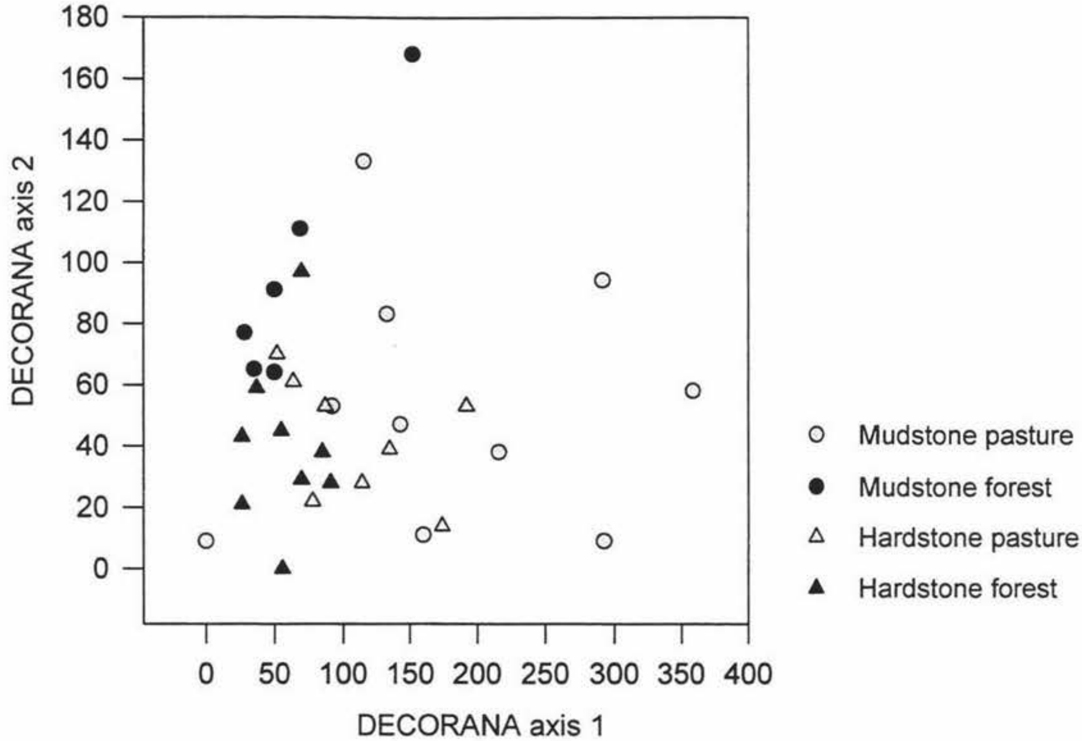


Figure 2.9. Axis 1 as a function of axis 2, for DECORANA analysis of community structure measured in 33 streams in the Whanganui River catchment, between March and April, 1996, which have mudstone or hardstone substrates and drain pasture or indigenous forest.

Land use effects 1997

*Physicochemical Characteristics*

At sites sampled in 1997 only two physicochemical parameters differed significantly between indigenous forest and pastoral land uses (Table 2.6). Conductivity was higher in the pasture streams (Fig. 2.11a) while velocity was lowest at these sites.

*Community structure*

The MCI and QMCI biotic indices and taxonomic richness were higher in the forested class (Fig. 2.11b). However, measures of eveness did not differ (Table 2.6). As in 1996, the pasture class had high numbers of Mollusca (34%), mainly *P. antipodarium*. Crustacea (Taltricidae, Amphipoda) were also common at pasture sites but principally at Upokongaro and Makirikiri. Therefore this abundance of Crustacea is not representative of the whole pasture class. Ephemeroptera were relatively more abundant in the forested class (40% opposed to 28%). *Deleatidium* spp., *Austroclima* spp. and *Nesameletus* spp. were all common in the forest class.

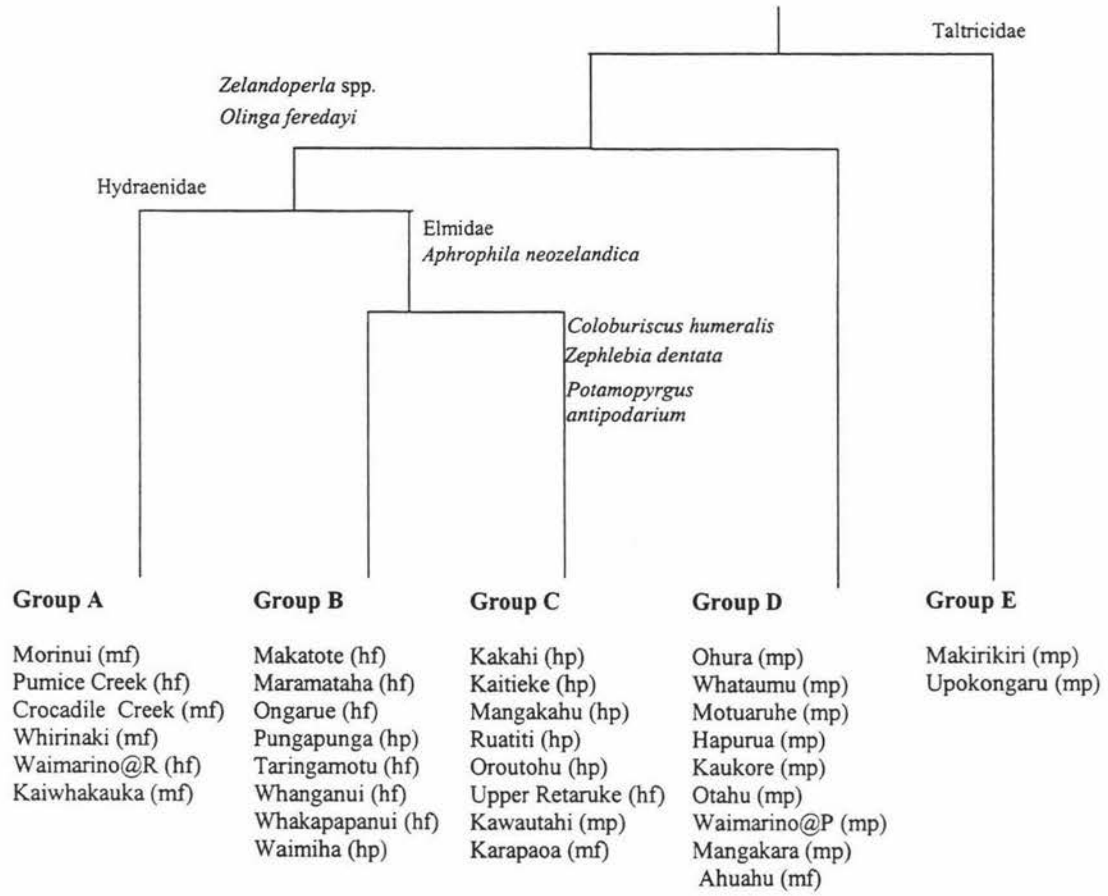


Figure 2.10. TWINSPAN analysis of 33 streams in the Whanganui River catchment sampled between March and April, 1996. m = mudstone, h = hardstone, f = forest, p = pasture.

Trichoptera were also abundant at forested sites and included *Pycnocentria funeria, Zelolessica cheira,* and *Olinga feredayi*. Plecoptera were rare in both classes but more abundant at forested sites (Fig. 2.12).

Overall community structure of the bush and pasture classes differed significantly (MRPP, $r$=0.025, $P$=0.01). Cluster analysis and TWINSPAN did not group sites distinctly in terms of land use although the initial groups formed using cluster analysis did comprise predominantly of bush or pasture sites. DECORANA also grouped bush sites to the lower left of a plot of axis 1 on 2 although there was considerable spread. (Fig. 2.13).

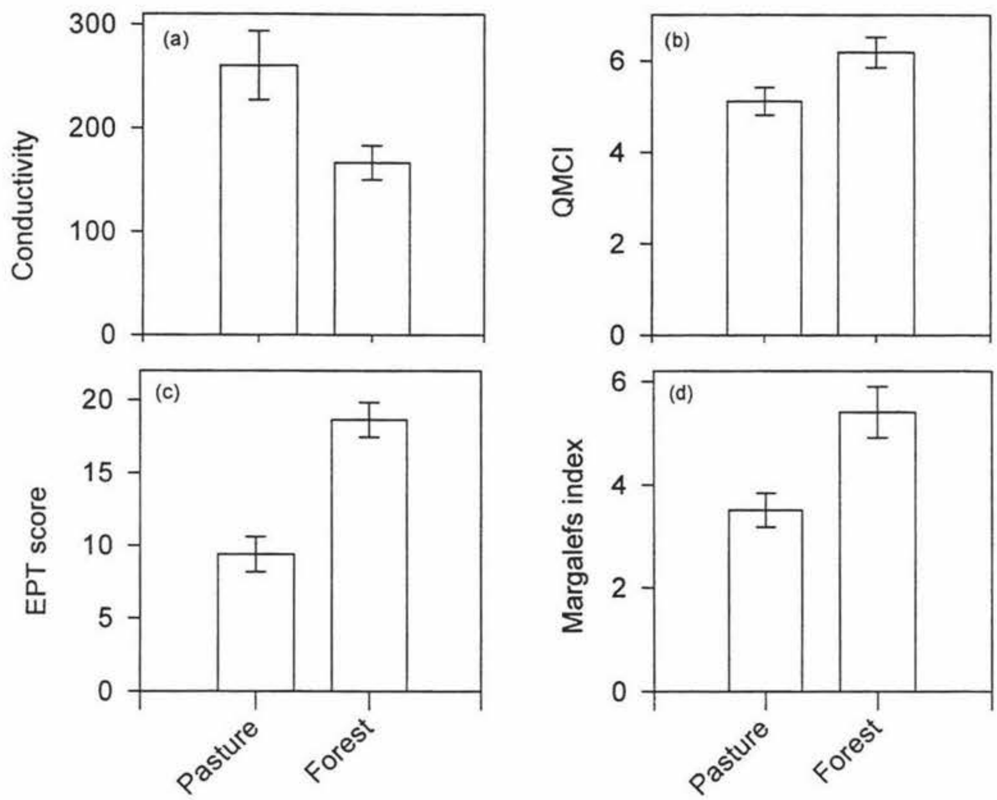


Figure 2.11. Mean (± 1 S.E.) (a) conductivity, (b) QMCI, (c) EPT score and (d) Margalefs index measured in 30 streams in the Whanganui River catchment, February, 1997, which have mudstone substrates and drain pasture or indigenous forest.

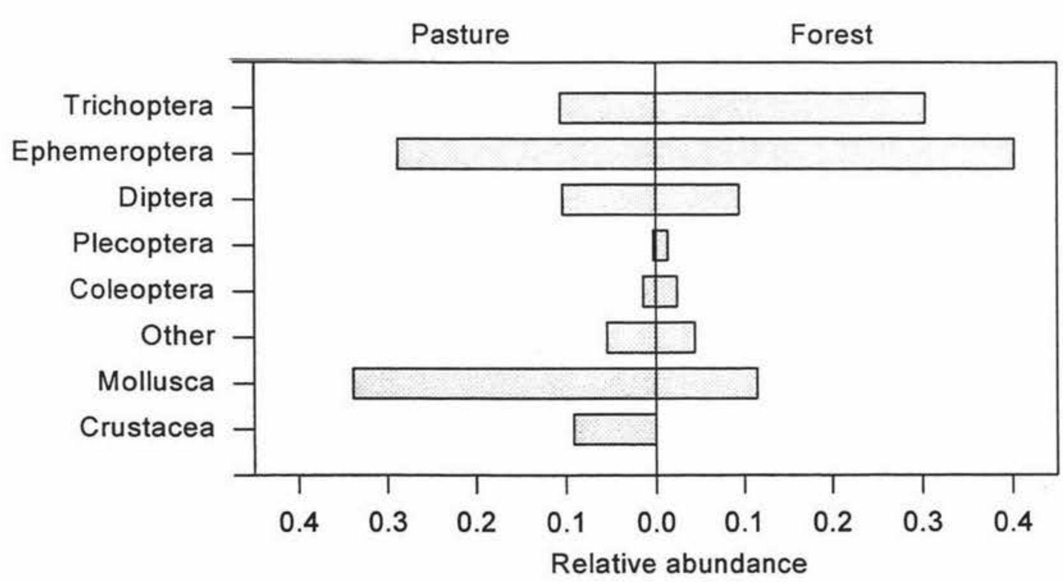


Figure 2.12. Mean relative abundance of individuals measured in 30 streams in the Whanganui River catchment, February, 1997, which have mudstone substrates and drain pasture or indigenous forest.

Table 2.6. One-way analysis of variance tests between environmental and biotic measurements for 30 streams in the Whanganui River catchment, February, 1997, which have mudstone substrates and drain pasture or indigenous forest.

| Variable | *F* Ratio | D.F. | *P* value | Variable | *F* Ratio | D.F. | *P* value |
|---|---|---|---|---|---|---|---|
| Physical | | | | Biological | | | |
| Conductivity | 7.69 | 1,29 | 0.01 | Chlorophyll *a* | 0.16 | 1,28 | 0.70 |
| Velocity | 7.14 | 1,29 | 0.01 | Total abundance | 1.60 | 1,29 | 0.22 |
| Water clarity | 0.82 | 1,29 | 0.37 | Taxa richness | 9.46 | 1,29 | 0.01 |
| Stability | 1.51 | 1,29 | 0.23 | Margalefs index | 10.1 | 1,29 | 0.00 |
| Substrate size | 1.08 | 1,29 | 0.31 | Simpson's index | 1.01 | 1,29 | 0.32 |
| Forest cover % | 49.91 | 1,29 | 0.00 | Berger-Parker index | 1.94 | 1,29 | 0.18 |
| Width | 2.44 | 1,29 | 0.13 | MCI | 17.92 | 1,29 | 0.00 |
| Depth | 0.55 | 1,29 | 0.46 | QMCI | 6.65 | 1,29 | 0.02 |
| pH | 0.48 | 1,29 | 0.49 | EPT score | 15.84 | 1,29 | 0.00 |
| Slope | 0.26 | 1,29 | 0.62 | Axis 1 scores | 3.55 | 1,29 | 0.07 |
| Altitude | 1.46 | 1,29 | 0.24 | Axis 2 scores | 9.54 | 1,29 | 0.0 |

Table 2.5. Pearson correlations between environmental and biotic measurements for 30 streams in the Whanganui River catchment, February, 1997, which have mudstone substrates and drain pasture or indigenous forest. * P < 0.05, ** P < 0.01, *** P < 0.001.

| Variable | Chlorophyll *a* | Water clarity | Conductivity | pH | Substrate size | Stability | Width | Depth | Velocity | Forest cover % | Slope | Altitude |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Water clarity | 0.18 | - | - | - | - | - | - | - | - | - | - | - |
| Conductivity | 0.28 | -0.07 | - | - | - | - | - | - | - | - | - | - |
| pH | -0.15 | 0.07 | 0.14 | - | - | - | - | - | - | - | - | - |
| Substrate size | 0.21 | 0.11 | 0.23 | 0.28 | - | - | - | - | - | - | - | - |
| Stability | -0.18 | -0.34 | -0.12 | -0.33 | -0.34 | - | - | - | - | - | - | - |
| Width | 0.19 | 0.10 | -0.08 | -0.31 | 0.14 | -0.32 | - | - | - | - | - | - |
| Depth | 0.21 | **0.46 | -0.03 | 0.00 | 0.03 | -0.16 | 0.31 | - | - | - | - | - |
| Velocity | -0.20 | -0.02 | -0.32 | 0.07 | -0.04 | -0.19 | ***0.50 | 0.01 | - | - | - | - |
| Forest cover % | 0.04 | 0.32 | ***-0.52 | 0.07 | 0.17 | -0.31 | 0.32 | *0.37 | 0.18 | - | - | - |
| Slope | 0.24 | 0.30 | 0.01 | 0.03 | 0.20 | **-0.49 | -0.09 | 0.20 | -0.17 | 0.17 | - | - |
| Altitude | -0.19 | 0.16 | ***-0.71 | -0.10 | -0.24 | -0.11 | 0.17 | 0.14 | *0.43 | 0.27 | 0.10 | - |
| Taxa richness | 0.12 | 0.09 | -0.19 | -0.05 | 0.16 | -0.29 | 0.35 | 0.34 | 0.12 | *0.42 | 0.07 | 0.17 |
| Total abundance | 0.09 | 0.23 | -0.08 | -0.13 | 0.12 | 0.00 | 0.06 | *0.39 | -0.09 | 0.21 | -0.30 | -0.10 |
| Margalefs index | 0.11 | 0.1 | -0.26 | -0.00 | 0.13 | -0.29 | 0.30 | 0.20 | 0.19 | 0.34 | 0.12 | 0.24 |
| Simpson's index | *-0.37 | *0.03 | -0.13 | 0.13 | -0.09 | 0.23 | *-0.39 | -0.5 | -0.13 | -0.10 | 0.05 | -0.05 |
| Berger-Parker index | -0.21 | -0.09 | 0.06 | -0.05 | -0.06 | 0.39 | -0.34 | -0.27 | -0.16 | -0.28 | 0.01 | -0.21 |
| MCI | -0.00 | *0.41 | -0.39 | -0.06 | 0.08 | ***-0.5 | 0.28 | 0.34 | 0.34 | ***0.62 | **0.48 | *0.44 |
| QMCI | -0.05 | 0.22 | -0.25 | -0.02 | -0.00 | **-0.33 | 0.13 | 0.05 | 0.25 | 0.25 | 0.17 | 0.23 |
| EPT score | 0.05 | 0.19 | -0.30 | 0.02 | 0.13 | *-0.43 | *0.38 | *0.39 | 0.18 | ***0.57 | 0.14 | 0.28 |
| Axis1 scores | 0.02 | *0.06 | 0.22 | 0.03 | -0.12 | ***0.60 | *-0.41 | 0.03 | ***-0.5 | -0.23 | *-0.40 | *-0.43 |
| Axis2 scores | -0.16 | -0.54 | 0.26 | 0.02 | 0.01 | 0.23 | -0.03 | -0.34 | -0.03 | *-0.47 | -0.34 | -0.18 |

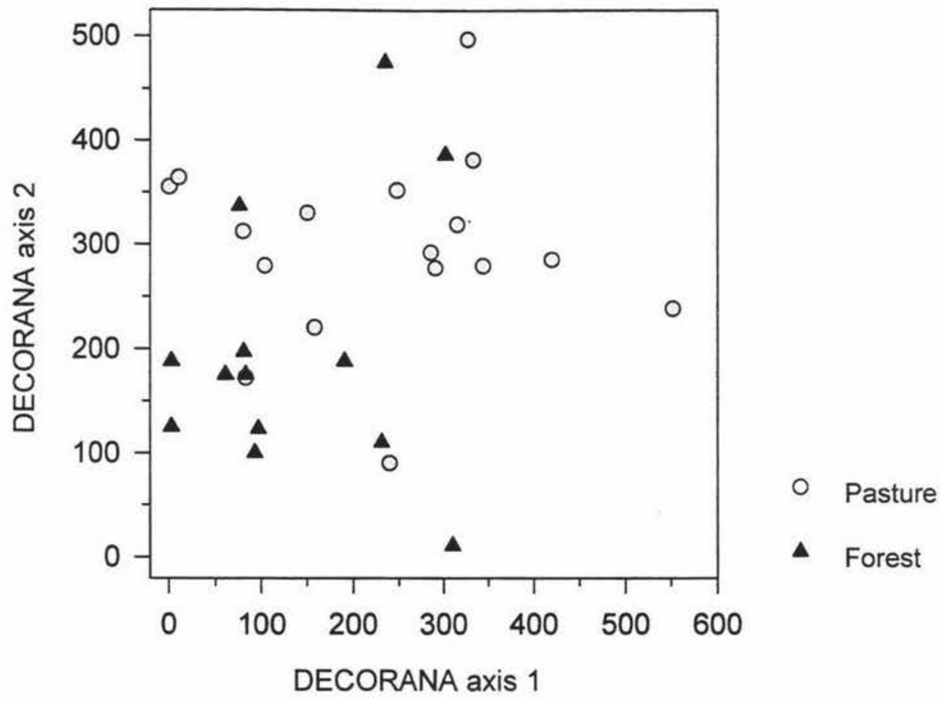


Figure 2.13. DECORANA analysis of community structure measured in 30 streams in the Whanganui River catchment, February, 1997, which have mudstone substrates and drain pasture or indigenous forest.

The percentage of variation explained by DECORANA axis 1 and 2 was 33% and 19%, respectively. Taltricidae, *P. antipodurium* and *Physa* spp. were associated with high axis 1 scores, with the mayflies *Nesameletus* spp., *Acanthophlebia cruentata* positioned at the opposite end. On the high end of axis 2 sites with *Hydrobiosis umbripennis* contrasted those at the lower with greater abundance of the mayfly's *Deleatidium* spp. and *Nesameletus* spp.

## DISCUSSION

Many physical and biological factors varied among the four land use and geology classes. Multivariate analysis of community structure suggests that macroinvertebrate assemblages have different characteristics depending on geology type and catchment land use and that these differences are significant. A distinction between geological classes provided by physical and biological variables was apparent and is accentuated by pasture land use.

Other studies have noted the importance of geomorphology and climate in influencing macroinvertebrate communities (Harding, 1994) and algal biomass (Friberg and Winterbourn, 1997). Variations in geology may also mask the affects of land use between catchments (Richards *et al.,* 1997).

Few environmental measurements or biotic characteristics were statistically different between 1996 and 1997 suggesting some continuity of sites between years. Lower flow conditions were responsible for greater conductivity (e.g., Henderson, 1996) and clarity in 1997. Temporal variation in community structure is considered to be a result of the combined effects of disturbance frequency and habitat heterogeneity (Townsend, 1989; Poff and Ward, 1990; Scarsbrook and Townsend, 1993; Townsend and Hildrew, 1994). While species diversity was similar between 1996 and 1997 numbers of dominant taxa differed significantly so the abundance of many common taxa may relate more to sampling period rather than site location. No major changes occurred in the main taxonomic orders between 1996 and 1997 apart from a large increase in Amphipods at two sites, both of which had sand substrate and low current velocity, conditions that suit aquatic Amphipoda.

Conductivity was higher in mudstone catchments of both land use types due to greater levels of erosion which continually expose fresh material to weathering (Biggs and Gerbeaux, 1993; Maasdam and Smith, 1994; Biggs, 1995; Davies-Colley and Stroud, 1995). There was high relative and numerical abundances of Ephemeroptera, principally *Coloburiscus humeralis*, as well as *Zephlebia dentata* and *Deleatidium* spp. in mudstone forested streams and it appears that Trichoptera and Coleoptera prefer streams in harder Andesitic/Jurassic catchments. Cased caddisflys may be less adept at adhering to softer, smoother substrate and require greater surface irregularities to resist removal, therefore making them more prone to dislocation than mayflies. The cased caddisflies *Olinga feredayi* and *Helicopsyche* spp. were found almost exclusively in hardstone streams. Contrasting to this another cased caddisfly, *Pycnocentria funerea* appeared to be almost exclusively found in mudstone forested streams and was the main species of caddis fly found in this class. *Pycnocentria funerea* are known to favour woody substrate (Henderson pers. comm.) and this may be related to greater amounts of this material present in these streams.

Dominance by the snail *Potamopyrgus antipodarium* in mudstone pasture streams can be attributed to the affects of agricultural development such as mild enrichment (e.g., Scott *et al.,* 1994; Harding and Winterbourn, 1995; Quinn *et al.*, 1997). Agricultural development was responsible for increases in the relative proportion of Diptera and Coleoptera, and a decrease in Ephemeroptera and Trichoptera. Similar species were found within all classes but the proportions of these taxa differed. In 1997, a third of all Trichoptera individuals in the mudstone forest class consisted of the cased caddis fly species *P. aureola* but were collected almost exclusively from one stream. *P. aureola* does not appear to be common in mudstone streams judging from abundance levels in both the 1996 and 1997 studies. This species commonly occurs in larger lowland streams of moderate to swift flow and feed on algae and fine detritus (Cowley, 1978). However, Pitangi stream where nearly all of the individuals were collected is small with comparatively moderate to low algal growth, velocity and channel stability. Most larvae consisted of small early instar individuals which coincides with what would be predicted for this species in autumn (Cowley, 1978). It is possible that chance oviposition by a few females could account for the large number of *P. aureola* larvae present in a particular reach as has been shown for an Australian cased caddisfly (Bunn and Hughes, 1997). Trichoptera were still a large proportion of the taxa found in streams suffering less from the impacts associated with agricultural development.

Conductivity was also higher in pasture streams independent of geology as found elsewhere (Close and Davies-Colley, 1990; Collier, 1995). In both years conductivity was an important chemical measure of land use as streams with predominantly forested catchments had lower conductivity. Parameters such as low turbidity, conductivity and a higher proportion of indigenous forest cover within a catchment appeared to relate to community structure and the number of highly scoring pollution sensitive taxa associated with the MCI water quality index. However, when accounting for the lower relative abundance of these high scoring tolerant species the indigenous forest streams appeared to have more of them hence a QMCI equivalent to any other class.

Many studies have found agricultural development to have a negative effect on species richness (Allan, 1959; Dance and Hynes, 1980; Winterbourn *et al.,* 1981; Lenat, 1988; Quinn and Hickey, 1990b; Scott *et al.,* 1994; Harding and Winterbourn, 1995).

In this study pasture streams had lower diversity although taxa richness was higher in hardstone pasture opposed to hardstone forest streams which is due to a greater abundance of individuals at the hardstone forest sites. Improved algal productivity brought about by increased levels of light (e.g., Behmer and Hawkins, 1986; Friberg and Winterbourn, 1997) and enrichment (Quinn *et al.,* 1992) may also be responsible for higher taxonomic richness and total abundance in the hardstone pasture streams. Increased periphyton biomass may lead to higher invertebrate abundance in mildly impacted streams (e.g., Jowett and Richardson, 1990). However, water quality at the hardstone pasture sites was still relatively high as indicated by low conductivity, turbidity and high MCI scores for these streams. The mean MCI score for the hardstone pasture class was 118 which is not considered polluted as scores of over 100 are considered indicative of 'healthy' streams (Stark, 1993). Predictably taxa diversity and the presence of pollution sensitive taxa positively related to the percentage of catchment in native forest as occurred in 1996 and elsewhere (Allan, 1959; Dance and Hynes, 1980; Winterbourn *et al.,* 1981; Lenat, 1988; Quinn and Hickey, 1990b; Scott *et al.,* 1994; Harding and Winterbourn, 1995). Diversity and pollution sensitive taxa were also negatively influenced by turbidity and suspensoids as extrapolated from visibility measurements (Davies-Colley and Stroud, 1995) which also coincides with other studies (Wagener and LaPerriere, 1985; Quinn *et al.,* 1992; Richards and Host, 1994).

Turbidity levels were higher in mudstone streams regardless of land use. However, these levels were accentuated by the use of land for agricultural purposes (Ryan, 1991; Quinn *et al.,* 1992; Harding and Winterbourn, 1995; Townsend *et al.,* 1997). If conductivity and turbidity are used to measure the effects of land use on waterways allowances must be made for inherent differences in conductivity and suspensoids that occur between streams when attempting to quantify water quality across streams that differ in geology type. Bed stability was lower in mudstone pasture streams compared to the other three classes. Due to the high erodibility of soft sedimentary rock (Maasdam and Smith, 1994) and a lack of indigenous vegetation (Hanchet, 1990; Townsend *et al.,* 1997) higher levels of unstable sediment which require lower critical stream velocities to move them (Newbury, 1984; Carson and Griffith, 1987; Ashworth and Fergusson, 1989) were found in mudstone pasture streams (e.g., Collier, 1995).

Lower abundance and diversity in mudstone pasture streams reflects the susceptibility of these catchments to the impacts brought about from agricultural development. Increased turbidity, suspended sediment and silt deposition have been implicated in other studies leading to reduced invertebrate species richness (Wagener and LaPerriere, 1985; Quinn *et al.,* 1992; Richards and Host, 1994) and abundance (Ryder, 1989; Quinn *et al.,* 1992). Decreases in macroinvertebrate abundance are thought to result from reduced primary production (Quinn *et al.,* 1992), a reduction in food quality (Stumn and Morgan, 1981; Gregory, 1983; Sweeney and Vannote, 1986), bed clogging and low interstitial oxygen (Ryder, 1989). Other studies have found that periphyton biomass and productivity can be increased (Quinn and Hickey, 1990b), decreased (Harding and Winterbourn, 1995) or unaffected (Collier, 1995; Townsend *et al.*, 1997) by agricultural activity. Through agricultural development, light access to the stream channel will increase as a result of the removal of riparian vegetation. However, a corresponding increase in suspensoids (Ryan, 1991) particularly in catchments with more erodable geology's may shade the benthos (Van Nieuwenhuyse and LaPerriere, 1986; Ryan, 1991; Davies-Colley *et al.*, 1992 ) countering light increases above the water column. In this study chlorophyll *a* concentrations were similar between the pasture and native forest classes and appeared to be unrelated to any other measured biotic or abiotic variables. Geology type may override any influence of land use on periphyton.

The many physical and biological variables correlated with axis 1 of the DECORANA analysis suggests that while not all variables differ significantly between land use and geology, or between themselves, a combination of many are responsible for influencing invertebrate community structure. The selection of sites contained in this study are not completely dichotomous in their land use or geological properties and this gradient is evident. However, it is clear from multivariate analysis that invertebrate community structure differs among land use and geological classes, and that these differences are a result of variations in environmental conditions distinct to these classes.

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# Chapter Three

# Longitudinal changes in macroinvertebrate community structure along a large New Zealand river

## ABSTRACT

Both artificial substrates and kick sampling were used to sample macroinvertebrate communities and explore the affects of physicochemical changes along a large New Zealand river. Taxonomic diversity and numbers of pollution sensitive taxa decreased along five sampling sites down the Whanganui River. This was correlated to reduced periphyton biomass and increased suspensoids. *Potamopyrgus antipodarium* was highly abundant at site three and five as both sites had similar macrohabitat suitable for this species. The influence of macrohabitat was apparent from similarities in community structure between site three and five and differences in community structure between sampling techniques.


**Keywords:** benthic macroinvertebrate communities, macrohabitat, artificial substrate, longitudinal, sampling precision, conductivity, water clarity, periphyton, Whanganui.

## INTRODUCTION

Land development practices result in changes in the structure and function of running water ecosystems through catchment modification (Winterbourn, 1986) significantly altering physicochemical parameters and algal biomass (Quinn and Hickey, 1990b). Invertebrate assemblages are integrally linked to in stream physical and chemical characteristics and for this reason are useful environmental indicators (Stark, 1993).

In developed countries the longitudinal ecological pattern of a large river is often strongly influenced by water pollution and flow regulation (Bournaud *et al.*, 1996; Zamora-Muñoz and Alba-Tercedor, 1996). Longitudinal changes of macroinvertebrate fauna may be closely related to the immediate environment or due to other causes such as physicochemical inputs from tributaries. Much of the work concerning water quality in New Zealand has been conducted on small to medium sized rivers, typically with stony substrate. As large rivers usually flow through lowland areas, they are susceptible to anthropogenic influence. However, few studies have been conducted on the effects of physicochemistry on macroinvertebrate communities in large non-cobble dominated waterways like the Whanganui River. Sampling techniques used in shallow stony streams are not always appropriate for large rivers. Consequently, artificial substrates such as multiplate samplers, are used for sampling (see review by Rosenberg and Resh, 1982), particularly reaches that are deep and have boulder or bedrock substrate (Boothroyd and Dickie, 1989), conditions that commonly occur in the middle and lower reaches of the Whanganui River.

It has been demonstrated previously in Chapter One that catchment land use combined with substrate geology influences invertebrate community composition in smaller headwaters of the Whanganui River catchment. The aim of this study is to explore the effects of water quality on macroinvertebrate communities in the main stem of the Whanganui River and to compare the efficiency of kick net and artificial substrate sampling techniques in doing this.

## STUDY SITES

Five sites were sampled along the Whanganui River between Taumaranui and Kaiwhaiki (Fig. 3.1). Site five was located 14 km upstream from the river mouth and site numbers decreased in order upstream. Altitude ranged from < 3 m to 160 m a.s.l.

Site one was situated 8 km upstream of Taumaranui were the river flows through land developed for pastoral agriculture with water extraction for hydroelectric power generation occurring on both the Whanganui and Whakapapa tributaries. However, these headwaters are partially snow fed arising at high altitude on Mt Tongariro and Mt Ruapehu in Tongariro National Park and the river is at its most 'natural' state at this site. The predominant geology type is of volcanic origin. Channel characteristics in this section of the river consist mainly of fast flowing riffle over boulder/cobble substrate.

Site two was approximately 300 m below the confluence of the Retaruke River. The main tributaries to enter the Whanganui above this point are the Ongarue, Ohura and the Retaruke. The Ongarue is a large tributary and the geology of its headwaters consist principally of the pumice soils of the Hauhungaroa range which are well forested. The Ohura river drains land predominantly in pastoral agriculture with soils formed from siltstone which are very prone to erosion. The Retaruke headwaters are located in indigenous bush with substantial areas downstream draining agriculturally developed land.

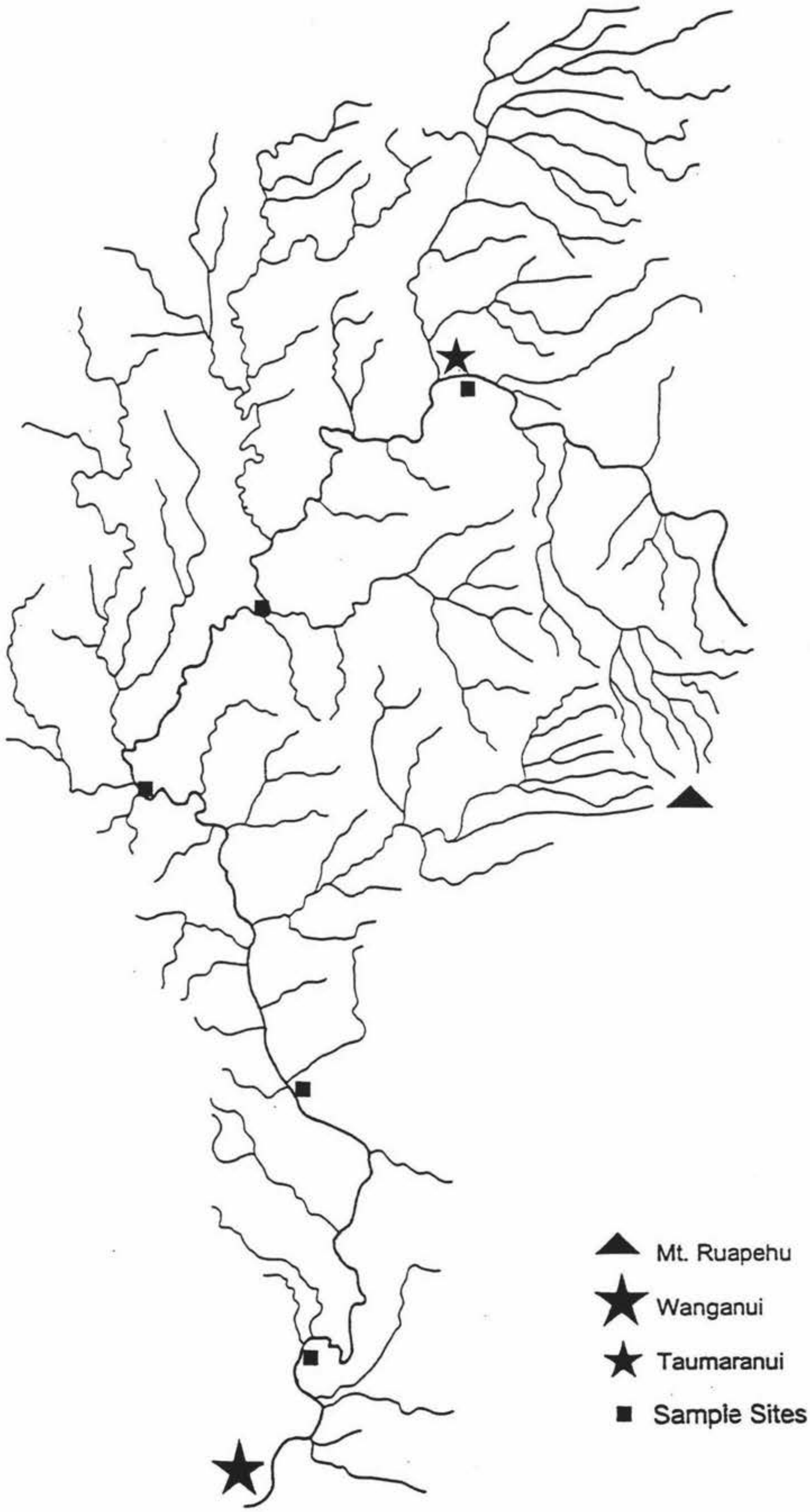


Figure 3.1. Location of the five sites along the main stem Whanganui River where artificial substrate samples and kick samples were collected between the 13 and 15 of May, 1997.

At site two geology formed by Tertiary sediments was influential in creating many deep sections with mudstone bedrock substrate mixed with riffle sections of large cobble substrate.

Site three was situated downstream of the Whangamomona river confluence. Both the Whangamomona and the Tangarakau Rivers, which enter the Whanganui further upstream, drain catchments comprising of sandstone and jointed mudstone. The banks of the Whanganui at this point are in many places steep and high restricting light entry. Riffles are fewer as the channel is confined and deep.

Macrohabitat at site four was similar to site two except levels of deposited sediment were greater. Macrohabitat is defined here as factors such as velocity and overall substrate qualities characteristic to a given reach. The last major tributary, the Manganuioteao, enters approximately 16 km upstream. This tributary drains the western indigenous forest/grassland slopes of Mt Ruapehu so water quality is higher than for other tributaries. No additional major tributaries enter between sites four and five. Substrate at site five consisted mainly of deposited silt and mudstone bedrock. Gradient and velocity were lower here than the upstream sites. Tidal influence and salt water penetration can occur at site five depending on river levels.

## METHODS

*Sampling protocol*

Invertebrates were collected between the 13 and 15 of May, 1997. At each of the five sites, three 60 second kick samples were taken using a 250 μm mesh kick net as the sampler moved progressively upstream. Two artificial substrates were collected from each site that had been placed in the river 6 weeks prior to their removal on the above date.

Samples were sieved through a 250 μm Endecott sieve and animals identified and enumerated with a 40× microscope using available keys (McFarlane, 1951; Winterbourn, 1973; Chapman and Lewis, 1976; Cowley, 1978; Winterbourn and Gregson, 1989; Towns and Peters, 1996). If taxa could not be identified to species they were assigned to apparent morphospecies.

Measurement of physicochemical parameters at each site were undertaken at the same time as the collection of invertebrate samples. Conductivity was measured using an Orion 122 portable conductivity meter and water samples were taken back to the lab for pH analysis using an Orion 250A pH meter. Black disk measurements were used to measure water clarity (Davies-Colley, 1988). Current speed at points where artificial substrates and kick samples were collected was categorised as being of low, medium or high velocity (Table 3.1). Depth of artificial substrates at the time of placement ranged from 60 cm to 100 cm.

Algal biomass was assessed by extracting photosynthetic pigments (chlorophyll *a* and phaeophytin) in 90% acetone, for 24 hours at 5 °C, from five small cobbles (mean circumference = 10 cm) collected on each sampling occasion. Absorption was measured on a Jenway 6105 Spectrophotometer. Total pigment concentration was calculated using the method of Moss (1967 a, b) and corrected for stone surface area (Graham *et al.*, 1988).

*Artificial substrates*

Artificial substrates were removed using a dip net placed around them before disturbance and transferred to a sorting tray. They were then dismantled with all animals removed using a scrubbing brush and preserved in 70% alcohol. Artificial substrates based on the multiplate design of Hester and Dendy (1962), comprised of 14 square perspex plates (80 mm × 80 mm) separated by 20 mm × 20 mm square spacers. Plates were mounted at the end of a 1 metre steel rod with eight upper spaces of 3 mm and five lower spaces of 6 mm. Plates were constructed of 3 mm clear perspex uniformly roughened from belt sanding of both sides using 60 grade sandpaper. The total area available for macroinvertebrate colonisation on each substrate was 0.187 $m^2$. Substrates were stabilised on the river bed using weighted grapnel type anchors with four protruding 20 cm steel rods attached to the steel shaft of each artificial substrate. Both artificial substrate replicates at each site were attached to stable submersed objects such as imbedded logs or very large boulders using 7 mm braided polypropylene rope. Artificial substrates were not placed at or near the surface because of the risk of snagging by large floating debris and it has been shown that multiplates located nearer the benthos display a greater consistency of macroinvertebrate assemblage structure among sites (Mason *et al.*, 1973; Boothroyd and Dickie, 1989).

Fixed surface placements for artificial substrates were unsuitable because highly fluctuating water levels can leave them exposed in the lower Whanganui River. Artificial substrate placement was influenced by the logistical constraints of suitable anchorage points and localities.

*Data analysis*

Longitudinal trends in physicochemical and biological variables were tested for significance with Spearman rank correlations. Spearman rank correlations were also calculated between physicochemical variables and six indices of community structure. Variation of indices between sampling techniques was investigated using paired t-tests and Pearson correlations. Community structure was analysed with detrended correspondence analysis (DECORANA), multi-response permutation procedure (MRPP) and two-way indicator species analysis (TWINSPAN) using the PC-ORD statistical package (McCune and Mefford, 1995). Pseudo-species cut levels were 1, 10, 100 and 1000 per sample.

Biological indices used consisted of the MCI (Stark, 1985), QMCI (Stark, 1993), EPT (Lenat, 1988), Simpson's eveness (Simpson, 1949), Berger-Parker dominance (Berger and Parker, 1970) and Margelefs (Clifford and Stevenson, 1975). The formula for these indices are contained in Chapter Two. Data was transformed where appropriate using $\log_{10}(x+1)$ transformations to remove heterogeneity of variance.

## RESULTS

*Physicochemical variables*

Water clarity was the only physicochemical variable to show a significant longitudinal trend ($r_S = 0.9$, $P = 0.04$) (Fig. 3.2a). Levels of chlorophyll *a* also decreased downstream (Fig. 3.2b) but a major drop occurred at site three ($r_S = 0.7$, $P > 0.05$). Conductivity (Fig. 3.2c) and pH (Fig. 3.2d) varied widely ranging between 127 to 178 $\mu Scm^{-2}$, and 7.08 to 7.38, respectively.

*Community structure*

EPT scores showed significant longitudinal change in the artificial substrate samples ($r_S = 1$, $P = 0.00$) with EPT scores and the MCI showing significant longitudinal change in the kick samples (both $r_S = 0.9$, $P = 0.04$).

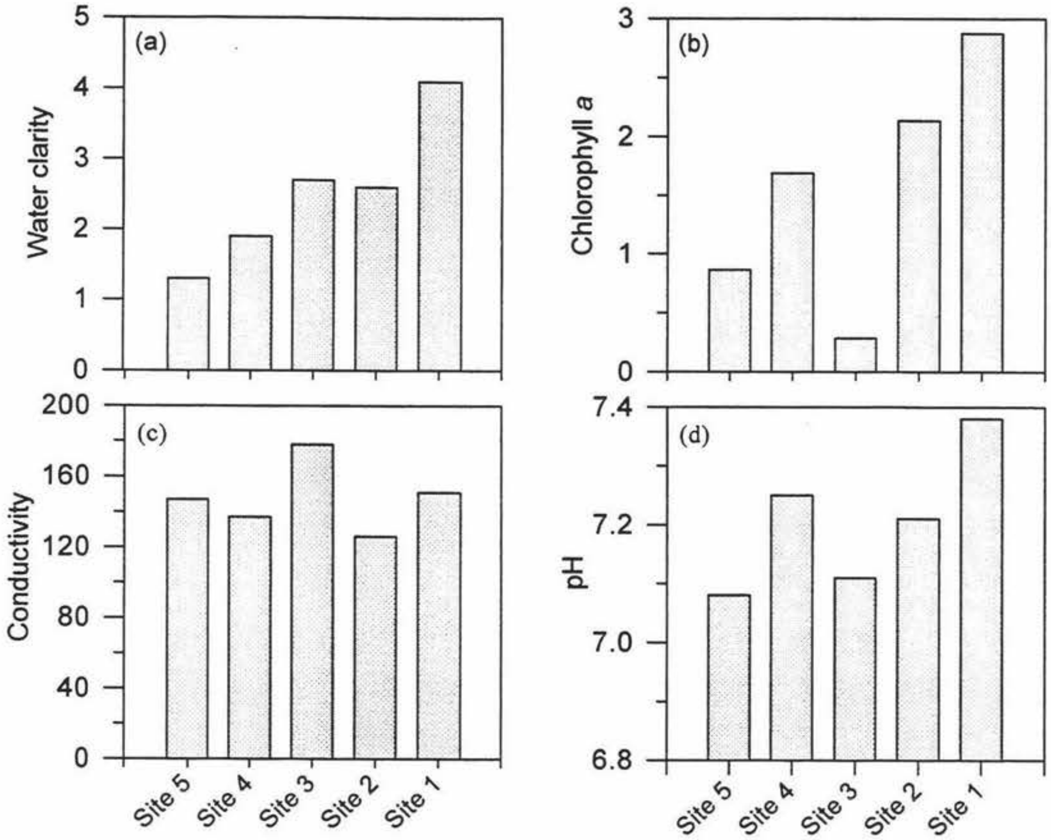


Figure 3.2. (a) Water clarity, (b) chlorophyll *a* ($\mu g\ cm^{-2}$), (c) conductivity ($\mu Scm^{-2}$), and (d) pH measured at five sites along the Whanganui River, May, 1997.

Margalefs index from artificial substrate samples (Fig. 3.3a) decreased longitudinally downstream ($r_S = 1$, $P = 0.00$) although taxonomic richness varied widely among sites and between sampling techniques with the lowest diversity occurring at site five (Fig. 3.3b). Diversity of the communities collected with artificial substrates only exceeded that of samples collected using kick sampling at site three. Macrohabitat at site three resembled that of site one having deep, low current conditions combined with large mudstone boulders, bedrock and deposited silt substrate. Both Margalefs index and taxonomic richness from artificial substrate samples were significantly correlated with water clarity ($r_S = 0.9$, $P = 0.04$ and $r_S = 1$, $P = 0.00$, respectively).

EPT scores for both sampling techniques decreased longitudinally downstream ($r_S = 1$, $P = 0.00$) (Fig. 3.3c). Artificial substrate EPT scores were correlated with water clarity while EPT scores for kick samples correlated with pH and chlorophyll *a* (both $r_S = 0.9$, $P = 0.04$). Total abundance of individuals was similar among kick samples but varied widely among artificial substrates (Fig. 3.3d). The Berger-Parker dominance and

Simpson's eveness indices (Fig. 3.3e) were similar with high values for these at sites three and five. The Berger-Parker dominance and Simpson's eveness indices were correlated between sampling techniques ($r = -0.72$, $P = 0.01$ and $r = -0.68$, $P = 0.04$) with high numbers of the mollusc *Potamopyrgus antipodarium* and an overall lack in taxonomic diversity at sites three and five responsible for high values for these indices (Fig. 3.4). MCI (Fig. 3.3f) and QMCI scores were similar and lowest at site five followed by site three for both artificial substrates and kick samples. MCI values for kick samples decreased significantly downstream ($r_S = 0.9$, $P = 0.04$), and were positively correlated to chlorophyll *a* ($r_S = 0.9$, $P = 0.04$) and pH ($r_S = 0.9$, $P = 0.04$). Kick sample QMCI's were also correlated to pH ($r_S = 0.9$, $P = 0.04$).

At site three over 90% of the individuals present were molluscs belonging to the genus *Potamopyrgus* with the crustacean shrimp *Paratya* common in the kick samples (Fig. 3.4). *Potamopyrgus antipodarium* was clearly the most abundant individual at site three, comparable to site five. However, Mollusca accounted for only one fifth of the individuals at site two and were not present at site one. Trichoptera increased significantly above site three and accounted for over 50% of the individuals collected with artificial substrates at site one. The net spinning caddis fly, *Aoteapsyche colonica* was the most common taxa on artificial substrates at site one and two. A high proportion of Ephemeroptera (predominantly *Coloburiscus humeralis*) were collected from the site one kick samples. Graphing the DECORANA ordination axis 1 against axis 2 displayed grouping between sample types within a site and a gradient of sites along the first axis (Fig. 3.5) starting from site five to site three, site four, site two and lastly site one.

Similarities between site five and site three were evident as these samples were closely grouped on all three ordination axis. Two-way indicator species analysis (TWINSPAN) classified both site five sample types, site four artificial substrate and the site three kick samples together. The remaining groups were characterised by the family Elmidae (Coleoptera) and consisted of two remaining groups. One contained all sample types from sites one and two together. The other contained the artificial substrate samples at site three and the kick sample at site four. Multi-response permutation procedure (MRPP), found invertebrate communities to differ significantly among sites for kick samples ($r = 0.33$, $P = 0.001$) and artificial substrates ($r = 0.44$, $P = 0.003$) but not between sampling techniques ($r = -0.003$, $P = 0.42$).

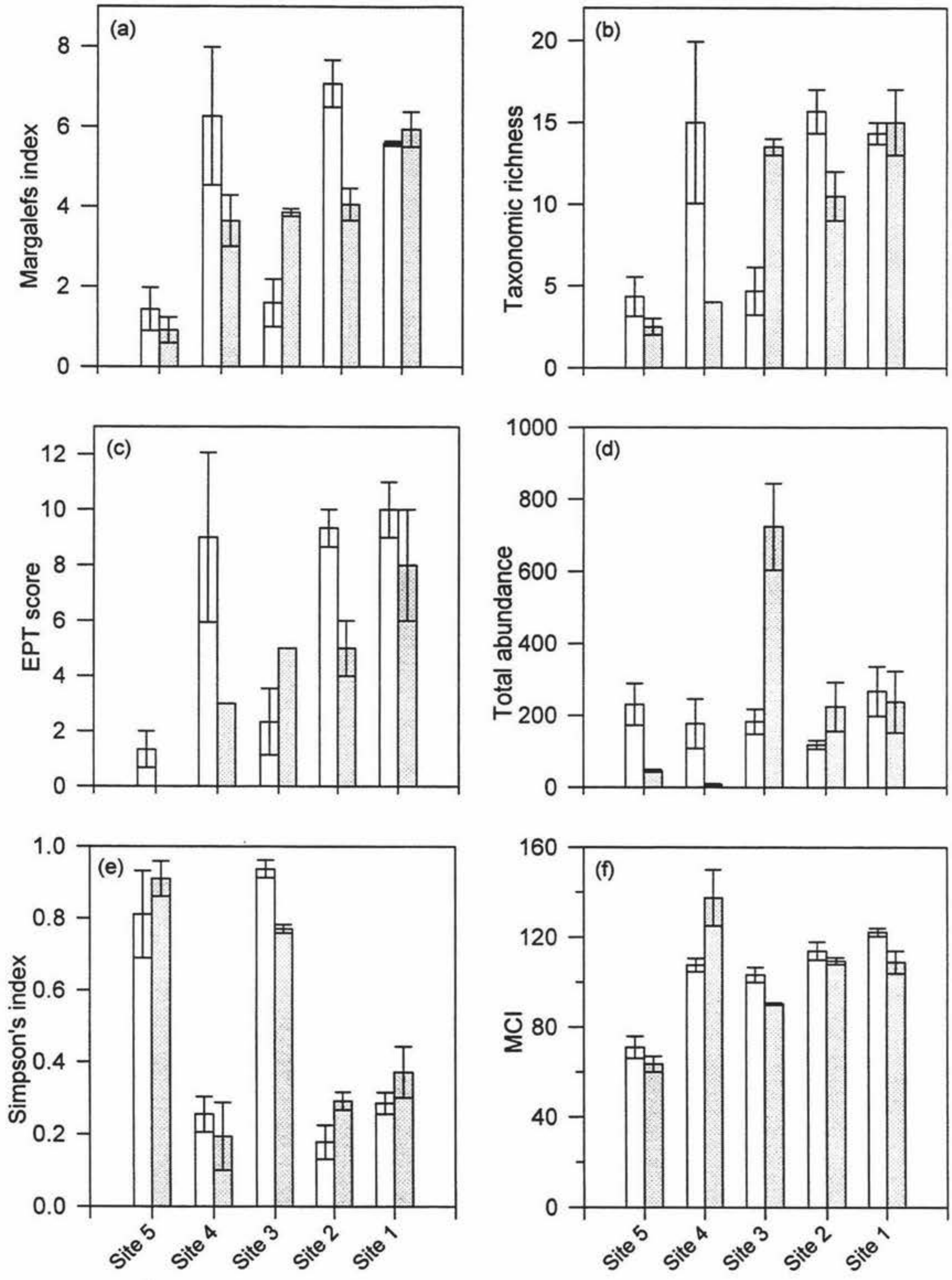


Figure 3.3. Mean (± 1 S.E.) (a) Margalefs index, (b) taxonomic richness, (c) EPT scores, (d) total abundance, (e) Simpson's index of eveness, and (f) MCI values measured at five sites along the Whanganui River, May, 1997. White fills represent kick samples and shaded fills represent artificial substrate samples.

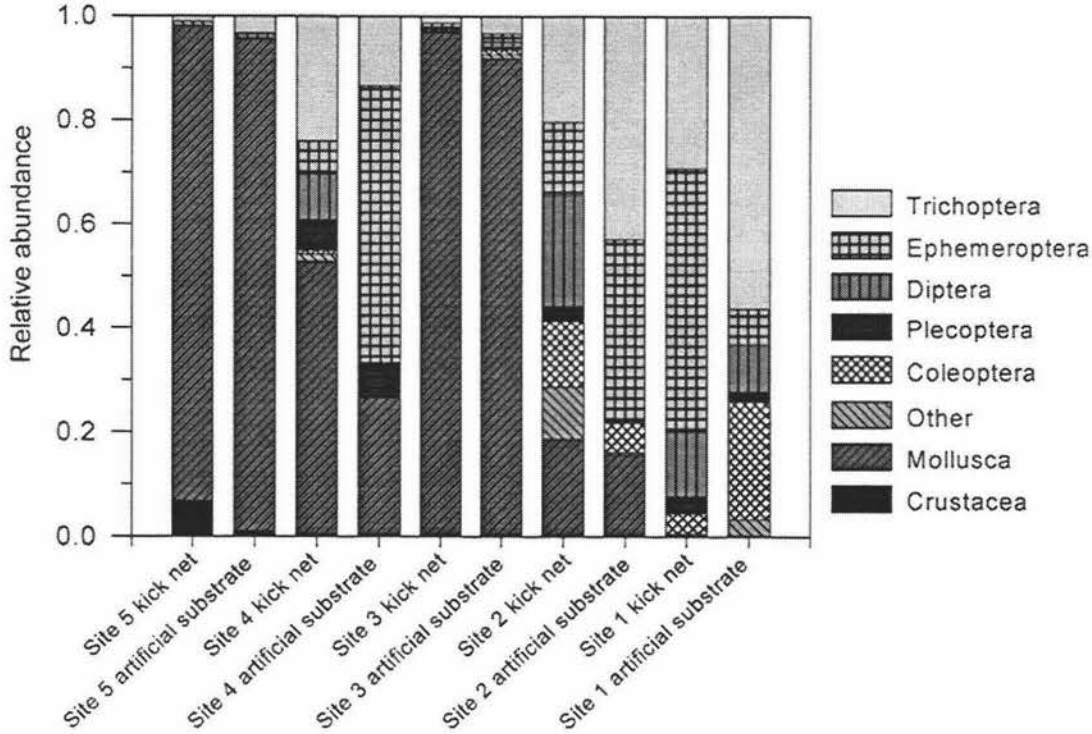


Figure 3.4. Mean relative abundance of individuals from eight taxonomic groups at five sites measured using both artificial substrate and kick sampling along the Whanganui River, May, 1997.

Table 3.1. Velocity and substrate categories for five sites along the Whanganui River, May, 1997.

| Site | Artificial substrates | Kick samples | |
|---|---|---|---|
| | *Velocity* | *Substrate composition* | *Velocity* |
| 1 | medium | boulder/cobble | high |
| 2 | low | boulder/cobble and bedrock | high |
| 3 | medium | large boulder/bedrock | medium |
| 4 | low | boulder/cobble and bedrock | high |
| 5 | medium | bedrock/large boulder and silt | medium |

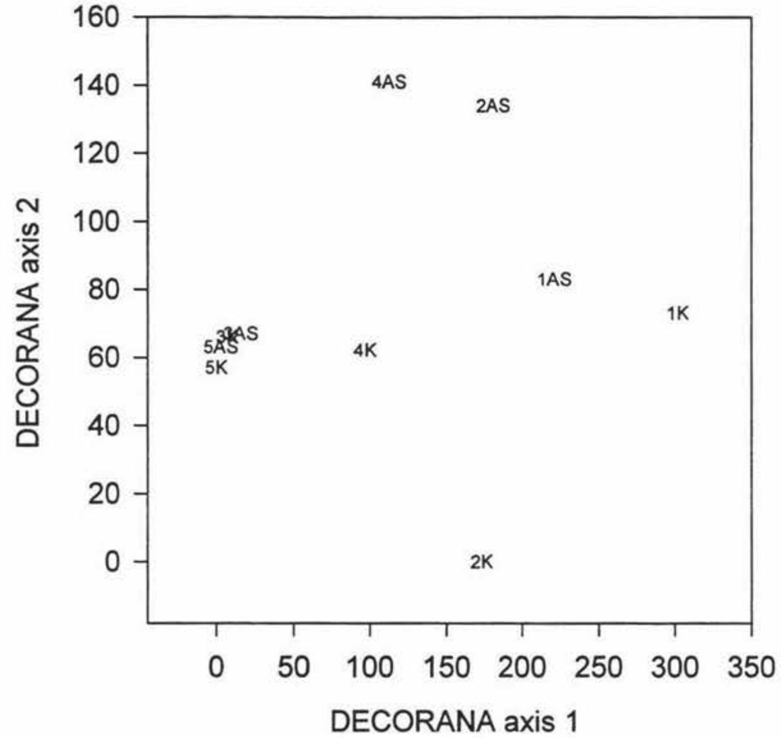


Figure 3.5. DECORANA analysis of community structure measured using both (AS) artificial substrates and (K) kick sampling along the Whanganui River, May, 1996.

## DISCUSSION

Over the course of 130 km between site one and site five, variation in biological and physicochemical factors were observed. Longitudinal trends in water clarity, the MCI, Margalefs index and EPT score were observed but (excluding EPT score) were dependant on the sampling technique used. Macroinvertebrate community structure was similar between site three and five as shown by the mean relative abundance of main taxonomic groups, community dominance and evenness, and ordination. No significant difference in macroinvertebrate community structure between sampling techniques was found with multi-response permutation procedure (MRPP) or paired t-tests. However, longitudinal trends in biological measurements as well as their relationship with physicochemical measurements varied between sampling techniques.

Varying conductivity levels reflect the impact of land use practices (Close and Davies-Colley, 1990; Collier, 1995) occurring in the respective tributaries that discharge into the Whanganui River. Contributions either dilute or increase upstream conductivity rather than having a cumulative effect as was found with water clarity. Water clarity decreased downstream representing an accumulation of deposited silt and suspended material in the river system. Shading from increased turbidity may be responsible for the overall decrease in periphyton biomass (Van Nieuwenhuyse and LaPerriere, 1986; Ryan, 1991; Davies-Colley *et al.*, 1992). The large drop in periphyton biomass at site three may be a result of decreased light access (e.g., Quinn *et al.*, 1997) caused by the presence of stream banks greater than 15 m high. Periphyton biomass may also be reduced by high numbers of the algal grazing gastropod *Potamopyrgus antipodarium* as has been shown elsewhere for grazing invertebrates (Winterbourn and Fegley, 1989; Rosemond, 1993; Kjeldsen, 1996).

Higher periphyton biomass positively influenced the number of pollution sensitive taxa as indicated by the MCI and numbers of Ephemeroptera, Trichoptera and Plecoptera (EPT). Other studies have found *Deleatidium* spp. to increase with periphyton biomass (e.g., Suckling, 1982; Jowett and Richardson, 1990) where growth is not overly dense or filamentous (Davies-Colley *et al.*, 1992) as was found in this study. The relationship between community evenness and dominance (as determined from kick samples) with conductivity could represent the positive response of *Potamopyrgus antipodarium* to increased organic enrichment associated with pastoral agriculture (Scott *et al.*, 1994; Harding and Winterbourn, 1995). However no relationship was found between periphyton biomass and dominance, evenness or conductivity. Growth rate rather than biomass may be a more appropriate measure if high algal grazing is occurring at site three.

Longitudinal change in community structure was shown by multivariate analysis as well as significant downstream decreases in overall taxa and EPT diversity, and pollution sensitive species. Species diversity and EPT numbers collected with artificial substrates were negatively correlated with turbidity and suspended solids as inferred from water clarity (Davies-Colley and Stroud, 1995). Negative impacts on diversity associated with increased suspensoids (e.g., Quinn *et al.*, 1992) are likely to result from substrate clogging (Ryder, 1989) and decreased primary production (Davies-Colley *et al.*, 1992).

Similarities in community structure between site three and five are caused primarily by high relative abundance of *Potamopyrgus antipodarium*. Both sites had similar substrate (very large boulders and bedrock) in areas of low to moderate velocity which provided a suitably stable macrohabitat for *P. antipodarium* to proliferate (Jowett and Duncan, 1990; Harding and Winterbourn, 1995). Increased diversity and changes in taxonomic composition from site three to site four kick samples may result from a combination of less organically enriched water from the Manganuioteao and different macrohabitat characteristics (e.g., Bournaud *et al.*, 1996) which play an important part in shaping community structure (Williams, 1980; Poff and Ward, 1990; Quinn and Hickey, 1990b; Collier, 1995). Two examples of macrohabitat differences from site three to four that are associated with increased diversity include a shift from bedrock/boulder to large cobble substrate (Quinn and Hickey, 1990b) and increased velocity benefiting taxa that have higher oxygen requirements such as Ephemeroptera, Plecoptera and Trichoptera (Nebeker, 1972; Wiley and Kohler, 1980; Williams *et al.*, 1987) which were more abundant at site four.

Part of the life cycle of the Atyid shrimp *Paratya* spp. occurs in or near the sea (Chapman and Lewis, 1976; Taranaki Catchment Commission, 1985; Quinn and Hickey, 1990a). Tidal influence occurs at site five which explains the high numbers of *Paratya* spp. found at this site. Increases in Ephemeroptera and Trichoptera at sites one and two and the absence of *P. antipodarium* at site one are most probably the product of an increase in water quality and velocity (e.g., Lenat, 1988; Quinn and Hickey, 1990b; Harding and Winterbourn, 1995).

At the top two sites, Trichoptera, principally the net spinner *Aoteapsyche colonica,* preferred the artificial substrates as has been found for similar species (e.g., Benfield *et al.,* 1974; Way *et al.,* 1995). At site three the mean abundance of *P. antipodarium* collected on the artificial substrates was three times of that collected with kick sampling where a lot of bedrock made kick sampling difficult. This is an example where artificial substrates were able to collect a greater variety and abundance of taxa because they provided a larger surface area and more suitable macrohabitat for many taxa (e.g., Dickson and Cairns, 1972; Minshall and Minshall, 1977; Taranaki Catchment Commission, 1985; Boothroyd and Dickie, 1989).

It is hard to determine whether six weeks are long enough for adequate colonisation of artificial substrates (Pearson and Jones, 1975; Fredeen and Spurr, 1978). Consideration for the accumulation of periphyton and detritus must also be taken into account as they greatly influence the colonising community (Roby *et al.*, 1978; Winterbourn *et al.*, 1981; Rounick and Winterbourn, 1983; Winterbourn *et al.*, 1984; Boothroyd and Dickie, 1987). Initially it may appear that the maximum quantity of periphyton biomass and detritus present on an artificial substrate would be equivalent or less than that found on the nearby benthos. However, a well placed artificial substrate in moderate to highly disturbed environments will remain stationary during disturbance events that would dislodge or scour equivalent sized natural substrate, thus allowing artificial substrates to build up higher levels of periphyton and detritus (e.g., Canton and Chadwick, 1983). Therefore invertebrate communities may continue to change well after six weeks as a result of the creation of new niches (Meir *et al.*, 1979). These changes may include a decrease in taxonomic richness and an increase in taxonomic domination (Boothroyd and Dickie, 1989). Periphyton growth and detritus on the artificial substrates at sites one, two, and three was noticeably higher than the surrounding benthos which lends support to this theory.

A potential advantage of artificial substrates is the increase of sampling precision between sites (Shaw and Minshall, 1980; Lamberti and Resh, 1985). To maximise precision between artificial substrates their site placement must be similar in terms of flow, depth and light regimes (Rosenberg and Resh, 1982). However, due to the constraints of suitable anchorage sites they were often placed in areas subject to lower velocity than where the kick samples were collected. The influence of velocity was apparent between sampling techniques where kick samples at sites one, two and four were all taken from a riffle and had greater numbers of Ephemeroptera, Plecoptera and Trichoptera, taxa which often have greater oxygen requirements (Nebeker, 1972; Williams *et al.,* 1987; Wiley *et al.,* 1997). These taxa were the main component missing from most of the artificial substrate samples where velocity was lower.

The combination of both sampling techniques suggest a longitudinal decrease in taxonomic diversity and pollution sensitive taxa. This decrease is influenced by a reduction in periphyton biomass and an increase in suspensoids associated with pastoral land use. The influence of macrohabitat is evident from differences between sampling techniques, and sites three and five. This influence should be considered when assessing the effect of physicochemistry on changes in community structure and choosing macroinvertebrate sampling techniques for studies of water quality.

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# Chapter Four

# Morphological characteristics of the New Zealand freshwater mussel *Hyridella menziesi* in the Whanganui River and its catchment

## ABSTRACT

The influence of habitat and environmental characteristics on the morphology of the freshwater mussel *Hyridella menziesi* were investigated in the Whanganui River catchment, North Island, New Zealand. Of the six sites sampled, one had mussels with distinctive shell morphology. Physicochemical and habitat variation was not related to shell morphology but geographical isolation from tectonic movement is one possibility. Habitat characteristics were the most important determinant of distribution. Body weight and annuli number showed greater variation in these populations from lotic habitats, than in previous reports on lentic populations.

**Keywords:** freshwater mussel, *Hyridella menziesi*, shell morphology, annuli, habitat, environmental variation, lotic.

## INTRODUCTION

Freshwater mussels have been used extensively in bio-accumulation studies investigating immediate and historical levels of deleterious substances such as heavy metals (e.g., Imlay, 1982; Millington and Walker, 1983; Carell *et al.,* 1987; Åberg *et al.,* 1995; Nyström *et al.,* 1996) and organic compounds (e.g., Kauss and Hamdy, 1985; Storey and Edward, 1989). Freshwater mussels display a high degree of morphological variability in response to a range of environmental variables (Tevesz and Carter, 1980). This variability can include changes in shell dimensions, shell and body weight, surface erosion and shell abnormalities (e.g., Cvancara, 1970; Green, 1972; Harmen, 1972; Hinch *et al.,* 1986; Roper and Hickey, 1994; Müller and Patzner, 1996; Kesler and Downing, 1997).

Physical characteristics such as dissolved oxygen, turbulence and temperature as well as general trophic dynamics are markedly dissimilar between lotic and lentic habitats (Nobes, 1980). The majority of morphological studies involving freshwater mussels overseas have been conducted in lentic environments. Studies of *Hyridella menziesi*, one of four New Zealand freshwater mussel species (Winterbourn, 1973), have been conducted in lakes or impounded rivers (e.g., Grimmond, 1968; Forsyth and McCallum, 1978; James, 1985; James 1989; Roper and Hickey, 1994). The aim of this study was to investigate the influence of environmental conditions and habitat type on lotic populations of *Hyridella menziesi* in the Whanganui River catchment.

## STUDY SITES

Mussel samples were collected between spring 1996 and summer/autumn 1997 from six sites within the Whanganui River catchment representing six geographic populations (Fig. 4.1). The area sampled for mussels varied greatly between sites due to major differences in mussel density and habitat type. Because of this sample sizes among the six sites varied from 11 and 71 mussels.

Table 4.1. Site characteristics and mussel sample sizes from six sample sites in the Whanganui River catchment taken between 1996 and 1997.

| Site | Width (m) | Depth (cm) | Mussel sample size | Area sampled (m) | Substrate characteristics | Altitude (m) |
|---|---|---|---|---|---|---|
| Pipiriki | 28 | 230 | 23 | 8×20 | sediment and large boulder | 60 |
| Jerusalem | 25 | 140 | 11 | 7×15 | sediment and cobble/boulder | 40 |
| Lismore | 1 | 38 | 40 | 1×1 | sand/silt | 40 |
| Nixons | 1.9 | 9-30 | 31 | 280 | silt/soft mud | < 20 |
| Hapurua | 3.1 | 15-30 | 19 | 180 | sand/silt | 180 |
| Kakahi | 6 | 30-60 | 71 | 1×1 | sand/silt | 280 |

Both sites at Pipiriki and Jerusalem were situated in the main Whanganui River channel by the townships of respective names. At Pipiriki, mussels were found in pockets of sediment between large stable boulders in groups of one to three individuals. Habitat and environmental characteristics 10 kilometres downstream at Jerusalem were similar to that of Pipiriki with slightly higher levels of deposited sediment.

At Lismore geology consisted of unstable Quaternary sands and silts that form the predominant substrate type for streams in this area. Mussels were buried with none visible at the substrate surface. Catchment land use comprised of exotic forestry and dairy/sheep farming. No current flow was apparent.

The site at Nixons Creek was located in suburban Wanganui with a catchment of low gradient pasture land. Mussels occurred in groups of between 1 to 10 individuals. In many areas urban vegetation and structures provided shelter from terrestrial predation and strong current with plant roots often improving the cohesion of soft consolidated mud where mussels were commonly situated.

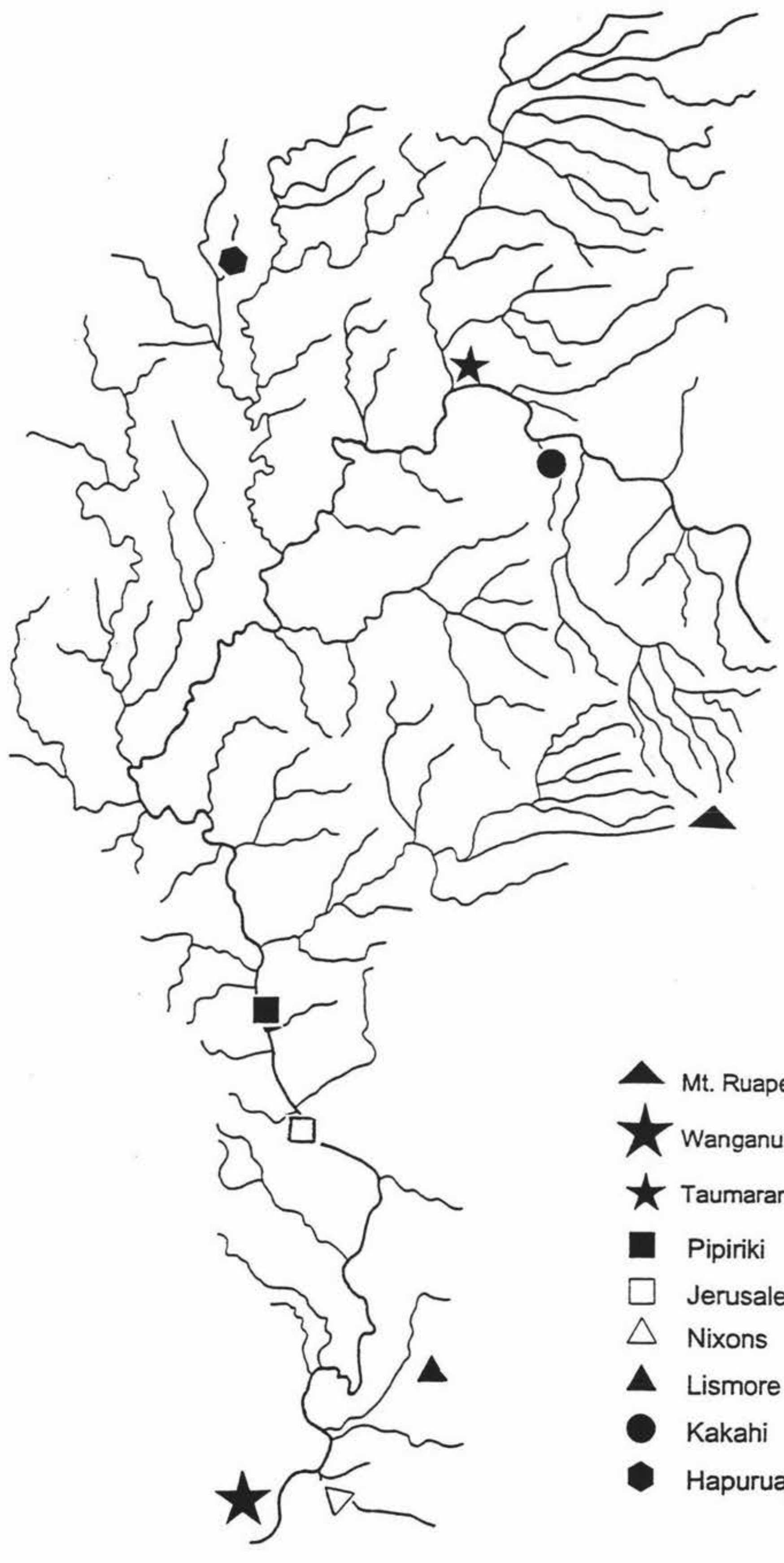


Figure 4.1. Location of the six sites where samples of the freshwater mussel *Hyridella menziesi* were collected during 1996 and 1997.

Hapurua stream is located in the north western boundary of the Whanganui River catchment and is similar geologically to the Lismore and Nixons sites. Current speed was typical of a low gradient stream. The banks were heavily eroded from cattle access which contributed to the consolidated sand/silt/mud substrate. Mussels were located sporadically in small groups or found individually.

Kakahi stream is situated 14 km east of Taumaranui with a catchment developed predominantly for pastoral agriculture. Mussels were collected from the banks of a large pool formed by a stream bend.

## METHODS

Mussel densities varied markedly between populations so quantitative estimates of density were not practical due to the relative scarcity of mussels at some sites. Sites were often located and searched on the basis of anecdotal evidence that suggested mussel presence. This evidence was acquired through feedback from local newspaper articles, and consultation with residents and local iwi. A total of over 50 localities were searched for *Hyridella menziesi* between December 1995 and November 1997. Mussels were located by visual contact and hand searches of the substrate which has been found effective in other studies (Cvancara, 1970; Roper and Hickey, 1994).

Sites in the main Whanganui river channel were both searched and sampled using snorkelling equipment. Water levels were low enough at other sites to collect mussels without diving. Physicochemical measurements and invertebrate samples were collected at each site. Macroinvertebrates were sampled by taking four replicate samples collected at each site using a 250 μm mesh, 0.1 $m^2$ Surber sampler and preserved in 70% alcohol. Samples were sieved using a 250 μm Endecott sieve and animals identified and enumerated with a 40× microscope using available keys (McFarlane, 1951; Winterbourn, 1973; Chapman and Lewis, 1976; Cowley, 1978; Winterbourn and Gregson, 1989; Towns and Peters, 1996). If taxa could not be identified to species they were assigned to apparent morphospecies.

Invertebrate samples were taken from riffles instead of the slow flowing areas that mussels commonly inhabited. This was to gain an understanding of water quality and the wider invertebrate fauna rather than to sample invertebrates characteristic to habitats similar to those occupied by *Hyridella menziesi*. Physicochemical measurements recorded consisted of conductivity, width, depth and pH.

All individuals used for morphometric analysis were greater than 45 mm long to avoid changes in morphology associated with allometric growth (Roper and Hickey, 1994). Length was measured as the maximum antero-posterior dimension of the shell, height as the maximum dorso-ventral dimension measured at right angles to length , and depth as the maximum transverse dimension of the shell with both valves in the normal closed position. Dry flesh weight was determined after careful removal of the body from the shell and air drying in an incubator at 70°C for 48 hours. Both valves were weighed after drying in the same conditions. Cavity volume was measured by weighing the amount of water that filled one valve and multiplying this by 2. Shell length, width, depth and weight were measured with vernier callipers. The left valve of each shell was examined for erosion and the area within which eroded periostracum occurred was recorded according to categories modified from Roper and Hickey (1994). These categories ranged from A to I. This technique was used to gain a rank measure of erosion between mussels rather than a quantative measure of eroded area. Attempts were made to age mussels by counting shell annuli using the preparation technique of Grimmond (1968).

Differences between morphometric variables were examined using one-way analysis of variance (ANOVA) using the SYSTAT statistical package (SYSTAT, 1996) and Pearson correlations were conducted between physicochemical and biological stream measures to investigate relationships between morphological and environmental characteristics. Data was transformed where appropriate using $\log_{10}(x+1)$ transformations to remove heterogeneity of variance. Two physical condition indices were calculated (e.g., Roper and Hickey, 1994).

$$CI_{\text{flesh: shellweight}} = \frac{\text{dry flesh weight (mg)}}{\text{shell weight (g)}}$$

$$CI_{\text{flesh: shellvolume}} = \frac{\text{dry flesh weight (mg)}}{\text{shell volume (ml)}}$$

Canonical discriminate analysis (SYSTAT, 1996) on $\log_{10}(x+1)$ transformed values of length, width, depth and shell weight was used to examine morphological differences between mussel populations. The MCI (Stark, 1985) was calculated for each site from macroinvertebrate samples (refer to Chapter Two for formula).

## RESULTS

Mussel densities varied greatly between sites and small sample sizes were directly related to low mussel densities. Average measures of length, width and depth suggested that variation among most sites in these three characteristics are the result of scale differences and not changes in the relative proportion of each characteristic. Mussels from Hapurua were the one exception where mean and variance of shell depth were markedly higher than any other site (Fig. 4.2).

Shell length distributions were comparable in width for all six geographic populations (Fig. 4.3). Mean shell weight showed a similar trend to mean length, width and height but this trend was not proportional between sites (Fig. 4.4a). Average shell weights were similar and heaviest in the Nixons, Jerusalem and Pipiriki mussel populations with shells at the other three sites, especially at Hapurua, weighing less (Fig. 4.4a). The largest mean shell cavity volume was found for mussels from Jerusalem (Fig. 4.4b). Shells from Hapurua had the lowest volume (despite their high depth) due to their overall small size in other dimensions (Fig. 4.4b).

Mussel populations at Kakahi and Pipiriki had the lowest values for both condition indices indicated by steeper slopes to the top left of Fig. 4.5 and Fig. 4.6 and both indices were highly variable among populations ($F_{5,189} = 20.6$, $P = 0.00$ for flesh weight/cavity volume and $F_{5,189} = 13.6$, $P = 0.00$ for flesh weight/shell weight). Least significance difference tests indicate that Pipiriki and Kakahi are responsible for the differences found by ANOVA for both condition indices. While index values were higher in the other four populations they varied depending on which index was used and were not significantly different ($F_{5,189} = 0.66$, $P = 0.15$). For example mussels at Lismore had high flesh weight to shell weight ratios but were lower when substituting shell volume for shell weight.

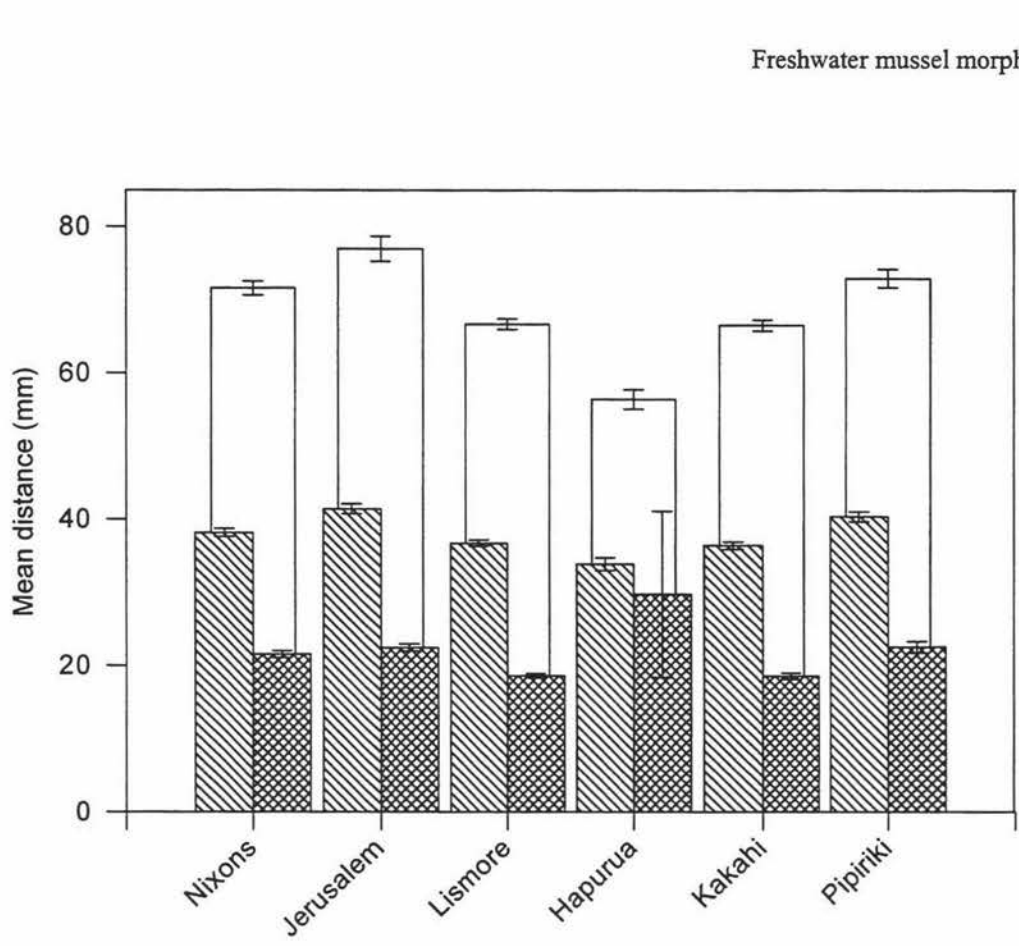


Figure 4.2. Mean (± 1 S.E.) measurements of mussel shell length (no hatching), width (diagonal hatching), and depth (reticulated hatching) from six sites in the Whanganui River catchment.

Mean shell erosion differed significantly among populations ($F_{5,189} = 19.7$, $P = 0.00$) being highest in shells from the main river sites at Jerusalem and Pipiriki, and lowest at Lismore (Fig. 4.7). Erosion varied widely among shells from Kakahi and this may result from a large sample size. Erosion was not correlated to any associated indicators of size/age like length, width and shell weight. Channel width was significantly correlated with shell erosion ($r = 0.95$, $P = 0.00$) but not with shell weight ($r = 0.64$, $P = 0.17$). Demarcation of annuli varied between shells and overall was not clear enough to provide age estimates.

MCI values ranged from 85 to 120 (Table 4.1). MCI scores of 80 or less are considered polluted while scores of 100 or more are considered healthy (Stark, 1985). Conductivity and pH ranged between 86 and 610 $\mu Scm^{-2}$ (Kakahi to Nixons), and 7.2 to 7.8 (Pipiriki to Lismore), respectively. It should be noted that chemical measures were not all recorded at the same time.

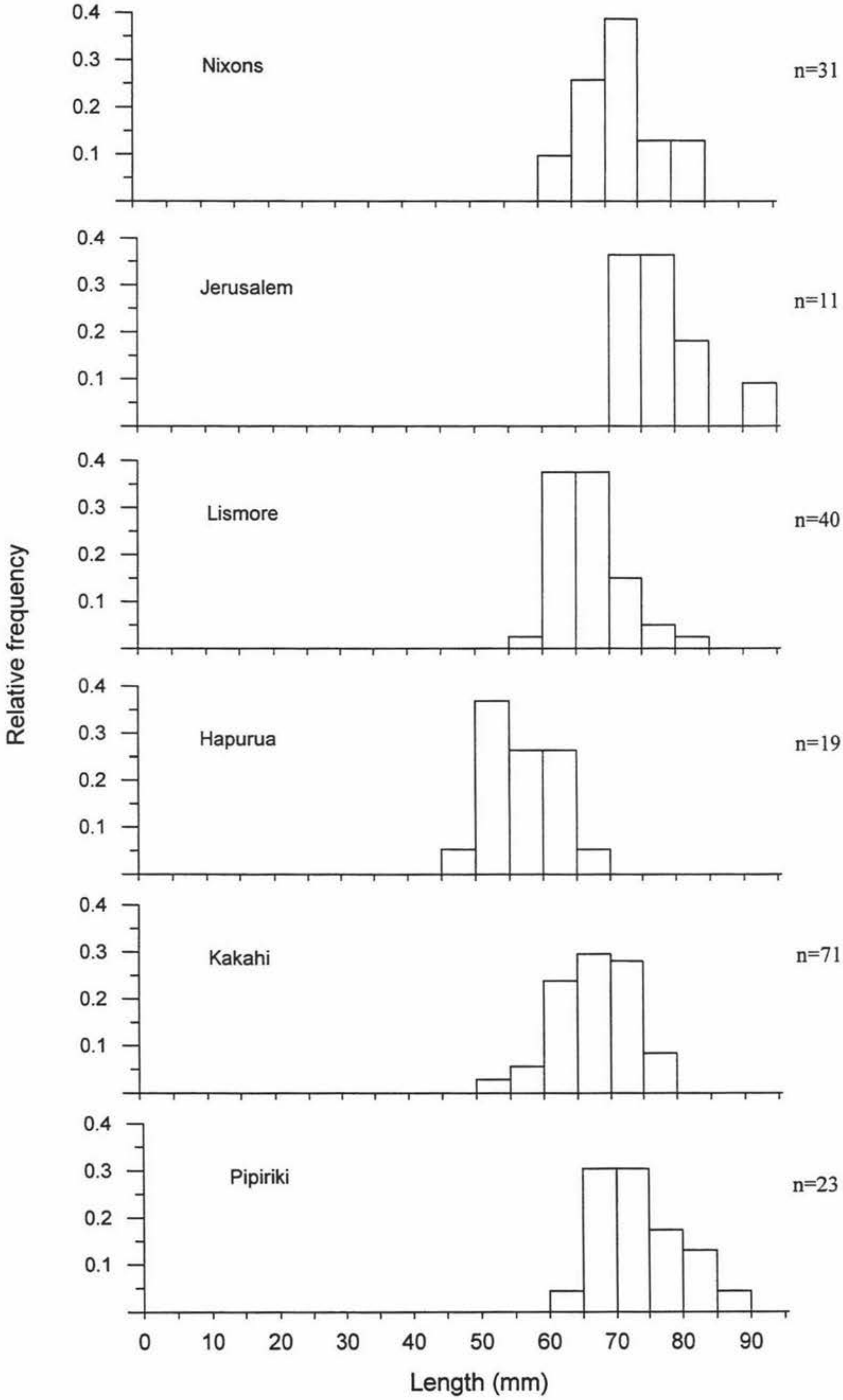


Figure 4.3. Relative frequencies of size classes for mussel shell length from six sites in the Whanganui River catchment.

Canonical discriminant analysis found significant differences between populations based on length, width, depth and shell weight ($F_{5,189} = 10.7$, $P = 0.00$). Plotting factor 1 and 2 canonical score means and values show that mussels from Hapurua are distinct from the other groups with lower factor 1 scores (Fig. 4.8). Factor 1 explained 81% of variation in the data and the classification of Hapurua mussels was of 95% certainty. There was little separation of the other sites by either factor 1 or factor 2, but Lismore and Kakahi do appear to have greater factor 2 scores. Lismore and Kakahi had almost identical means. Factor 2 was responsible for 13% of the variation in the data.

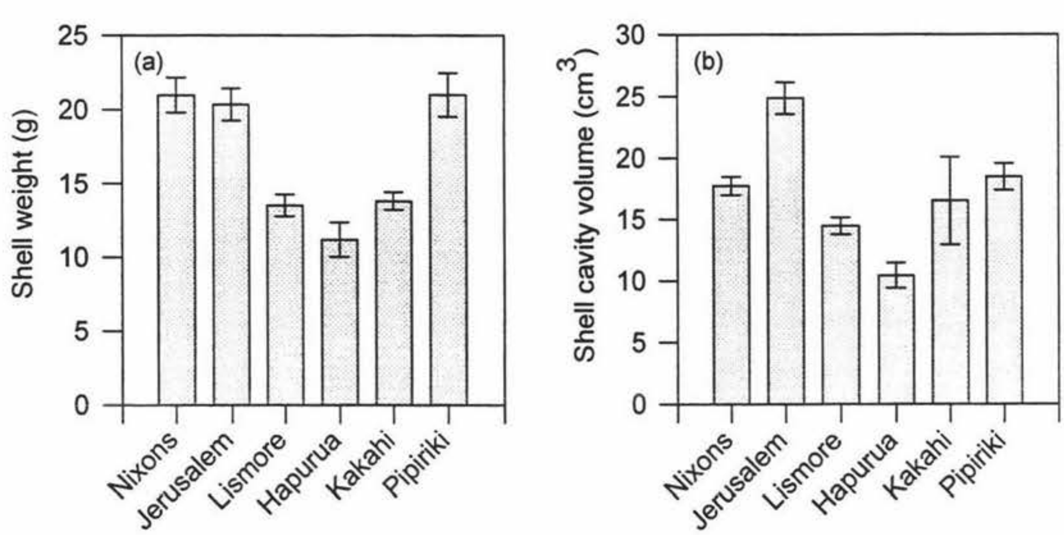


Figure 4.4. Mean (± 1 S.E.) (a) shell weight and (b) shell cavity volume of mussels from six sites in the Whanganui River catchment.

Shell length was the most important influence on factor 1 with shell weight being of primary importance along factor 2. Overall, length was the most important morphological characteristic (F to enter = 26.8). Shell depth had the lowest F to enter value (3.95) but was second to highest (F to enter =10.4) when using a stepwise discrimination. This is due to a strong allometric correlation between shell weight and length ($r = 0.89$, $P = 0.02$).

Levels of pH were negatively correlated with factor 1 scores ($r = -0.7$, $P = 0.04$) with values ranging from 7.2 to 7.8. MCI and conductivity used to estimate water quality were not significantly correlated with mussel dimensions or factor 1 scores.

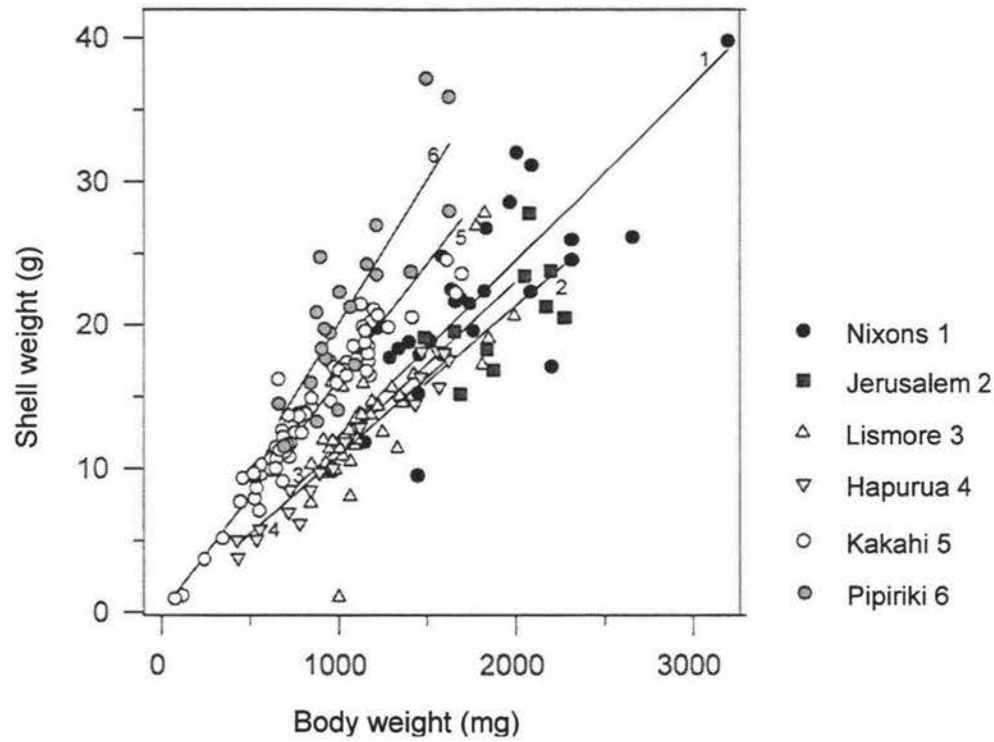


Figure 4.5. Shell weight vs. body weight for mussels from six sites in the Whanganui River catchment. Regression lines are shown.

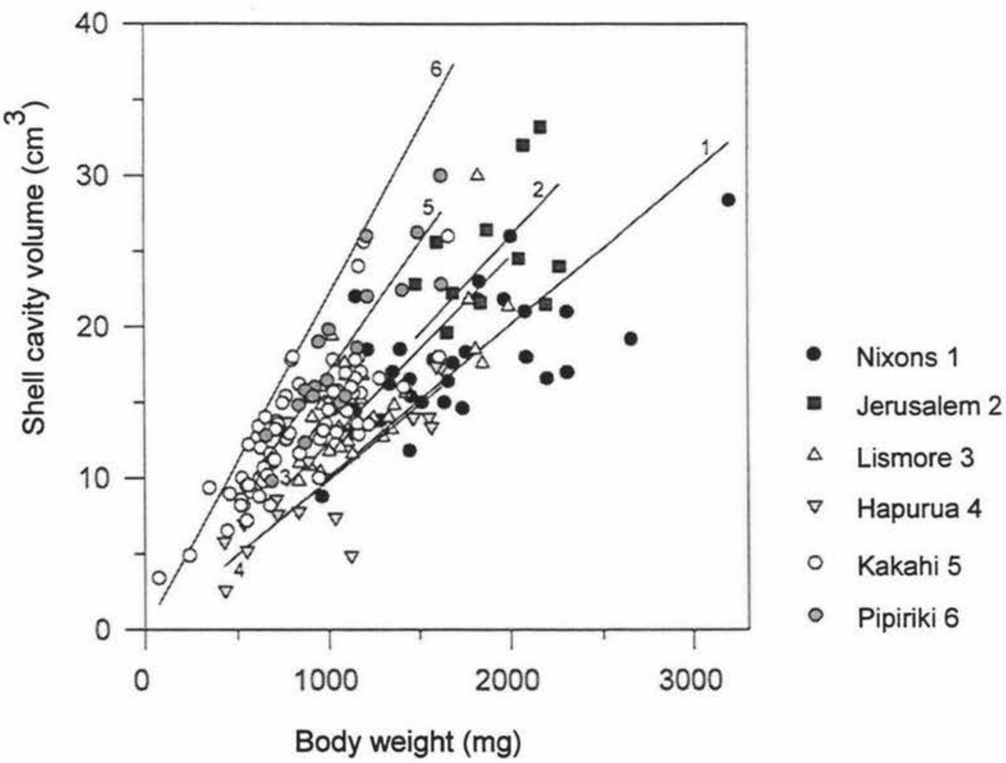


Figure 4.6. Shell cavity volume vs. body weight for mussels from six sites in the Whanganui River catchment. Regression lines are shown.

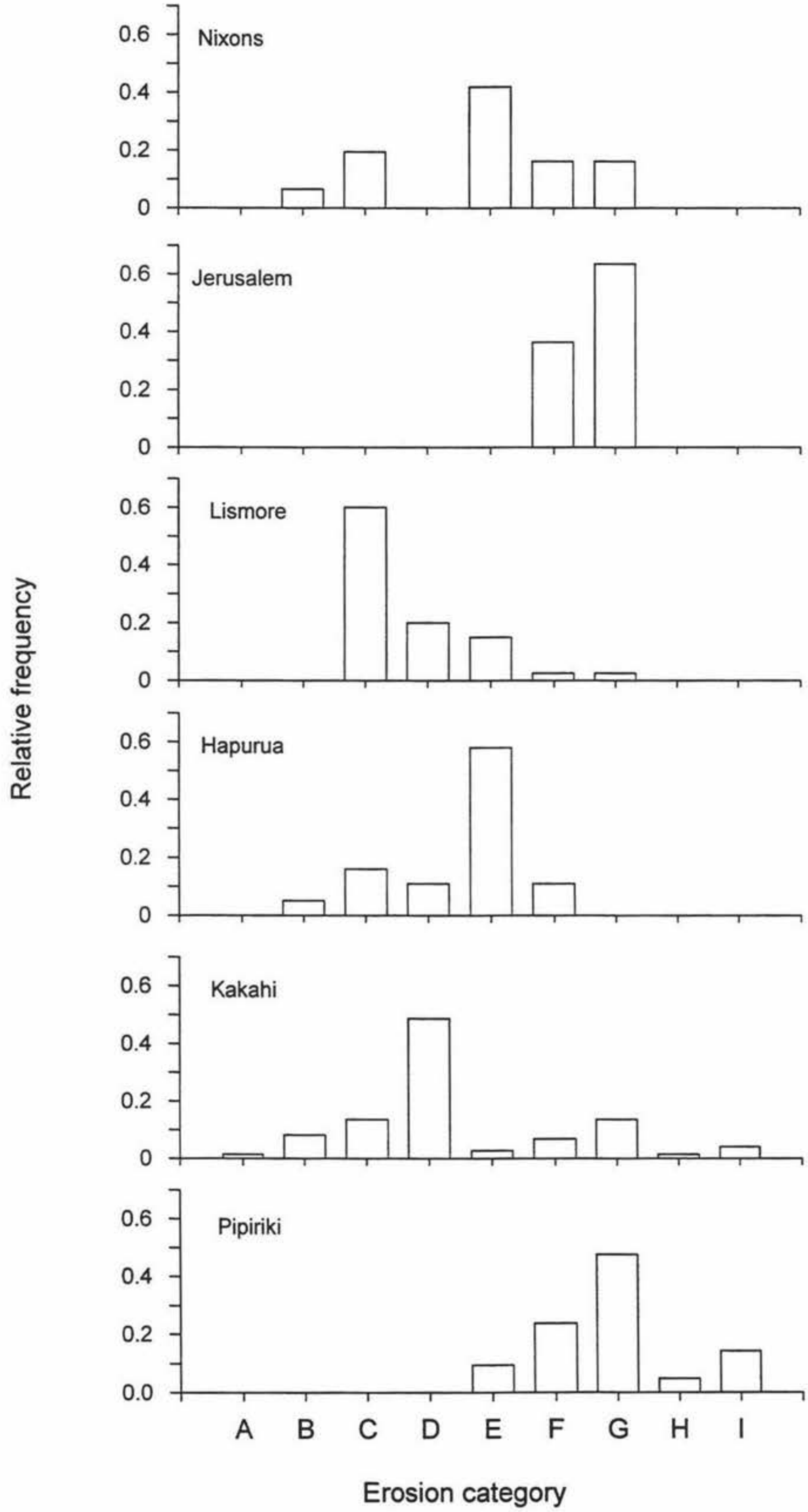


Figure 4.7. Relative frequencies of shell erosion from six sites in the Whanganui River catchment. Letters A to I represent increasing erosion levels. A < 1%; B 1-2%; C 2-5%; D 5-12%; E 12-20%; F 20-30%; G 30-40%; H 40-50%; I > 50%.

Table 4.2. Physicochemical and biotic measures of water quality from the six sample sites in the Whanganui River catchment taken between 1996 and 1997.

| Site | Width (m) | Depth (cm) | pH | Conductivity ($\mu Scm^{-2}$) | MCI |
|---|---|---|---|---|---|
| Nixons | 1.9 | 9-30 | 7.3 | 610 | 85 |
| Jerusalem | 26 | 150 | 7.3 | 140 | 108 |
| Lismore | 1 | 38 | 7.8 | 359 | 85 |
| Hapurua | 3.1 | 15-30 | 7.5 | 137 | 102 |
| Kakahi | 6 | 30-60 | 7.3 | 86 | 121 |
| Pipiriki | 27 | 230 | 7.2 | 222 | 100 |

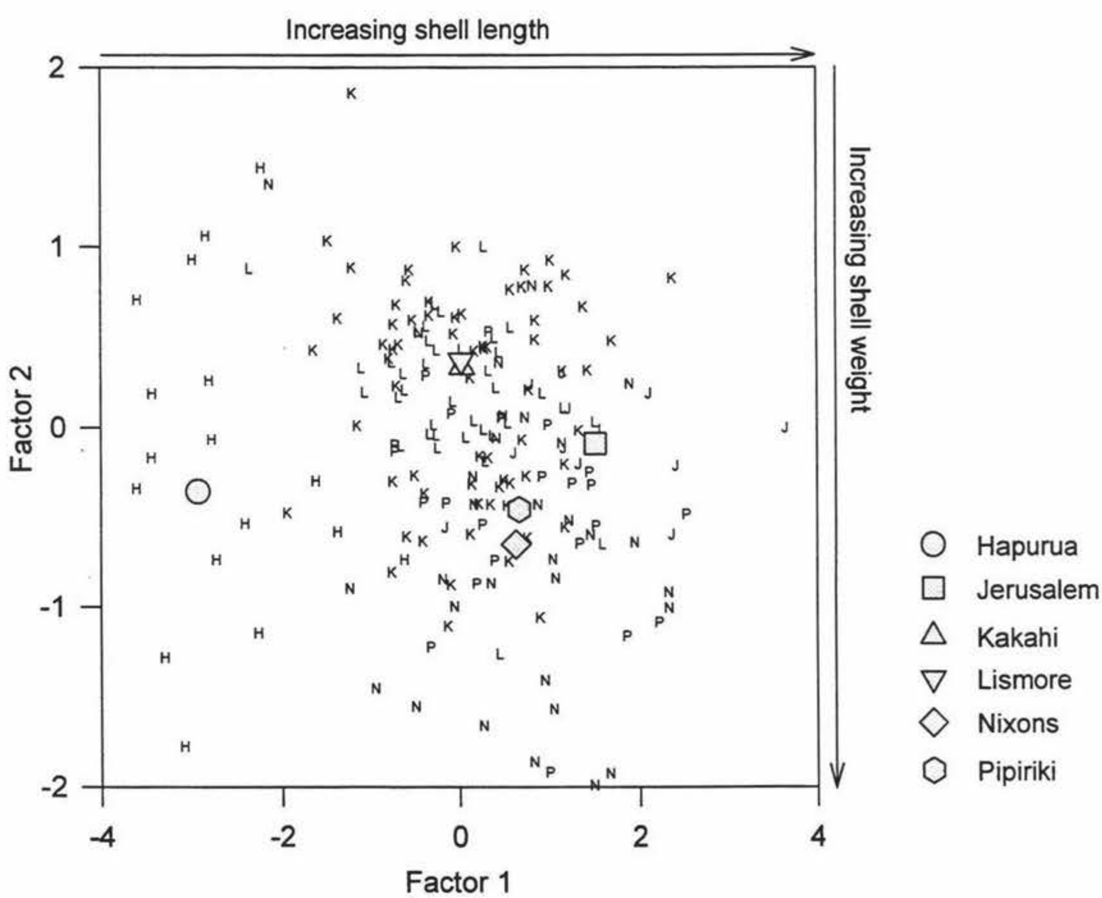


Figure 4.8. Plot of Factor 1 and factor 2 of the canonical discriminant analysis based on length, width, depth and shell weight. Means are shown by symbols and individual mussels are indicated by the first letter of their respective site.

## DISCUSSION

The density and spatial patterns of mussels varied widely between sites but overall shape and structure was similar (excluding scale differences) for all but the Hapurua population. Amounts of shell erosion and levels of condition varied significantly among populations. However, morphology was not related to any chemical or biological measures of water quality.

Population density appears to be regulated primarily by temporal and spatial habitat stability (e.g., Cvancara, 1970) combined with the presence of fine substrate. A lack of resilience compared to other stream invertebrates means they are reliant on suitably sheltered or stable habitat (Cvancara, 1970) such as areas of lower velocity (Salmon and Green, 1983). From the anecdotal information collected (much of which dated back 50 years), it appears that populations of *Hyridella menziesi* are becoming scarcer in the Whanganui River. Possible reasons for this include increased silt loading, agrochemical pollution and lower water quality. I suggest that reduced habitat stability over the history of agricultural development in the Whanganui River catchment may have lead to reduced abundances of *Hyridella menziesi*. The effects of pastoral agriculture on aquatic ecosystems are discussed further in Chapter Two.

Overall, shell proportions were similar between sites but differed in mean size. This suggests mean age differences between sites deduced from shell length can relate well to accurate annuli counts (James, 1985) but the possibility of morphological clines (Dell, 1952) between sites hinders a comparison of this kind. It is difficult to decide what constitutes a 'population' as all sites occur in the same catchment and are not isolated except by distance. However, the mussels from Hapurua stream proved to be distinctive in that they were relatively deep in comparison to their other dimensions. Water quality was not correlated with any measures of mussel morphology and a MCI score of 2 assigned to *Hyridella menziesi* indicates its tolerance of poor water quality. Therefore it is unlikely that water quality, chemistry or habitat are responsible for the distinctive nature of mussels from Hapurua.

Hapurua stream is situated at the north western point of the catchment where streams flow along low gradients much closer to the coast north of Taranaki than to the outlet of the Whanganui River, well to the south. A geological map of the area (Grange, 1927) shows a fault line running between the headwaters of the Mokau and Whanganui Rivers, 8 km from where mussels were collected. Land on the Mokau side of the fault has been elevated simultaneously as the Whanganui side has dropped, potentially cutting off the Mokau headwaters and diverting them into the Whanganui River (Neal pers. comm.). Hence mussels from Hapurua may have originated from the Mokau catchment explaining their dissimilarity to other mussels in the Whanganui. Clearly, examination of more mussel populations in both the Mokau and Whanganui catchments needs to be undertaken to test this hypothesis.

Body mass and condition varied widely between mussel populations. Starvation for a period of 31 days has been shown to produce 23% to 34% losses in body mass of *Hyridella menziesi* (Roper and Hickey, 1995). This loss can presumably be regained in favourable conditions suggesting that body mass is in a continual state of flux, particularly in highly variable environments. Condition indices then are only representative of a single point in time being potentially confounded by the spatial and temporal factors of the sampling regime. Certainly, applying indices calculated for one population to another is risky due to the likelihood of morphological clines (Dell, 1952; Cvancara, 1970; Green, 1972). Mussels from Pipiriki and Kakahi were sampled much earlier in the study (spring, 1996) whereas others were sampled later (summer/autumn 1997). Low condition for Pipiriki and Kakahi mussels could be the result of less favourable growth conditions over winter.

In older mussels erosion may often be more pronounced (Hinch and Green, 1987). However, no significant correlations were found between mussel size (i.e., length/width/depth/shell weight) and erosion. It has been suggested that chemical causes can influence shell etching (Åberg *et al.*, 1995). Despite the relationship between pH and factor 1 of the canonical descriminant analysis, the lack of a significant direct correlation between erosion and pH, and the small pH range among sites suggests water chemistry is not important. If larger streams and rivers are considered to have greater levels of turbulence, particularly in spate, turbulence would appear to be affecting shell erosion (e.g., Hinch and Green, 1987; Green *et al.,* 1989).

Attempts were made in this study to count shell annuli but estimates were not accurate enough for statistical comparison. Difficulties in determining true annuli from false have been noted elsewhere (e.g., Coon *et al.,* 1977; Carell *et al.,* 1987; Green *et al.,* 1989; Kesler and Downing, 1997). The technique used was that of Grimmond (1968) and has been applied successfully in other studies of *Hyridella menziesi* (e.g., Grimmond, 1968; James, 1985; James, 1989; Roper and Hickey, 1994). However these involved mussels in lentic habitats (except for the study by Roper and Hickey (1994) in impounded reaches of the Waikato River). Errors occur in annuli counts when difficulty arises in distinguishing dark winter lines from growth disturbance lines (Carell *et al.*, 1987). Variation in physical factors such as rapid temperature fluctuations, storm disturbance and environmental stress, all of which are more pronounced in lotic environments (Salmon and Green, 1983), make for more ambiguous annuli records (Shuster, 1951; Stansbery, 1967; Imlay, 1982; Green *et al.*, 1989). For example, mussels can migrate vertically through the substrate in response to adverse environmental conditions (Amyot and Downing, 1997). In this study Lismore mussels became endobenthic in response to summer drought. This response will lead to growth disturbance annuli unrelated to winter growth interruption. As well as the timing of variation, the magnitude of variation may be important. Finally, other lentic studies involving the assessment of winter growth annuli have been conducted predominantly in temperate to subartic regions of the Northern Hemisphere (e.g., Bailey and Green, 1987; Carell *et al.*, 1987) where seasonal changes are more pronounced leading to greater definition in shell annuli (McCuaig and Green, 1983).

In this study shell proportions were similar among all sites except for Hapurua. While habitat stability is likely to be responsible for the degree of shell erosion and population densities neither physical or chemical factors appear responsible for the distinctive morphology of Hapurua shells. Further work needs to evaluate the possibility of biogeographical changes in this part of the catchment. Environmental variability within most stream ecosystems means condition indices may be misleading as single spot measures of body mass used to calculate these are in constant flux. This variability is also apparent in shell annuli making ageing techniques using this method unreliable in this study.

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