The influence of hydrology and landscape on stream invertebrate communities of the Whanganui Catchment, New Zealand

A thesis presented in partial fulfilment
of the requirements for the degree of
Masters of Philosophy in Ecology
at
Massey University
Palmerston North
New Zealand

Dawn Lemke

2002

Abstract

This study examines the effect of the hydrological regime and landscape characteristics on benthic stream macroinvertebrate communities in the Whanganui Catchment, New Zealand, using a variety of statistical techniques. A total of 36 hydrological variables were considered. Thirteen of these were calculated from long-term flow and rainfall records, and interpolated using geographic information systems (GIS) to cover the area of interest. Ten hydrological variables were derived from topographical maps though GIS, with a further 13 hydrological variables determined from rainfall information relating to the timing since the last event. Catchment and riparian landscape variables (land cover, geology and soil type) were derived from New Zealand's land classification database and land resource inventory using GIS. Community structure was measured using an ordination, species richness, total number of animals and the densities of 22 common taxa. Biological data was also quantified with a variety of diversity and water quality indices including Simpson's, Margalef's and Macroinvertebrate Community Index.

Individual links between biological and all environmental data were explored through Pearson's correlations. Multiple regression was used to examine the combination of the environmental variables that were best in determining individual characteristics of community structure. Canonical correlation was utilized to assess overall concurrent patterns between landscape and biotic data. Stepwise logistic regression and classification trees were used to explore occurrence of the 22 selected taxa in relation to environmental variables. Of the modelling techniques assessed for prediction of taxa occurrence classification trees gave as good or better predictions than the other models and tended to produce simpler models, suggesting that it is probably a better modelling technique for this data.

Of the environmental variables, FRE3 (number of flood events per year over 3 times the medium flow) was the best individual predictor of community structure, showing the greatest number of links and strongest relationships with the biotic variables. The other hydrological variables of river size and specific discharge also had numerous individual correlations, and as they are easier to calculate they may be more appropriate for use in ecological studies. Time since the last event was another important component in

determining invertebrate community structure. Hydrological characteristics dominated the explanatory variables in many of the models representing water quality and diversity indices, suggesting that indices that measure water quality and diversity may only be comparable over areas with similar hydrological regimes.

Landscape variables, as measured by this study, were also shown to influence invertebrate community structure but to a lesser extent than hydrology. All indices and ordinations show significant multiple regressions with landscape variables with 12 out of the 22 taxa models being highly significant. Canonical correlations of landscape variables showed catchment scale variables to be more predictive of community structure than riparian scale variables. When separating landscape variables into land cover and geology no clear pattern of dominance was shown.

Explanation of text

This thesis is a combination of 4 papers. This has resulted in some repetition in introductions, site description and methods between chapters. Chapter 2 is in the process of being submitted to Journal of the North American Benthological Society and Chapters 3, 4 and 5 to New Zealand Journal of Marine and Freshwater. Formatting of text follows the guidelines set out by the Journal of the North American Benthological Society.

Numbering of figures and tables is restarted for each chapter.

Raw data

Data for this study has been sourced from numerous individuals and organisations and as such raw data is not supplied in the Appendices. If you with to obtain original data please contact Dawn Lemke [redacted] or Russell Death (Ecology Department, Massey University, R.G.Death@massey.ac.nz).

Acknowledgments

I am grateful for the support and encouragement of all those that I have interacted with over the length of my Masters. In particular to Russell Death and Derek Williams. As my supervisors they have given me invaluable help particularly in the technical applications and writing processes. Russell for his willingness to take on a hydrologist and enthusing me with ecology, for translating all the technical jargon and helping me develop skills to express my thoughts and ideas. Derek for introducing me to the wonderful world of GIS and giving me confidence and encouragement. Without their involvement none of this would have been possible. I would also like to thank Siva Ganesh who first introduced me to multivariate statistics, then made them make sense.

I am very appreciative of Genesis Power, horizons.mw (Marianne Watson and Sean Hodges), Department of Conservation (Rosemary Miller and Theo Stephens), National Institute of Water and Atmosphere (Mike Scarsbrook), Kimberley Dunning, Johnny Horrox, Chris Fowles (Taranaki Regional Council) and John Stark (Cawthron Institute), all of whom gave data that made this project possible.

My family, Davie, Bruno and Meredith, I would like to thank for their support, both moral and financial that has allowed me to finish. To Martin Doyle for the original encouragement that got me started on the whole University thing, it was a good move, thanks. All the others at TDC, Envirolink and Landcare who introduced me to the world of water and science, and to the other Nelsonians, in particular, Bernard, Jacob, Sam, Max, Moritz who have looked after and encouraged me over the years, you've all given me so much, thanks.

It is only through the help offered by many students and staff at Massey University that this work has been an enjoyable challenging experience. In particular Rachael Summers, Mike Touhy, Ian Henderson and Richard Heerdegen who have answered many of my small questions. Pepe, Steve, Ang.G., Mike and all the members of the Stream Team and Community Lab, past and present, for all the encouragement, words of wisdom and general fun we've had, thanks. I have great appreciation of the Parations for feeding me, discussing the possibility of flight and generally accepting an 'ecologist' into their midst. To Wendy for showing me it's possible, Ang.M. for struggling through it with me, Carlos for the entertainment and support you have given me, and Penny and Matt for those excellently distracting adventures, you're the best!

Table of Contents

Abstract	ii
Explanation of Text	iv
Acknowledgments	v
Chapter 1: General Introduction	1
Chapter 2: Influence of the hydrological regime on stream invertebrate community structure in the rivers of the Whanganui Catchment, New Zealand	8
Chapter 3: Influence of landscape on stream invertebrate community structure in the rivers of the Whanganui Catchment, New Zealand	45
Chapter 4: Predicting the occurrence of stream invertebrate taxa using classification trees and logistic regression	76
Chapter 5: Effect of hydrology and landscape on densities of stream invertebrates in the rivers of the Whanganui Catchment, New Zealand	97
Chapter 6: Synthesis	116
Appendices	120

1

General Introduction

New Zealand's distinctive environment has resulted in a stream invertebrate fauna that has many unique characteristics including endemism and trophic generalist(summary in Boothroyd 2000). New Zealand is a small landmass of highly variable environments, steep mountains, gentle flood plains and extreme variation in rainfall. The stream invertebrates show high levels of endemism this is likely due to the long period of isolation from other landmasses and the wide variety of habitats available for specialisation (Boothroyd 2000). It is also these historical biogeographic events that have created the template on which current species distribution is derived. Stream invertebrates represent and important part of water resources both in there composition and densities.

There is strong interest in the use of invertebrates as biological indicators for monitoring and assessment tools in the management of water resources (e.g. Richards et al. 1997, Rothrock et al. 1998, Doledec et al. 1999, Charvat et al. 2000, Malmqvist 2002). To ensure effective use of biological communities as indicators it is important to understand the mechanisms that determine the presence and abundance of a species within the environment (Richards et al. 1997, Manel et al. 2001). Many studies have identified landscape and hydrology as the dominating factors in controlling stream invertebrate communities (e.g. Resh et al. 1988, Poff and Ward 1989, Barton 1996, Clausen and Biggs 1997, Johnson et al. 1997, Poff et al. 1997, Townsend et al. 1997, Rothrock et al. 1998, Cresser et al. 2000, Ometo et al. 2000).

Predictive modelling is one way to gain a better understanding of stream invertebrates and their environments. International exploration of predictive modelling of stream invertebrates has been widespread. Multivariate models such as RIVPACs have been developed for many countries (e.g. Moss et al 1999, Turak et al. 1999, Hawkins et al. 2000), with numerous other studies examining the relationship between invertebrate communities and landscape characteristics (e.g. Hynes 1970, Hynes 1975, Winterbourn et al. 1981,Townsend et al. 1983, Osbourne and Wiley 1988, Close and Davies-Colley 1990, Maasdam and Smith 1994, Harding and Winterbourn 1995, Richards et al. 1997, Raven et al. 1998, Ometo et al. 2000, Thompson and Townsend 2000, Church 2002). The hydrological regime has been largely ignored in predictive modelling of macroinvertebrate communities and the assessment of biotic health (Richards et al. 1997, Lammert and Allan 1999, Turak et al. 1999, Hawkins et al. 2000). However, it is an important component in determining the benthic invertebrate community composition of a river (Minshall 1988, Resh et al. 1988, Poff and Ward 1989, Clausen and Biggs 1997). Some work has been carried out on identifying links between the hydrological regime and community structure but this has normally been at small scales and experimentally or using subjective measures (e.g. Robinson and Minshall 1986, Death and Winterbourn 1995, Jowett and Biggs 1997, Bond and Downes 2000). Clausen and Biggs (1997) explored the use of long-term stream flow information from 83 sites throughout New Zealand. One disadvantage of working with such long-term flow records is the lack of suitable data for many locations. An alternative approach is to use information from long-term hydrological sites, field observations, topographical data and local knowledge to estimate or interpolate hydrological information to cover the area of interest (Smakhtin 2001). This is the approach taken in this study.

The advancement of tools in geographic information systems (GIS) now allows for the analysis of large spatial data sets with ease. Calculations of both regional and local scale variables from digital information is possible, reducing the need for full site surveys and allowing multi scale issues to be approached more efficiently (Allan and Johnson 1997). It is also through GIS that data interpolation of hydrological variables can be carried out, integrating landscape and hydrological variables into one data set.

Based on the idea that hydrology, land cover and geology are intrinsically linked to riverine biotic communities (as illustrated above), this study explores the relationship of diversity indices, water quality measures, ordination axes and 22 invertebrate taxa with environmental variables. This is done through the use of statistical analysis using hydrological, geological, land cover and invertebrate taxa data from the Whanganui Catchment of New Zealand. The aim of this project was to gain a better understanding of what environmental components strongly influence invertebrate communities.

Literature Cited

Allan, J. D., and L. B. Johnson. 1997. Catchment-scale analysis of aquatic ecosystems. Freshwater Biology 37: 107-111.

Barton, D. R. 1996. The use of percent model affinity to assess the effects of agriculture on benthic invertebrate communities in headwater streams of Southern Ontario, Canada. Freshwater Biology 36: 397-410.

Bond, N. R., and B. J. Downes. 2000. Flow-related disturbance in streams: an experimental test of the role of rock movement in reducing macroinvertebrate population densities. Australian Journal of Marine and Freshwater Research 51: 333-337.

Boothroyd, I. 2000. Biogeograpgy and biodiversity. Pages 30-53 in K. J. Collier and M. J. Winterbourn (editors). New Zealand stream invertebrates: ecology and implications for management. New Zealand Limnological Society, Christchurch.

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

Church, M. 2002. Geomorphic thresholds in riverine landscapes. Freshwater Biology 47: 541-557.

Clausen, B., and B. J. F. Biggs. 1997. Relationships between benthic biota and hydrological indices in New Zealand streams. Freshwater Biology 38: 327-342.

Close, M. E., and R. J. Davies-Colley. 1990. Baseflow water chemistry in New Zealand Rivers 2. Influence of environmental factors. New Zealand Journal of Marine and Freshwater Research 24: 343-356.

Cresser, M. S., R. Smart, M. F. Billett, G. Soulsby, C. Neal, A. Wade, S. Langan, and A. C. Edwards. 2000. Modelling water chemistry for a major Scottish river from catchment attributes. Journal of Applied Ecology 37: 171-184.

Death, R. G., and M. J. Winterbourn. 1995. Diversity patterns in stream benthic invertebrate communities - the influence of habitat stability. Ecology 76: 1446-1460.

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

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

Hawkins, C. P., R. H. Norris, J. N. Hogue, and J. W. Feminella. 2000. Development and evaluation of predictive models for measuring the biological integrity of streams. Ecological Applications 10: 1456-1477.

Hynes, H. B. N. 1970. The ecology of running waters. Liverpool University Press, Liverpool.

Hynes, H. B. N. 1975. The stream and its valley. Verhandlungen der internationalen Vereinigung fur theoretische und angewandte Limnologie 19: 1-15.

Johnson, L. B., C. Richards, G. E. Host, and J. W. Arthur. 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology 37: 193-217.

Jowett, I. G., and B. J. F. Biggs. 1997. Flood and velocity effects on periphyton and silt accumulation in two New Zealand rivers. New Zealand Journal of Marine and Freshwater Research 31: 287-300.

Lammert, M., and J. D. Allan. 1999. Assessing biotic integrity of streams: Effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. Environmental Management 23: 257-270.

Maasdam, R., and D. G. Smith. 1994. New Zealand's National Water Quality Network: Relationships between physico-chemical data and environmental factors. New Zealand Journal of Marine and Freshwater Research 28: 37-54.

Malmqvist, B. 2002. Aquatic invertebrates in riverine landscapes. Freshwater Biology 47: 679-694.

Manel, S., H. C. Williams, and S. J. Ormerod. 2001. Evaluating presence-absence models in ecology: the need to account for prevalence. Journal of Applied Ecology 38: 921-931.

Minshall, G. W. 1988. Stream ecosystem theory: a global perspective. Journal of the North American Benthological Society 7: 263-288.

Moss, D. W., J. F. Furse, M. T. Clarke, R. T. 1999. A comparison of alternative techniques for prediction of the fauna of running-water sites in Great Britain. Freshwater Biology 41: 167-181.

Ometo, J. P. H. B., L. A. Martinelli, M. V. Ballester, A. Gessner, A. V. Krusche, R. L. Victoria, and M. Williams. 2000. Effects of land use on water chemistry and macroinvertebrates in two streams of the Piracicana river basin, south-east Brazil. Freshwater Biology 44: 327-337.

Osbourne, L. L., and M. J. Wiley. 1988. Empirical relationship between land use/cover and stream water quality in an agricultural watershed. Journal of Environmental Management 26: 9-27.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47: 769-784.

Poff, N. L., and J. V. Ward. 1989. Implications of stream flow variability and predictability for lotic community structure: a regional analysis of stream flow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46: 1805-1818.

Raven, P. J., H. N.T.H., F. H. Dawson, and M. Everars. 1998. Quality assessment using river habitat survey data. Aquatic Conservation: Marine and Freshwater Ecosystems 8: 477-499.

Resh, V. H., A. V. Brown, A. P. Covich, M. E. Gurtz, H. W. Li, W. Minshall, S. R. Reice, A. L. Sheldon, B. Wallace, and R. C. Wissmar. 1988. The role of disturbance in stream ecology. Journal of the North American Benthological Society 7: 433-455.

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

Robinson, C. T., and G. W. Minshall. 1986. Effects of disturbance frequency on stream benthic community structure in relation to canopy cover. Journal of the North American Benthological Society 5: 237-248.

Rothrock, J. A., P. K. Barten, and G. L. Ingman. 1998. Land use and aquatic biointegrity in the Blackfoot River watershed, Montana. Journal of the American Water Resources Association 34: 565-581.

Smakhtin, V. U. 2001. Low flow hydrology: a review. Journal of Hydrology 240: 147-186.

Thompson, R. M., and C. R. Townsend. 2000. Is resolution the solution?: the effect of taxonomic resolution on the calculated properties of three stream food webs. Freshwater Biology 44: 413-422.

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

Townsend, C. R., A. G. Hildrew, and J. Francis. 1983. Community structure in some southern English streams: the influence of physicochemical factors. Freshwater Biology 13: 521-544.

Turak, E., L. K. Flack, R. H. Norris, J. Simpson, and N. Waddell. 1999. Assessment of river condition at a large spatial scale using predictive models. Freshwater Biology 41: 283-298.

Winterbourn, M. J., J. S. Rounick, and B. Cowie. 1981. Are New Zealand stream ecosystems really different? New Zealand Journal of Marine and Freshwater Research 15: 321-328.

2

Influence of hydrological conditions on stream invertebrate community structure in rivers of the Whanganui Catchment, New Zealand

Abstract

The relationship between large scale spatial and temporal hydrological parameters and benthic stream macroinvertebrate communities in the Whanganui Catchment of New Zealand was investigated. Thirteen hydrological variables were calculated from long-term flow and rainfall records. They were then interpolated using geographic information systems (GIS), to cover the river basin. Ten of the hydrological variables were derived from topographical maps though GIS analysis, with a further 13 hydrological variables related to the timing since the last event calculated from rainfall information. Stream invertebrate community structure was measured using an ordination, species richness, total number of animals and the densities of 22 common taxa. This biological data was also explored through diversity and water quality indices. The study showed 4 major hydrological components, that can be calculated from long-term hydrological records and topographical information (i.e. flood frequency, river size, specific discharge and timing since the last event), to be important in determining invertebrate community structure. Of the hydrological variables, FRE3 (number of flow events per year over 3 times the medium flow) had the greatest number of significant links with the biotic variables in particular the biotic indices (apart from Simpson's), ordination axis, *Deleatidium* sp., *Maoridiamesa* sp. *Potamopyrgus antipodarum*, *Psilochorema* sp. and *Zelandoperla* sp. River size and specific discharge also had numerous individual correlations and are easier to calculate, therefore they may be more appropriate for use in ecological studies than FRE3. Combining the hydrological variables to predict biotic variables through multiple regressions showed the water quality indices to have strong relationship with the hydrological conditions. This suggests that these indices may be highly influenced by the hydrological regime and as such cannot be used for water quality assessment across broad range of hydraulic environments.

Key Words: Benthic macroinvertebrate, flow regime, geographic information systems, interpolation, multivariate statistics, New Zealand, regression, Whanganui Catchment.

Introduction

Hydrological conditions are important in determining the composition of benthic invertebrate communities in a river (Statzner and Higler 1986, Minshall 1988, Resh et al. 1988, Poff and Ward 1989, Clausen and Biggs 1997, Wood et al. 2000). There are many aspects to hydrological regime, but generally it can be broken down into 5large scale components: magnitude of flow, frequency of high and low flow events, predictability of flow (occurrence during the year), duration and timing of events (Poff and Ward 1989, Angradi 1997, Poff et al. 1997, Matthaei and Townsend 2000, McCabe and Gotelli 2000). Many studies have shown high discharge events can cause population loss and change in biotic community structure (e.g. Flecker and Feifarek 1994, Clausen and Biggs 1997, Townsend et al. 1997b). Events of the same magnitude can have differing effects in individual streams (Biggs and Close 1989) with season also influencing the effect (Boulton et al. 1992). Thus, it may not be just the event, but the history of the area and the effect of the event that is significant (Death 1996b). Reynolds (2000) identified general variation in flow as being important in an ecological context, suggesting that not only high flow but also low flow events need to be considered. Each hydrological characteristic is likely to affect different components of the invertebrate fauna to differing extents, and at different times. Consequently, the hydrological regime cannot be encapsulated in a single variable when considering the link between hydrology and the benthic biota.

Many studies focus on using instantaneous field measurements (e.g. Death and Winterbourn 1995, Jowett and Biggs 1997, Townsend et al. 1997d, Doisy and Rabeni 2001, Hardison and Layzer 2001) or small-scale experimental disturbance (Malmqvist and Otto 1987, Robinson and Minshall 1986, Death 1996b, Matthaei et al. 1996, Bond and Downes 2000) to measure hydrological characteristics. Local measures of hydrological characteristics, such as Index of Stability (Pfankuch 1975), movement of painted tracer particles, local velocity, and Froude number are problematic because they have subjective components or are highly variable over small temporal or spatial scales (Downes et al. 1997, Hannaford et al. 1997, Downes et al. 1998, Death in Press). Experimental disturbances, although measurable, rarely incorporate all natural aspects at an appropriate scale (Matthaei et al. 1996). An alternative is to use long-term hydrological records (Poff and Allan 1995, Poff 1996, Clausen and Biggs 1997, Clausen and Biggs. Clausen and Biggs (1997, 2000) examined a number of hydrological variables and how they linked to characteristics of the benthic biota, focusing on sites with long-term flow records. They found strong relationships between some hydrological and biological variables; in particular, low mean flow sites had a higher total number of animals and species richness, and FRE numbers (number of flow events per year over a particular threshold) related negatively to periphyton diversity. One disadvantage of working with long-term flow records is the lack of suitable data for many locations. Within New Zealand, at best, there are only a few flow recorders per catchment.

An alternative approach is to use information from long-term hydrological sites and local knowledge to estimate or interpolate hydrological information to cover the area of interest (Smakhtin 2001). This study explores the relationship between different measures of the hydrological regime, in particular, specific discharge, river size, frequency and magnitude of low and high flow events, and timing since those events with the invertebrate communities recorded at those sites.

The hydrological regime has been largely ignored in predictive modelling of benthic macroinvertebrate communities and the assessment of biotic health (Richards et al. 1997, Lammert and Allan 1999, Turak et al. 1999, Hawkins et al. 2000), although it is well integrated in water chemistry modelling (Reckhow 1999, Bobba et al. 2000). It is a fundamental controller of the available habitat and thus should be one of the underpinning components of any riverine study. This paper explores variables commonly used in hydrology and water resource management as well as other biologically relevant variables (Gurnell et al. 2000) to define a number of calculable hydrologic characteristics that may be beneficial for use in understanding invertebrate communities. In particular, this study examines how much of the variance in characteristics of the benthic stream macroinvertebrate communities can be explained by the assessed hydrological variables in the Whanganui Catchment of New Zealand.

Study Site

The Whanganui River, located in the North Island of New Zealand, has a catchment of 7100 km^2. The catchment vegetation varies from evergreen temperate rainforest and tussock through to exotic forest and intensive farming, with the lower regions of the catchment more intensively farmed. The dominant geology types include tertiary sandstone, mudstone and siltstone with streams around Mt. Ruapehu flowing through bedrock of volcanic origin (predominately andesite). Lower streams contain laha material with andisite transported from upper catchments often composing much of the substraight. The climate varies from high rainfall areas in the upper reaches, to areas in the mid and lower catchment that experience periods of prolonged dry weather.

Methods

Biological measures

This study is based on invertebrate data from 6 different studies but with sampling and taxonomy methods comparable between studies (Department of Conservation 1988, Dunning 1998, Horrox 1998, Jowett et al. 2000, M.R. Scarsbrook National Institute of Water and Atmospheric Research unpublished data, J.D. Stark Cawthron Institute unpublished data). From these studies data from a total of 97 sites, in 3rd to 7th order rivers, was collected over 12 years, with 1 to 8 sampling occasions per site (a total of 138 sampling occasions) and each comprising at least 4 Surber samples. To standardise the data, information from only 4 Surber samples was randomly selected for each sampling occasion.

Invertebrate community structure

Variables used as measures of community structure were (Appendix 1); total number of animals, species richness and the densities of those individual genera present at over 30% of the sites (*Aoteapsyche* sp., *Aphrophila neozelandica*, *Archichauliodes diversus*, *Austroclima* sp., *Austrosimulium* sp., *Coloburiscus humeralis*, *Deleatidium* sp., Eriopterini sp., *Hydrobiosis* sp., *Maoridiamesa* sp., *Nesameletus* sp. *Neurochorema confusum*, *Olinga feredayi*, *Oxyethira albiceps*, *Potamopyrgus antipodarum*, *Psilochorema* sp., *Pycnocentria* sp., *Pycnocentrodes* sp., *Tanytarsus vespertinus*, *Zelandobius* sp., *Zelandoperla* sp. and *Zephlebia* sp.). Individual taxa and total density were natural log transformed ($\log_e (x+1)$) to improve normality of the data. To examine overall community structure, detrended correspondence analysis (DECORANA) was carried out based on raw data. This ordination technique is designed to produce an objective arrangement of communities that reflect the similarities in their taxonomic composition (McCune and Mefford 1999). DECORANA analysis is widely used in stream ecology to examine community structure (e.g. Townsend et al. 1983, Furse et al. 1984, Harding 1994, Death 1996b, Gislason et al. 2001). In this study, two DECORANA axes (DEC1 and DEC2) of the full communities (as represented by taxa densities) are used.

Biotic indices

An alternative approach to characterising communities is to use biological indices (Appendix 1). Indices calculated in this study included the water quality indices; Macroinvertebrate Community Index (MCI) (Stark 1985), Quantitative Macroinvertebrate Community Index (QMCI) (Stark 1993), Semi Quantitative Macroinvertebrate Community Index (SQMCI) (Stark 1998), %EPT (Lenat 1988) and %EPT-*O. albiceps* (%EPT-Oxy). *O. albiceps* was removed from the EPT indices as it is insensitive to changes in water quality. Higher values of the water quality indices implies low enrichment. The diversity indices of Simpson's (Simp) (Simpson 1949), Berger-Parker (BP) (Berger and Parker 1970), and Margalef's (Marg) (Clifford and Stephenson 1975) were also derived. Indices were calculated for the total pooled samples and the Insecta component of the pooled samples (IEPT-oxy, IEPT, Isimp, IBP, Imarg). Insects were considered separately to determine if variation in taxonomic resolution of the non-Insecta component in each of the studies had an impact on any of the relationships.

Hydrological measures

To determine hydrological characteristics, information from long-term(6 to 40 years) flow sites(18) and rainfall sites (15) (Fig. 1B), as well as instantaneous gaugings, regional hydrological and topographic maps were used (Tonkin and Taylor Consultant Engineers 1978, Land Information New Zealand 2000, National Institute of Water and Atmospheric Research unpublished data, horizons.mw unpublished data, Genesis Power unpublished data). For ease of explanation, hydrological variables were considered in 3 groups; interpolated variables, map variables and temporal variables. Interpolated variables were those that were calculated from point source data, map variables were taken from topographic information and temporal variables relate to timing of events (Appendix 1).

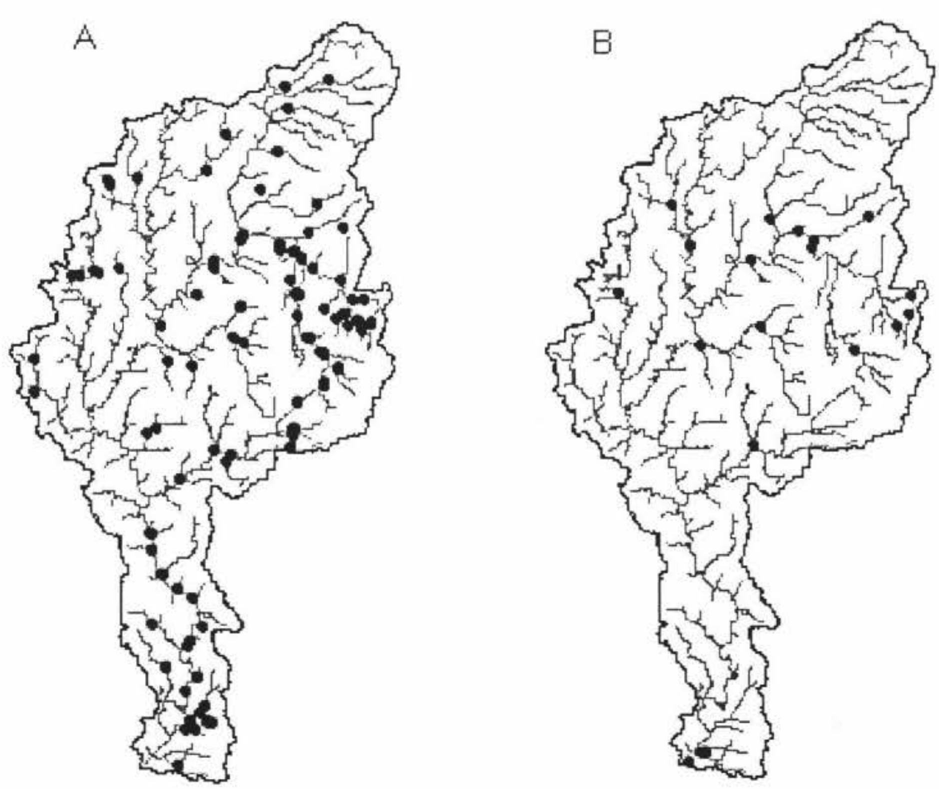


Figure 1: Whanganui Catchment, New Zealand. A – location of invertebrate sampling sites, B – location of flow recorders.

Interpolated variables

There are a multitude of hydrological variables that can be calculated from long term records many of which represent similar hydrological characteristics. A range of commonly used variables that may represent differing aspects of the hydrological regime were selected (Appendix 1). Flow parameters were calculated from long-term records using TIDEDA (Thompson and Walter 1996) and HILLTOP (Rodgers 2001), and included flow mean ($\bar{Q}$), standard deviation (SD), median (Q50), lower (LQ) and upper quartile (UQ), and for 90% (Q90) and 10% (Q10) flow exceedance. Event magnitude was calculated with frequency analysis (McKerchar et al. 1997) on 7-day low flows and instantaneous high flows using the Gumbel distribution (Gumbel 1958). The 100, 50, 20, 10, 5 year and mean annual 7 day low flow (L100, L50, L20, L10, L5 and L1), 100, 50, 20, 10, 5 year and mean annual high flow (H100, H50, H20, H10, H5 and H1) variables were estimated using this procedure. Flood (FF) and drought frequency (DF) were calculated from these estimates (100 year return period / annual return period) (Mosley 1992). The number of events over 3, 6 and 12 times the median flow were calculated to give FRE3, FRE6 and FRE12 (Clausen and Biggs 1997).

Interpolation, the next step of data processing, is time intensive, thus to reduce the amount of redundant analysis carried out, a set of relatively uncorrelated variables are selected from the 25 variables derived from long term flow records. Pearson's correlations showed high correlation between most variables, with 4 distinct groups, FRE numbers, drought frequency, flood frequency and other hydrological variables (Appendix 2). FRE numbers were correlated with each other and the low flow frequency variables, FRE3 was selected to represent this group as it more commonly used. Drought frequency and flood frequency had no strong links with any of the variables and were thus retained in further analysis. The remainder of the variables had strong inter-correlation and mean annual flow was selected to represent them as it is commonly used and the most readily available. Of this last group size of low flow and high flow events had weakest correlations with mean annual flow, thus 10 year high and low flows were also included for further analysis. This yielded 6 variables, mean annual flow, 10-year low flow, 10-year high flow, FRE3, drought frequency and flood frequency to be used in further analyses.

Information was exported to a GIS package, ArcView (Environmental Systems Research Institute 1999), where contours were drawn based on site values (Fig. 2A), knowledge of topography and rainfall (common practice in hydrological contour mapping, areas that have high rainfall will also have a high specific discharge). From the contour information (Fig. 2B), a grid was interpolated (Fig. 2C) to cover the whole catchment. Interpolation was done using inverse distance weighted interpolation (Environmental Systems Research Institute 1999) to estimate values between contours. The 12 nearest neighbours were used with no barriers and power of 2 weighting.

A number of other parameters, mean annual rainfall (MAR), 10 year 24 hour rainfall (rain24) and evapotranspiration (Evapo), were also available for the area as contour maps. These contours were digitised into ArcView, checked with current data and grids interpolated.

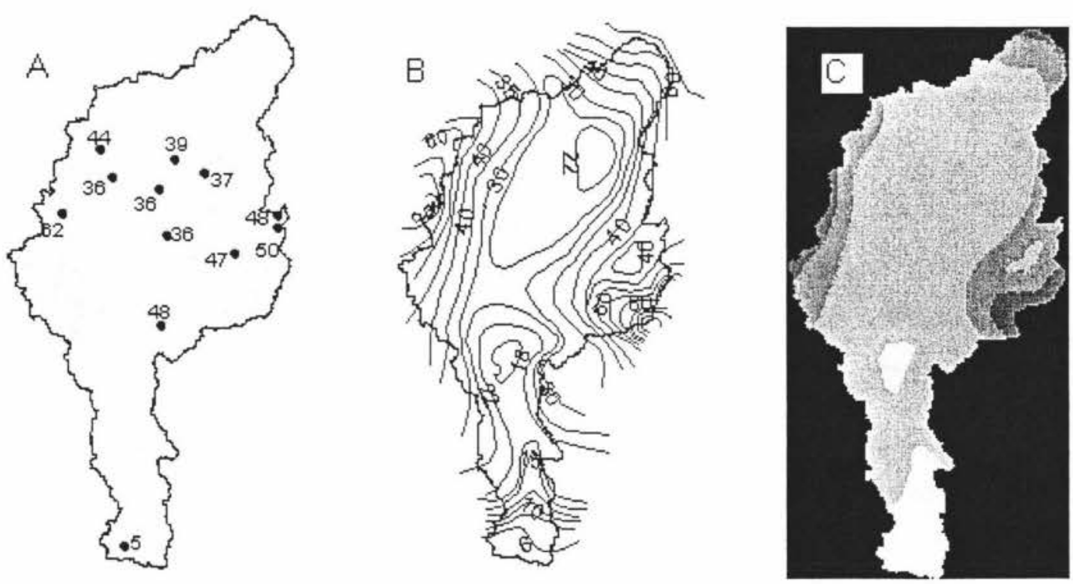


Figure 2: Interpolation process for spatial hydrological information in the Whanganui Catchment, New Zealand. A – flow recorder sites and specific discharge values, B – specific discharge contour map derived from recorder information, topography, rainfall and local knowledge, C – interpolated grid of specific discharge (light areas represent low specific discharge, dark areas high specific discharge).

Catchments were defined using a digital elevation grid created from 20 metre contours in ArcView. From the elevation grid, flow direction and flow accumulation grids were generated allowing the delineation of catchments and calculation of stream networks. A catchment polygon was created for each site sampled, and this overlaid on the hydrological grids to calculate mean and total interpolated variables.

This gave 13 variables; mean annual flow ($\bar{Q}$), 10-year low flow (L10), 10-year high flow (H10), FRE3, drought frequency (DF) and flood frequency (FF), catchment mean annual rainfall (MAR), 10 year 24 hour rainfall (Rain24) and evapotranspiration (Evapo), local mean annual rainfall (RMAR), and the flow variables standardised by the catchment area of specific mean annual flow (q), specific10-year low flow (StdL10) and specific 10-year high flow (StdH10) to be used in further analyses.

Map variables

A further 11 variables were calculated from topographic data (Appendix 1). Catchment polygons were used to determine catchment area, perimeter (peri) and catchment shape (area/peri). Slope was calculated from the elevation grid, calculated as a catchment average and site slope. The information on stream networks was used to derive the stream order at a site and relative density of streams in the catchment. Variables calculated were: area, perimeter (peri), river density (RD), slope, stream order (order), riparian slope (Rslope), catchment shape (CS), density of small order(first to third) streams (DSO), medium order(third to fifth) streams (DMO), large order(greater hand fifth) streams (DLO), and drainage density (Dd), which is the total stream density per unit area.

Temporal variables

The final set of hydrological variables was used to quantify variation in flow prior to sampling. Ten temporal parameters were derived (Appendix 1): amount of rain in the last 1, 7, 14, 21 and 28 days (T1, T7, T14, T21, T28), number of days since a total of 20, 40 and 60 mm of rain had fallen (DT20, DT40, DT60) and the number of days since 20 and 40 mm of rain had fallen in a single day (D20, D40).

Six of the rainfall recorder sites in the catchment had data for the whole period of interest. These 6 sites were used to interpolate the given temporal variables for each invertebrate sampling point. The 10 rainfall variables were calculated for the 6 rainfall sites and for any given location at any given time the data was interpolated from the closest sites to give an average. This procedure, although not exact, gave an estimate of the temporal variables at each of the sampling sites.

Statistical methods

Once all hydrological variables, indices and DECORANA axes had been calculated, data was checked for univariate and multivariate normality and variables transformed if necessary. Natural log transformations were carried out on taxa densities, total density and a number of hydrological parameters. Analysis was carried out on two biologically distinct perspectives, one representing community structure (individual taxa densities, DECORANA axes of total communities, total number of animals and species richness) and another related to anthropogenic impact, which included a number of biotic indices. Individual taxa were selected based on their presence at over 30% of the sites so valid statistical inferences could be made. Indices were also selected based on their low correlations with other biotic indices.

The aim of this study was to determine what, if any, of the hydrological variables are useful in explaining benthic invertebrate communities of river systems. The simpler and easier the method of calculating these relationships the more likely it will be used, thus usability was also assessed. Alternatively, a combination of variables are likely to have greater explanatory power, thus both simple univariate and more complex multivariate regression were used to explore the relationships.

Univariate regression involved the following two steps:

- Data was explored to determine if relationships were linear, logistic, exponential or polynomial by plotting data points. The majority of variables showed linear or near linear relationships apart from FRE3 and river size ($\bar{Q}$, HF10, LF10, Area, Perimeter and Catchment Shape), with total number of animals and species richness.
- To examine the relationship between individual hydrological variables and individual biological variables, Pearson's correlation coefficients were used for those that showed linear relationships and second order polynomial regressions for the non-linear relationships.

Two multivariate methods were used to examine the overall relationship between the hydrological regime (as measured by all the variables calculated) and individual components of the stream community. Both used principal component (PC) scores calculated independently on the three groups of hydrological variables. This was done to dimensionally reduce variables and derive a new set of less correlated variables. Principal component analysis (PCA) calculates the line that extracts the maximum amount of statistical variance from the data, thus reducing the number of variables required to represent the information. PCA is widely used in analysis of ecological data (e.g. Franklin et al. 1995, Fore et al. 1996, Townsend et al. 1997a, Jeffers 1998, Bachman and Donato 2000, Brittain et al. 2001, Friberg et al. 2001).

Pearson's correlations between the new hydrological component scores and biotic measures were carried out to determine if these gave a better relationship than the individual variables. Multiple regressions using the hydrological component scores were considered to examine the overall explanatory value of all hydrological variables. Multiple regression is widely used in ecology for predictive modelling and data exploration (e.g. Hawkins et al. 1997, Jowett and Biggs 1997). By using the new hydrological component scores in multiple regressions the inter-correlation between input variables was reduced. To check for independence between the 3 groups, Pearson's correlations were calculated. They revealed all interpolated and temporal PC axes to be relatively uncorrelated, but some strong inter correlation between the map and interpolated PC axes (Appendix 3). Thus, the first three interpolated PC axes (IPC1, IPC2 and IPC3), the first 2 temporal PC axes (TPC1 and TPC2) and the second and third map PC axes (MPC2 and MPC3) were used in multiple regressions. This data set now fits the criteria for multiple regression, in that there is little or no inter-correlation among the hydrological variables. Plotting the principal component axes showed IPC1, IPC2 and MPC2 to have some non-linear relationships with the biotic variables, therefore, 2nd order polynomial relationships for these variables were also included in the appropriate regressions.

For variable reduction among the interpolated hydrological variables and biotic indices, high levels ($r>0.8$) of inter-correlation were identified and one variable from within a group of highly correlated variables were retained for further analysis. In the remainder of the analysis, examining the relationship between hydrological and biological variables, all significant relationships were identified. Significance level was set at $p = 0.0001$, with Bonferroni corrections this gave an overall comparison wise error of 0.15. DECORANA ordination was carried out in PC-ord (McCune and Mefford 1999), all other statistical analyses were performed using the SAS system (SAS 2000).

Results

Relationship between hydrological variables and stream invertebrate community structure

Individual correlations

To examine which single hydrological variable was most closely linked with community structure, DECORANA scores, species richness, total number of animals and densities of 22 taxa were correlated with individual flow variables (Table 1 and 2).

Some densities of individual taxa had little correlation with any hydrological variables; *Austroclima* sp., *Austrosimulium* sp., *C. humeralis*, Eriopterini sp., *Pycnocentria* sp. and *Zelandobius* sp. and are not discussed further. The remaining taxa densities were at least partly explained by 12 of the single hydrological variables. FRE3 was the most useful, having the strongest relationship with 5 of the taxa (Table 1).

Community structure was reduced to 2 detrended correspondence analysis (DECORANA) axes (DEC1 and DEC2). The first axis explained 22% of the variation in the data and the second 17% based on relative Euclidean distances. These axes are one measure of community structure so although they only account for 39% of the variation in community they are still a useful representation. Sites on the right of axis 1 are those high in Ostracoda, *P. antipodarum*, Oligochaeta and *Physa* sp., sites to the left of axis 1 are dominated by Elmidae, Orthocladiinae, *Zelandoperla* sp. and *Maoridiamesa* sp. Axis 2 represents variation within Chironomidae communities, with the bottom of the axis representing sites dominated by Orthocladiinae and Tanytarsini sp. and the top those high in *Naonella forsythia* and Macropelopiini sp. (Fig. 3).

The DECORANA axes show strong links with FRE3, discharge, evapotranspiration, mean annual rainfall, specific discharge and drought frequency (Table 1). *Deleatidium* sp., *P. antipodarum* and *Zelandoperla* sp. were most strongly linked with hydrological variables. *Deleatidium* sp. and *Zelandoperla* sp. densities increased with FRE3, local and catchment rainfall and specific discharge and down with evapotranspiration and drought frequency. *Maoridiamesa* sp., and *Psilochorema* sp. had weaker links, but in the same direction as *Deleatidium* sp.. *P. antipodarum* density, in contrast, went down with an increase in FRE3, rainfall, and specific discharge, and up with evapotranspiration and drought frequency. *Aoteapsyche* sp. densities increase with higher flows. Species richness and total number of animals showed no strong linear correlations with interpolated hydrological variables (Table 1).

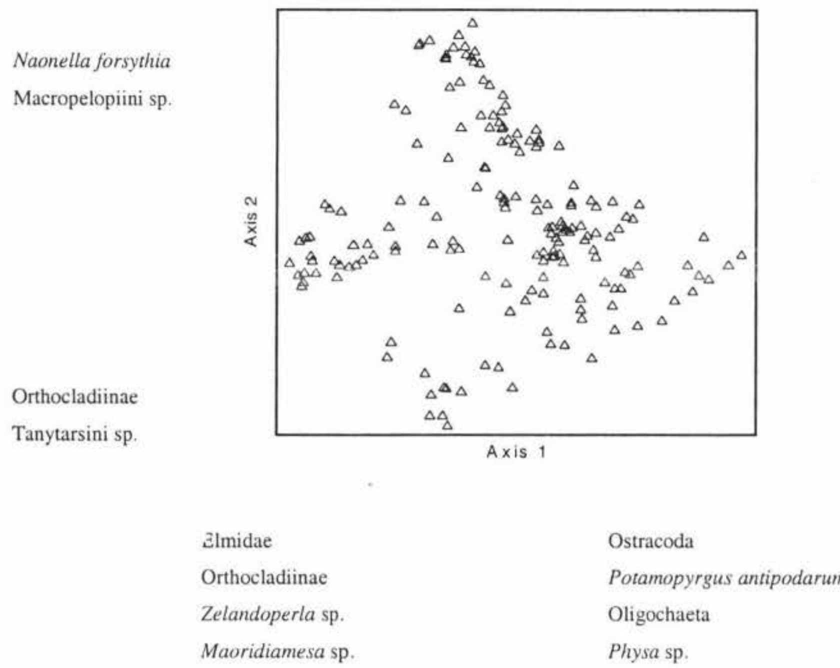


Figure 3: Plot of axis 1 versus axis 2 from a DECORANA ordination of taxa collected from 138 sampling occasions in the Whanganui Catchment, New Zealand.

Table 1: Pearson's correlation of interpolated hydrological variables that show strong relationships with taxa densities and DECORANA axes (only those with significant relationships are presented), from 138 sampling occasions in the Whanganui Catchment, New Zealand (+ or – indicates significance, full correlations in Appendix 4).

	FRE3	$\bar{Q}$	L10	H10	Evapo	Rain24	MAR	RMAR	q	StdL10	StdH10	DF
DEC1	-		-		+	-	-	-	-	-	-	+
DEC2	-	+	+	+				-				
Aoteapsyche sp.		+	+	+								
Aphrophila neozelandica					-							
Deleatidium sp.	+				-		+	+	+		+	
Hydrobiosis sp.											+	
Maoridiamesa sp.	+				-		+		+	+	+	
Potamopyrgus antipodarum	-				+	-	-	-	-	-	-	+
Psilochorema sp.	+						+	+				
Zelandoperla sp.	+				-	+	+	+	+	+	+	-

Of the correlations between map variables and community measures only the DECORANA scores (DEC1 and DEC2), *Aoteapsyche* sp., *Zelandoperla* sp. and *P. antipodarum* had more than 1 significant correlation (Table 2). *P. antipodarum* increase in density with average catchment slope and river density. *Zelandoperla* sp. densities decreased with an increase in slope and catchment shape and a decrease in density of large streams. The 2nd DECORANA axis and *Aoteapsyche* sp. showed a similar relationship increasing with area, perimeter, river density, order and catchment shape. Species richness and total number of animals showed no correlation with any map hydrological variables. There were few significant correlations between individual biological variables and temporal variables with only DECORANA axis 1, *A. diversus*, *Nesameletus* sp. and *O. feredayi* showing strong correlations. D40 was negatively correlated with *Nesameletus* sp. and *O. feredayi*. T14 had the opposite relationship with these taxa and *A. diversus*.

Table 2: Pearson's correlation of map and temporal hydrological variables that show strong relationships with taxa densities and DECORANA axes (only those with significant relationships are presented), from 138 sampling occasions in the Whanganui Catchment, New Zealand (+ or - indicates significance, full correlations in Appendix 4).

	Area	Peri	RD	Slope	Order	CS	DSO	DLO	Dd	T14	T21	T28	DT60	D20	D40
DEC1				+					-		-	-	+	+	+
DEC2	+	+	+		+	+		+							
Aoteapsyche sp.	+	+	+		+	+		+							
Aphrophila neozelandica				-											
Archichauliodes diversus										+					
Maoridiamesa sp.				-											
Nesameletus sp.										+	+	+	-		-
Olinga feredayi								-		+					-
Potamopyrgus antipodarum				+			-		-						
Tanytarsus vespertinus								+							
Zelandoperla sp.				-		-			+						

Non linear relationships

Species richness and total number of animals showed only a weak r^2 comparison with the hydrological variables. Polynomial regressions on FRE3 and river size however showed stronger relationships (Fig. 4).

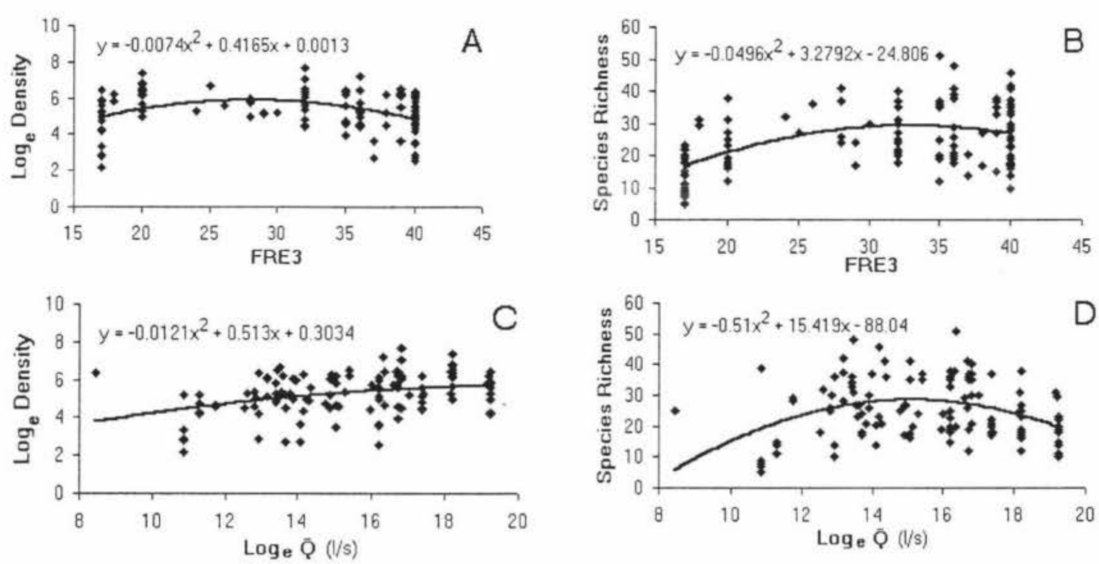


Figure 4: Polynomial regression between total number of animals (density) and species richness, and FRE3 and $\log_e \bar{Q}$ in the Whanganui Catchment, New Zealand (A – number of animals verses frequency of events (r^2=0.11), B – species richness verses frequency of events (r^2=0.23), C – number of animals verses mean annual flow (r^2=0.12), D – species richness verses mean annual flow (r^2=0.14)).

Principal component analysis on hydrological measures

Principal component analysis on the 15 interpolated hydrological variables gave 3 new variables retaining 91% (51%, 25% and 8% respectively) of the variation in the data. When eigenvectors were examined, the first principal component (IPC1) can be seen as a representation of the unit area variation in discharge (specific discharge), and yearly frequency of events, shown as a weighting of standardised flows, mean annual rainfall, FRE3 and negative evapotranspiration (Table 3). The second principal component (IPC2) represents the relative size of the rivers, or a weighting of the flow variables (Table 3). The third (IPC3) is a representation of flood frequency (Table 3).

The first 3 axes of a principal component analysis on the map variables explained 86% (53%, 22% and 13% respectively) of the data. The first principal component (MPC1) relates to catchment size (river density and actual size), MPC2 is a contrast of slope and river size, and MPC3 is a contrast between the density of small and medium rivers (Table 4).

The first 2 variables of a PCA on the temporal hydrological variables explain 75% (63% and 12% respectively) of the data. The first principal component (TPC1) represents a contrast between the amount of rain fallen and the number of days since rain, and the second (TPC2) is a contrast between variables that measure medium disturbance with short-term disturbance (Table 5).

Table 3: Eigenvectors of a principal component analysis on the 13 interpolated hydrological variables from sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand (shading indicates strongest component loadings).

	IPC1	IPC2	IPC3
$\bar{Q}$	0.03	0.55	-0.01
FRE3	0.31	-0.23	0.08
L10	0.10	0.53	0.01
H10	0.05	0.55	-0.11
Evapo	-0.35	-0.03	-0.14
Rain24	0.33	0.04	-0.04
MAR	0.36	-0.01	0.01
RMAR	0.31	-0.21	0.16
q	0.34	0.06	0.16
StdL10	0.35	-0.01	0.08
StdH10	0.33	0.01	-0.32
DF	-0.28	-0.05	-0.07
FF	-0.12	0.11	0.89

Table 4: Eigenvectors of a principal component analysis on the 11 map hydrological variables from sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand (shading indicates strongest component loadings).

	MPC1	MPC2	MPC3
Area	0.41	0.03	0.03
Perimeter	0.41	0.05	0.01
RD	0.41	0.05	0.03
Slope	0.04	-0.54	0.20
Order	0.40	0.04	-0.04
Rslope	-0.15	-0.31	0.23
CS	0.41	0.00	0.05
DSO	-0.13	0.49	0.47
DMO	-0.02	0.05	-0.82
DLO	0.35	0.02	0.13
Dd	-0.05	0.60	-0.01

Table 5: Eigenvectors of principal component analysis on the 10 temporal hydrological variables from sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand (shading indicates strongest component loadings).

	TPC1	TPC2
T1	-0.03	0.69
T7	-0.31	0.29
T14	-0.35	-0.10
T21	-0.37	-0.16
T28	-0.32	-0.35
DT20mm	0.30	-0.31
DT40mm	0.36	-0.07
DT60mm	0.36	0.16
D20mm	0.30	-0.34
D40mm	0.31	0.21

Principal components and community structure

Correlations were used to explore the relationship between these new summary variables and the community structure. The first DECORANA axis shows a negative relationship with the first interpolated PC score and the second map PC score (Table 6). The second DECORANA axis had a positive correlation with the second interpolated PC score, first map PC score and the first temporal PC score. None of the principal components were more useful than the individual variables at explaining variance in any of the individual community structure variables.

Table 6: Pearson's correlations between hydrological principal components variables and taxa densities and DECORANA axes, from 138 sampling occasions in the Whanganui Catchment, New Zealand (+ or – indicates significance, full correlations in Appendix 5).

	IPC1	IPC2	MPC1	MPC2	TPC1	TPC2
DEC1	-			-		
DEC2		+	+		+	
Aoteapsyche sp.		+	+			
Deleatidium sp.	+					
Maoridiamesa sp.	+			+		
Nesameletus sp.					-	
Olinga feredayi					-	
Oxyethira albiceps						+
Potamopyrgus antipodarum	-			-		
Psilochorema sp.	+					
Zelandoperla sp.	+		-	+		

Non linear relationships

Of the measures of community structure, species richness and total number of animals show some second order polynomial relationships with the principal components (Fig. 5). In particular with the first interpolated axis (total number of animals $r^2 = 0.10$ and species richness $r^2 = 0.17$) and for species richness with interpolated axis 2 and total number of animals with map axis 2. Other polynomial relationships explored were not significantly different from linear and were not considered further.

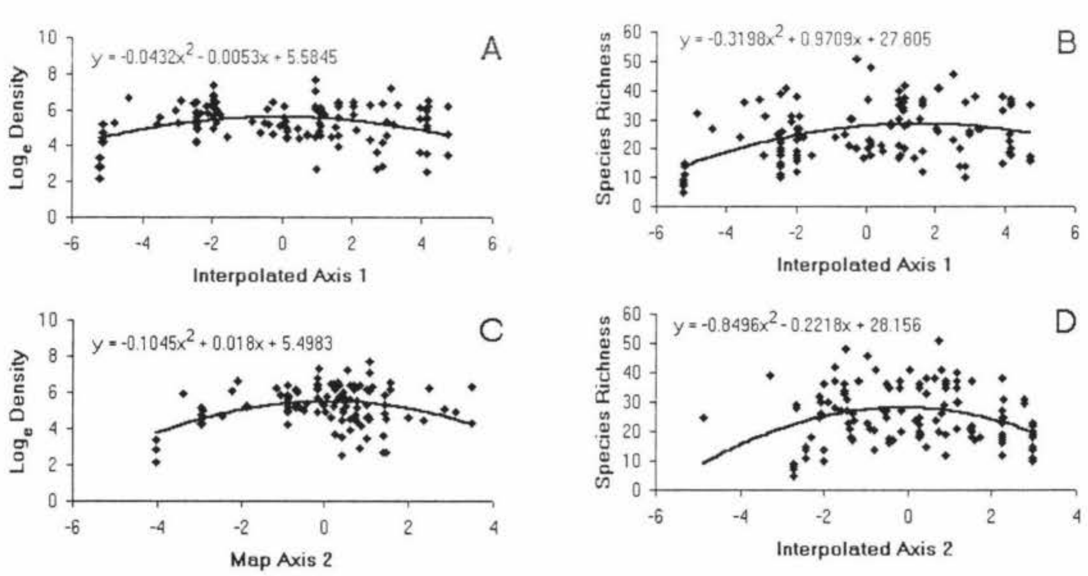


Figure 5: Polynomial regression between total number of animals (density) and species richness and, interpolated principal component 1 and 2 and map principal component 2 in the Whanganui Catchment, New Zealand (A – number of animals verses interpolated principal component 1 (r^2=0.10), B – species richness verses interpolated principal component 1 (r^2=0.17), C – number of animals verses map principal component 2 (r^2=0.14), D – species richness verses map principal component 2 (r^2=0.10)).

Multiple regressions

Multiple regressions were used to explore the combined influence of hydrological variables on community structure (Fig. 6). A number of models were highly significant but the residual plots of individual densities show signs of non-normality, *probably because of* the large number of zeros in the data set. *P. antipodarum* has the highest r-squared of the individual taxa models at 0.49, the only model with a better fit was that of the first DECORANA axis. IPC1 and IPC2 were significant in the majority of models. MPC3 and TPC2 were only significant in two models.

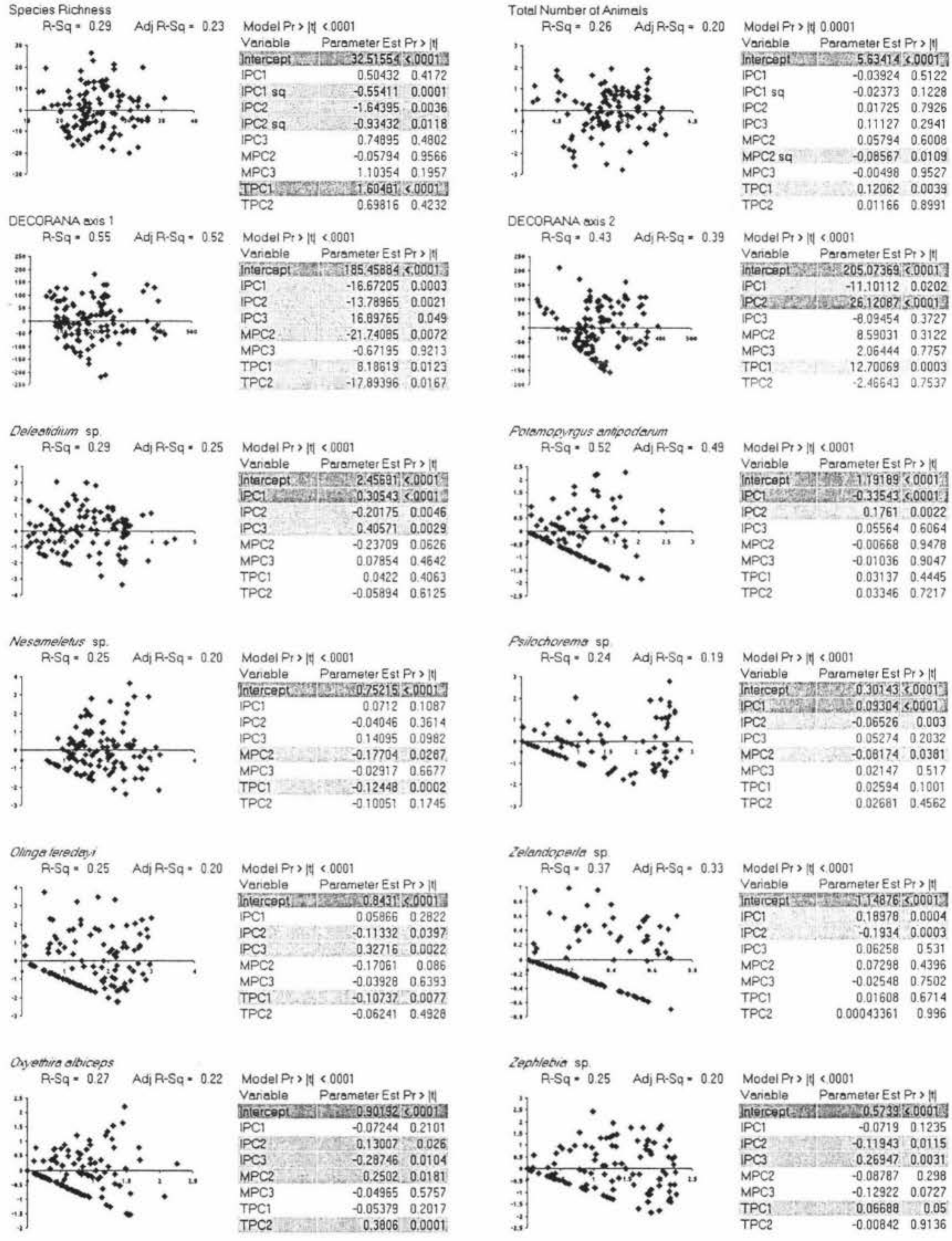


Figure 6: Taxa densities and DECORANA multiple regressions for the Whanganui Catchment, New Zealand. Residual plot (residuals vs. predicted), r-squared and p-values of explanatory variables (significance highlighted). Note stepwise regression was not used as input variables represent differing aspects of the hydrological regime.

Relationship between hydrological variables and biotic indices

Selection of indices

Twelve biotic indices were calculated, but many were strongly correlated (Table 7). In particular, IBP was highly correlated with Isimp, BP with Simp, Marg with Imarg, and QMCI with SQMCI. All EPT indices were also correlated and in preliminary exploration of the data responded in similar ways, %EPT-Oxy was chosen to represent EPT. To reduce redundant analysis only the indices of Isimp, Simp, Marg, %EPT-Oxy, MCI and QMCI were considered further.

Table 7: Pearson's correlations between biotic indices calculated for 138 sampling occasions in the Whanganui Basin, New Zealand (+++ or --- highly correlated, r>0.80, full correlation in Appendix 6).

	Isimp	Simp	IBP	BP	Imarg	Marg	EPT	EPT-Oxy	IEPT	IEPT-Oxy	MCI	QMCI	SQMCI
Isimp			+++										
Simp				+++									
IBP	+++												
BP		+++											
Imarg						+++							
Marg					+++								
EPT								+++	+++	+++		+++	
EPT-Oxy							+++		+++	+++		+++	
IEPT							+++	+++		+++			
IEPT-Oxy							+++	+++	+++				
MCI													
QMCI							+++	+++					+++
SQMCI												+++	

Individual correlations

A significant amount of variance (15-41%) in the 6 indices considered was explained by correlations (linear) between individual hydrological variables (Tables 8 and 9). Three of the indices, Isimp, Marg and MCI were best explained by FRE3. The other 3 indices, Simp, %EPT-Oxy and QMCI, were best explained by catchment area. Of the interpolated variables FRE3, discharge, evapotranspiration, local and catchment mean annual rainfall, drought frequency and specific discharge variables showed a number of strong correlations (Table 8). MCI, QMCI, Marg and EPT increased with FRE3 and specific discharge and decreased with discharge and evapotranspiration.

Table 8: Pearson's correlation of interpolated variables that show strong relationships (only those with significant relationships are presented) with indices, from 138 sampling occasions in the Whanganui Catchment, New Zealand (+ or - indicates significance, full correlation in Appendix 7).

	FRE3	$\bar{Q}$	L10	H10	Evapo	Rain24	MAR	RMAR	q	StdL10
Isimp	-									
Simp										
Marg	+				-		+	+	+	
EPT-Oxy	+	-	-	-				+		
MCI	+	-		-	-	+	+	+	+	+
QMCI	+	-	-	-				+		

EPT, MCI and QMCI were negatively correlated with area, perimeter, river density, slope, stream order, catchment shape and density of large rivers (Table 9). There were very few correlations between individual biological variables and temporal variables, with only T28 and D40 showing any significant relationships. EPT, MCI and QMCI increased with a greater T28, Isimp decreased with a greater T28, with D40 having the opposite relationship to T28.

Table 9: Pearson's Correlation of map and temporal variables that show strong relationships (only those with significant relationships are presented) with indices, from 138 sampling occasions in the Whanganui Catchment, New Zealand (+ or - indicates significance, full correlation in Appendix 7).

	Area	Perimeter	RD	Slope	Order	CS	DSO	DLO	Dd	T28	D40
Isimp										-	+
Simp											
Marg				-			+		+		
EPT-Oxy	-	-	-		-	-		-		+	-
MCI	-	-	-	-	-	-		-		+	-
QMCI	-	-	-		-	-		-		+	-

There were no strong polynomial relationships between indices and hydrological variables.

Principal components and indices

The hydrological principal components calculated in the community structure section, were also evaluated against the indices (Fig. 7). The first two interpolated principal components (specific discharge and flood frequency, and river size respectively) show some highly significant correlations. In particular, QMCI decreased with increasing river size. Of the map principal components, the first two principal components (catchment size and river size) have significant correlations with a number of indices. As catchment size increased EPT, MCI and QMCI decreased. None of the principal components were more useful than the individual variables at explaining variance in the indices.

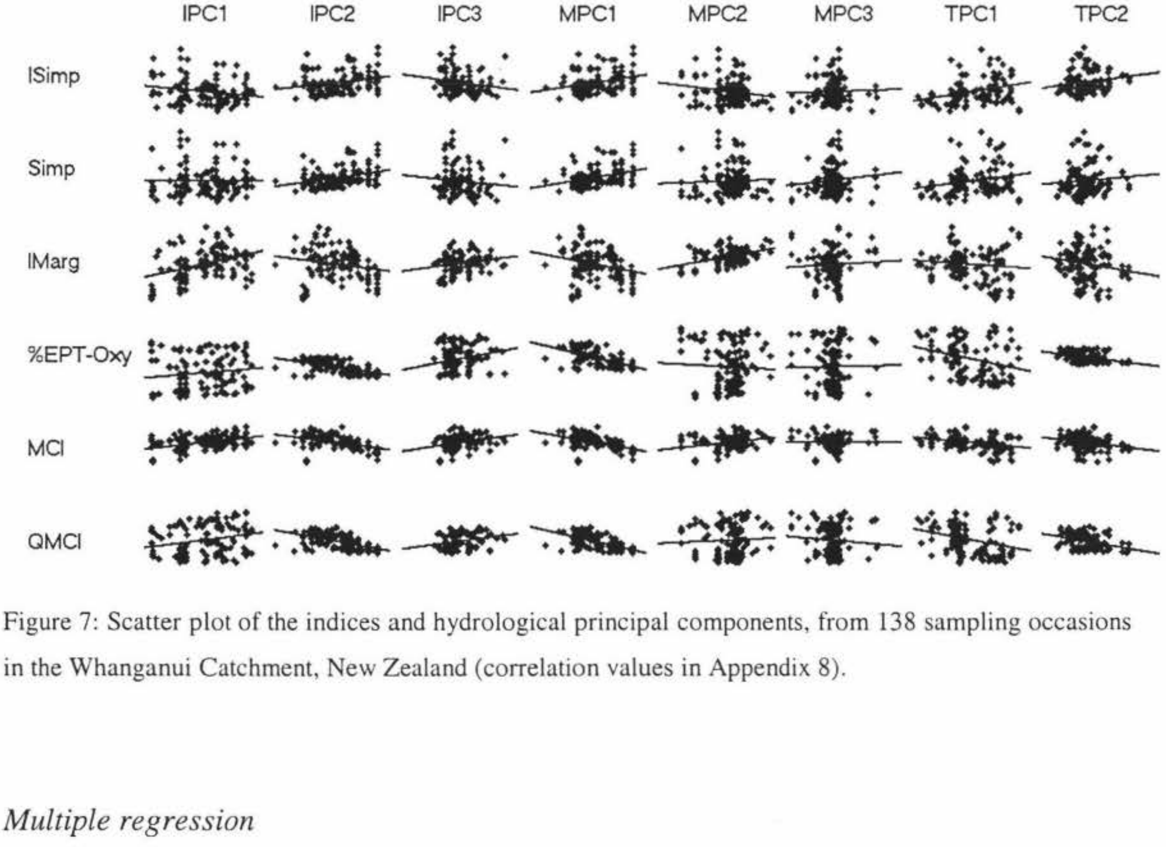


Figure 7: Scatter plot of the indices and hydrological principal components, from 138 sampling occasions in the Whanganui Catchment, New Zealand (correlation values in Appendix 8).

Multiple regression

Of the biotic index multiple regressions, all models were significant at the 0.05 level (Fig. 8). %EPT-Oxy, MCI and QMCI show strong relationships with r-squares of greater than 0.5. IPC2 came out as being significant in all regressions, and highly significant in most of them. MPC3 and TPC1 showed very few significant relationships. The residual plots show data to be relatively normal and evenly distributed (Fig. 8).

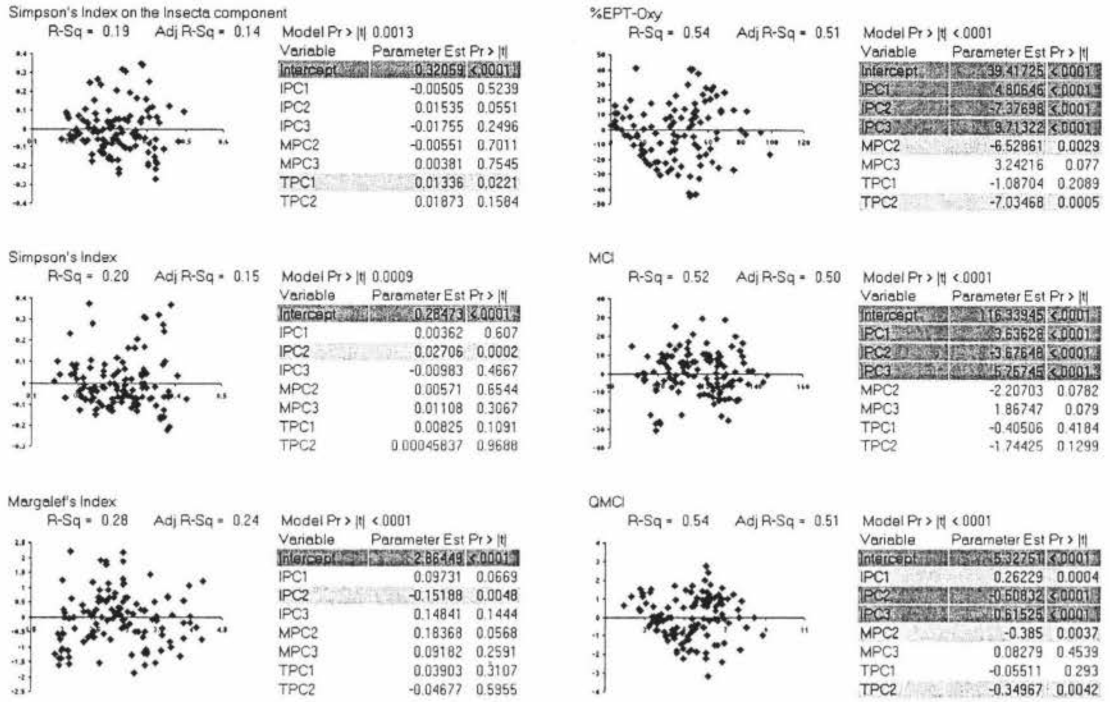


Figure 8: Whanganui Basin biotic index multiple regressions, residual plot (residuals vs. predicted), r-squared and p-values of explanatory variables (significance highlighted).

Results summary

Individual Correlations

Of the hydrological variables number of events per year over 3 times the median flow (FRE3) had the strongest links with species richness (2nd order polynomial), *C. humeralis* (+), *Deleatidium* sp. (+), *P. antipodarum* (-), *Pycnocentrodes* sp. (+) and *Zelandoperla* sp. (+). Specific discharge of a 10-year flood (StdH10) had the strongest correlations with *Austroclima* sp. (-), *Hydrobiosis* sp. (+), *Maoridiamesa* sp. (+), *N. confusum* (+) and *Zephlebia* sp. (-). Number of days since 40 mm of rain has fallen (D40) had the strongest links with Eriopterini sp. (+), *Nesameletus* sp. (-) and *O. feredayi* (-). Total number of animals and species richness showed some strong 2nd order polynomial relationships. FRE3 was the best predictor for Simpson's index based on the Insecta taxa only (-), Margalef's index (+) and the water quality index of MCI (+). Catchment area was the best predictor for Simpson's index based on the full sample (+), and the water quality indices of %EPT- *O. albiceps* (-) and QMCI (-).

PCA Correlations

The first two interpolated (specific discharge and frequency of events, and river size respectively) and map (catchment size, and slope and river size respectively) scores had strong links with most biological variables. None of the correlations between biotic variables and principal component scores were stronger than with individual hydrological variables.

Multiple Regressions

P. antipodarum showed the best taxa model (r^2=0.49), predicted by the first two interpolated principal components, specific discharge and frequency of events, and river size. These first two interpolated scores were significant in the majority of models. Water quality indices(EPT, MCI and QMCI) had good multiple regressions (r^2 > 0.50) predicted by a combination of all principal components apart from the third map and 1st temporal scores.

Discussion

The results of this study build on information showing that the hydrologic regime is fundamental in understanding stream invertebrate communities (Minshall 1988, Poff and Ward 1989, Clausen and Biggs 1997, Wood et al. 2000). Ordinations, which represent overall community structure, suggest biological communities are responding to the specific discharge, catchment size, number of events over time and timing since the last event at a river site. These relationships were shown in both the individual correlations and multiple regressions. To examine how these aspects relate specifically, each aspect of the hydrological regime will be discussed independently. In particular, FRE3, river size and specific discharge showed the strongest individual relationships with biological variables. Of the temporal variables, amount of rain over the last 14 days and number of days since 40 mm of rain fell on a single day, had the best correlations with the biotic variables.

The number of flow events over 3 times the medium flow (FRE3) was the hydrological variable most often correlated with biological variables; 15 out of 32 of the biotic variables had significant correlations with FRE3, suggesting that the number of events of a magnitude 3 times the medium flow is important in influencing numerous aspects of community structure. This supports other studies that have shown flood frequency to be an important factor in influencing a stream community (e.g. Scarsbrook and Townsend 1993, Death and Winterbourn 1995, Clausen and Biggs 1997, McCabe and Gotelli 2000). Evenness (Simpson's and Berger-Parker) and richness (Margalef's) were highest with high FRE3, thus more regularly flushed sites have richer and evener communities. This appears contrary to most literature (Biggs and Close 1989, Scarsbrook and Townsend 1993, Death and Winterbourn 1995, McCabe and Goteli 2000) that suggests richness and evenness will be lower with increased bed disturbance. FRE3 has been suggested as a useful measure of bed disturbance (Sagar 1986, Clausen and Biggs 1997). In this study, FRE3 is higher in the upland smaller catchments as rainfall is higher in these areas and localised events will have a greater impact on smaller catchments. Thus, FRE3 may be representative of both frequency of events, and location and size of streams, with streams in these locations (upland streams of predominantly native vegetation) having both higher richness and evenness.

A number of individual taxa had strong correlations with FRE3; in particular, *Deleatidium* sp. and *Zelandoperla* sp. densities increased with higher FRE3, and *P. antipodarum* densities decreased. *Deleatidium* sp. and *Zelandoperla* sp. are good colonisers (Mackay 1992, Townsend et al. 1997b) thus would be more likely to be found in areas that are flushed regularly in contrast to *P. antipodarum* that is a poor coloniser and therefore more likely to be found in areas that are more stable (Holomuziki and Biggs 1999). Water quality, as measured by indices, increased with higher FRE3. Taxa that are sensitive to water quality tend to be good colonisers (Mackay 1992, Townsend et al. 1997b, Stark 1998, Jowett 2000) and thus more likely to be found in streams with higher disturbance regimes, as previously stated. It is also possible that water quality may be influenced by the frequency of events, with a greater frequency resulting in more flushing and thus lower enrichment in periods of base flow. Species richness and total density had 2nd order polynomial relationships with FRE3. This may be explained by the intermediate disturbance hypotheses, the greatest diversity will be at an intermediate disturbance when neither biotic controls or disturbance are dominating the community (Townsend et al. 1997c). Alternatively, the low species richness and total number of animals at high FRE3 could be a result of high bed disturbance, elevated current velocities and substrate movement causing mortality and the loss of individuals (Resh et al. 1988). The low species richness and total number of animals at low FRE3 may be associated with the fact that these locations were in lowlands where the land is more developed and streams tend to have lower species richness and total number of animals (Collier et al. 2000).

River size as represented by catchment size (area, perimeter, catchment shape), discharge ($\bar{Q}$, H10, L10), stream order and river density and combined into IPC2 and MPC1 all show similar relationships with invertebrate communities. Ormerod and Edwards (1987) also found size, as measured by distance from the source, as an important factor in determining invertebrate community structure. Evenness and richness were lower in large rivers. There are a number of possible explanations for this; larger rivers are further down in the catchment, thus have a high possibility of being impacted (Harding et al. 1999). Alternatively, larger rivers may have greater periphyton biomass and thus less diverse algal grazed dominated communities. Larger rivers may also offer less variety in habitat and thus develop communities that are less rich and even. Jowett and Duncan (1990) identified flow variability to decrease with catchment size in New Zealand rivers, thus large rivers will show similar relationships to low FRE3. Water quality, as measured by indices, was degraded with increased size and larger rivers generally have more organic enrichment. In addition, water quality measures are increased by taxa that like swift water and stony river beds (Winterbourn and Gregson 1981, Jowett 2000), and this habitat is more likely to occur in smaller streams. This suggests that these indices can only be used to compare areas of similar hydrological regimes, in similar sized rivers (Death et al. 1999). Of the individual taxa, few had strong correlations with river size. *Aoteapsyche* sp. and *P. antipodarum* increase in density while *Psilochorema* sp. and *Zelandoperla* sp. decrease in density with larger river size. Large rivers have a tendency to be slower flowing which *P. antipodarum* prefers (Jowett 2000). *Psilochorema* sp. and *Zelandoperla* sp. favour stony streams (Winterbourn and Gregson 1981), which in the Whanganui Catchment stony habitat tends to be smaller rivers.

Specific discharge was represented by 7 interpolated variables that respond in similar ways (evapotranspiration, 24 hour rainfall, catchment mean annual rainfall, local mean annul rainfall and the flow variables of mean, 10 year low and high flows standardised by the catchment area). Richness and evenness increased with specific discharge. A high specific discharge will mean that low flows will not be as extreme, thus droughts have less effect on habitat availability and richness and evenness will be higher. Enrichment, as measured by the water quality indices, decreased with greater specific discharge. Specific discharge was higher in the upper catchment were rainfall is high and it is these locations where the streams are smaller and water relatively swift as preferred by those taxa that have higher values in the water quality indices (lower enrichment) (Jowett 2000). An alternative explanation may be that areas with higher specific discharge have a higher dilution of any enrichment. Of the individual taxa *A. neozelandica*, *Deleatidium* sp., *Hydrobiosis* sp., *Maoridiamesa* sp., *N. confusum*, *Psilochorema* sp. and *Zelandoperla* sp. increase with higher specific discharge. These taxa tend to be the faster colonisers and more capable of dealing with faster flowing water . *Austroclima* sp., *P. antipodarum* and *Zephlebia* sp. on the other hand, decrease with high specific discharge. These taxa are more associated with slow flowing waters in low land areas (Jowett 2000).

Recent event history as measured by the amount of rain and the number of days since an event showed some relationship to the biotic indices and individual taxa densities but not to the same extent as the other hydrological variables. Hendricks et al. (1995) also identified the time since the last event as being an important factor in determining stream communities. Samples used in this study were not taken directly after high flow events, thus invertebrate responses immediately following such events were not explored. The more rain that had fallen in the couple of weeks prior to sampling the higher the evenness. This may be a result that greater stability of a stream will allow communities to become more dominated by a few species (Death 1996b). Alternatively areas more likely to have rain in the last couple of weeks are those with a greater frequency of flooding. Water quality indices increased (less enriched) with amount of rain. This may be the influence of more recent flushing and thus less enrichment; alternatively, the smaller stream catchments high in the Whanganui catchment (which were often more pristine) have a greater chance of having a shorter time since the last event because mean annual rainfall is higher in these regions. Of the individual taxa *A. diversus*, *Nesameletus* sp., and *O. feredayi* densities go up with greater amount of rain in the last couple of weeks and a shorter time since the last event, with *P. antipodarum* has the opposite relationship. *P. antipodarum* has a preference for slow flowing (Jowett

2000) stable areas (Holomuziki and Biggs 1999) and thus prefer areas where a longer time since the last event occurs.

Linear multiple regression of the indices showed the hydrological variables combined do not explain a high proportion of community richness or evenness (15%–22%) although river size is the dominant explanatory variable. When polynomial relationships were added, the relationship is stronger but still not high (20%-23%). This fits with explanations given previously in relation to FRE3 polynomial regression. The water quality indices were explained by a combination of hydrological variables (50%–51%) and relate to most aspects (spatial and temporal) of the hydrological regime. This suggests that use of water quality indices in a comparison of sites with differing hydrological characteristics may be biased. As data collected has been observational not experimental data distribution is not completely balanced. It is possible that water quality indices are working and coincide with hydrological characteristics. Further exploration with experimental study would be useful. Of the individual taxa, 8 were strongly linked ($p < 0.0001$) with a combination of hydrological variables explaining from 19% to 49 % of the data. The *P. antipodarum* model was the strongest, and was dominated by specific discharge (negatively) and river size (positively). Thus suggests that larger streams, more influenced by droughts will have a higher density of *P. antipodarum*.

This study has shown 4 major hydrological components that can be calculated from long-term hydrological records and topographical information; flood frequency, river size, specific discharge and timing since the last event are important in influencing invertebrate communities. FRE3 was the most highly correlated of the hydrological variables. Two other variables that had strong links were river size and specific discharge. River size can be determined from either mean annual flow or catchment area. Mean annual flow had slightly stronger correlations with biota but it may be easier to obtain an accurate estimate of catchment area. Specific discharge can be determined from mean annual flow divided by catchment area, or alternately one can use mean annual rainfall. In New Zealand, and many other parts of the world, isohyet maps of mean annual rainfall are readily available making this relatively easy to calculate. FRE3 and timing since the last event require more intensive data. To calculate FRE3, data records of at least 10 years are required; this is not always available and interpolation of data points is needed. This also requires some local knowledge of the area, making FRE3 a more complex variable to be used in assessment. Suggesting that river size or specific discharge may be more useful than FRE3. Timing since the last event, in this case, was calculated from rainfall sites but flow sites would be as applicable. The requirement to calculate timing is that there is at least daily information on rainfall or flow over the sampling period involved. A combination of these 4 hydrological components gave the best relationships between the hydrological regime and biota.

There are a number of limitations with this study in that it only explored the relationship between variables in one catchment, over a narrow range of environments and is based on pre-collected data. However, it has clearly shown that hydrology and limnology are intrinsically linked and thus it is important to integrate the two disciplines to better understand the riverine environment (Statzner and Higler 1986). This study has examined a variety of hydrological parameters that may be useful to integrate in biological studies but further work is needed to better define measurable variables of the hydrological regime and their relationship on a larger scale. In New Zealand and worldwide there is a wealth of hydrological information that is under utilized. By gaining a better understanding of how the hydrological regime relates to benthic biota it may be possible to analyse this data ecologically and allow for more effective management of river systems.

Literature Cited

Angradi, T. R. 1997. Hydrologic context and macroinvertebrate community response to floods in an Appalachian headwater stream. The American Midland Naturalist 138: 371-386.

Bachmann, C. M., and T. F. Donato. 2000. An information theoretic comparison of projection and principal component features for classification of Landsat TM imagery of central Colorado. International Journal of Remote Sensing 21: 2927-2935.

Berger, W. H., and F. L. Parker. 1970. Diversity of plankton Formininfera in deep sea sediments. Science 168: 1345-1347.

Biggs, B. J. F., and M. E. Close. 1989. Periphyton biomass dynamics in gravel bed rivers: the relative effects of flows and nutrients. Freshwater Biology 22: 209-231.

Bobba, A. G., V. P. Singh, and L. Bengtsson. 2000. Application of environmental models to different hydrological systems. Ecological Modelling 125: 15-49.

Bond, N. R., and B. J. Downes. 2000. Flow-related disturbance in streams: an experimental test of the role of rock movement in reducing macroinvertebrate population densities. Australian Journal of Marine and Freshwater Research 51: 333-337.

Boulton, A. J., C. G. Peterson, N. B. Grimm, and S. G. Fisher. 1992. Stability of an aquatic macroinvertebrate community in a multiyear hydrologic disturbance regime. Ecology 73: 2192-2207.

Brittain, J. E., S. J. Saltveit, E. Castalla, J. Bogen, T. E. Bonsnes, I. Blakar, T. Bremnes, I. Haug, and G. Velle. 2001. The macroinvertebrate communities of two contrasting Norwegian rivers in relation to environmental variables. Freshwater Biology 46: 1723-1736.

Clausen, B., and B. J. F. Biggs. 1997. Relationships between benthic biota and hydrological indices in New Zealand streams. Freshwater Biology 38: 327-342.

Clausen, B., and B. J. F. Biggs. 2000. Flow variables for ecological studies in temperate streams: groupings based on covariance. Journal of Hydrology 237: 184-197.

Clifford, H. T., and W. Stephenson. 1975. An introduction to numerical classification. Academic Press, New York.

Collier, K. J., B. J. Smith, J. M. Quinn, M. R. Scarsbrook, N. J. Halliday, G. F. Croker, and S. M. Parkyn. 2000. Biodiversity of stream invertebrate faunas in a Waikato hill-country catchment in relation to land use. New Zealand Journal Entomologist 23: 9-22.

Death, R. G. 1996a. The effect of habitat stability on benthic invertebrate communities: The utility of species abundance distributions. Hydrobiologia 317: 97-107.

Death, R. G. 1996b. The effect of patch disturbance on stream invertebrate community structure: The influence of disturbance history. Oecologia 108: 567-576.

Death, R.G. in press. Predicting diversity from disturbance regimes in forest streams. Oikos.

Death, R. G., H. McWilliam, and M. Rodway. 1999. Macroinvertebrate monitoring: lowland streams and order non-shingle river ecotypes. Pages 33-41 in Ministory for the Environment (editors). The use of macroinvertebrates in water management. Ministory for the Environment , Wellington,: 33-41

Death, R. G., and M. J. Winterbourn. 1995. Diversity patterns in stream benthic invertebrate communities - the influence of habitat stability. Ecology 76: 1446-1460.

Department of Conservation. 1988. Wanganui River minimum flows submission : for consideration by the Rangitikei-Wanganui Catchment Board. Department of Conservation, Wellington.

Doisy, K. E., and C. F. Rabeni. 2001. Flow conditions, benthic food resources, and invertebrate community composition in a low-gradient stream in Missouri. Journal of the North American Benthological Society 20: 17-32.

Downes, B. J., A. Glaister, and P. S. Lake. 1997. Spatial variation in the force required to initiate rock movement in 4 upland streams: Implications for estimating disturbance frequencies. Journal of the North American Benthological Society 16: 203-220.

Downes, B. J., P. S. Lake, A. Glaister, and J. A. Webb. 1998. Scales and frequencies of disturbances: rock size, bed packing and variation among upland streams. Freshwater Biology 40: 625-639.

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

Environmental Systems Research Institute. 1999. ArcView Version 3.2. Redland, California.

Flecker, A. S., and B. Feifarek. 1994. Disturbance and the temporal variability of invertebrate assemblages in two Andean streams. Freshwater Biology 31: 131-142.

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

Franklin, S. B., D. J. Gibson, P. A. Robertson, J. T. Pohlmann, and J. S. Fralish. 1995. Parallel analysis: A method for determining significant principal components. Journal of Vegetation Science 6: 99-106.

Friberg, N., A. M. Milner, L. M. Svendsen, C. Lindegaard, and S. E. Larsen. 2001. Macroinvertebrate stream communities along regional and physico-chemical gradients in Western Greenland. Freshwater Biology 46: 1753-1762.

Furse, M. T., J. F. Moss, and P. D. Armitage. 1984. The influence of seasonal and taxonomic factors on the ordination and classification of running water sites in Great Britain and in the prediction of their macroinvertebrate communities. Freshwater Biology 14: 256-280.

Gislason, G. M., H. Adalsteinsson, I. Hansen, J. S. Olafsson, and K. Svavarsdottir. 2001. Longitudinal changes in macroinvertebrate assemblages along a glacial river system in central Iceland. Freshwater Biology 46: 1737-1751.

Gumbel, E. J. 1958. Statistics of Extremes. Columbia University Press, New York.

Gurnell, A. M., C. R. Hupp, and S. V. Gregory. 2000. Linking hydrology and ecology. Hydrological Processes 14: 2813-2815.

Hannaford, M. J., M. T. Barbour, and V. H. Resh. 1997. Training reduces observer variability in visual-based assessments of stream habitat. Journal of the North American Benthological Society 16: 853-860.

Harding, J. S. 1994. Variations in benthic fauna between differing lake outlet types in New Zealand. New Zealand Journal of Marine and Freshwater Research 28: 417-427.

Harding, J. S., R. G. Young, J. W. Hayes, K. A. Shearer, and J. D. Stark. 1999. Changes in agricultural intensity and river health along a river continuum. Freshwater Biology 42: 345-356.

Hardison, B. S., and J. B. Layzer. 2001. Relationships between complex hydraulics and the localized distribution of mussels in three regulated rivers. Regulated Rivers-Research & Management 17: 77-84.

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

Hawkins, C. P., R. H. Norris, J. N. Hogue, and J. W. Feminella. 2000. Development and evaluation of predictive models for measuring the biological integrity of streams. Ecological Applications 10: 1456-1477.

Hendricks, A. C., L. D. Willis, and C. Snyder. 1995. Impact of flooding on the densities of selected aquatic insects. Hydrobiologia 299: 241-247.

Holomuzki, J. R., and B. J. F. Biggs. 1999. Distributional responses to flow disturbance by a stream- dwelling snail. Oikos 87: 36-47.

Horrox, J. V. 1998. Benthic communities of the Whanganui River catchment: the effects of land use and geology. M.Sc. Thesis. Massey University, Palmerston North.

Jeffers, J. N. R. 1998. Characterization of river habitats and prediction of habitat features using ordination techniques. Aquatic Conservation: Marine and Freshwater Ecosystems 8: 529-540.

Jowett, I. 2000. Flow management. Pages 289-312 in K. J. Collier and M. J. Winterbourn (editors). New Zealand stream invertebrates: ecology and implications for management. New Zealand Limnological Society, Christchurch.

Jowett, I. G., and B. J. F. Biggs. 1997. Flood and velocity effects on periphyton and silt accumulation in two New Zealand rivers. New Zealand Journal of Marine and Freshwater Research 31: 287-300.

Jowett, I., J. Boubee, and B. J. F. Biggs. 2000. Fish, benthic invertebrate, periphyton ans instream habitat in the upper Whanganui River catchment above and below the Western Diversions. Consultantly report ELE90243. National Institue of Water and Atmospher Research Ltd, Hamilton.

Jowett, I. G., and M. J. Duncan. 1990. Flow variability in New Zealand Rivers and its relationship to in-stream habitat and biota. New Zealand Journal of Marine and Freshwater Research 24: 305-317.

Lammert, M., and J. D. Allan. 1999. Assessing biotic integrity of streams: Effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. Environmental Management 23: 257-270.

Land Information New Zealand. 2000. NZMS 1:50000 Series LINZ, Wellington.

Lenat, D. R. 1988. Water quality assessment of streams using a qualitative collection method for benthic invertebrates. Journal of the North American Benthological Society 7: 22-233.

Mackay, R. J. 1992. Colonization by lotic macroinvertebrates: a reiew of processes and patterns. Canadian Journal of Fisheries and Aquatic Sciences 49: 617-628.

Malmqvist, B., and C. Otto. 1987. The influence of substrate stability on the composition of stream benthos: an experimental study. Oikos 48: 33-38.

Matthaei, C. D., and C. R. Townsend. 2000. Long-term effects of local disturbance history on mobile stream invertebrates. Oecologia 125: 119-126.

Matthaei, C. D., U. Uehlinger, E. I. Meyer, and A. Frutiger. 1996. Recolonization by benthic invertebrates after experimental disturbance in a Swiss prealpine river. Freshwater Biology 35: 233-248.

McCabe, D. J., and N. J. Gotelli. 2000. Effects of disturbance frequency, intensity, and area on assemblages of stream macroinvertebrates. Oecologia 124: 270-279.

McCune, B., and M. J. Mefford. 1999. PC-ORD for Windows - Multivariate Analysis of Ecological Data Version 4.0. Gleneden Beach, Oregon, MjM Software.

McKerchar, A., R. Ibbitt, and R. Woods. 1997. Analysis and estimation of extreme events: deterministic methods. Pages 51-63 in M. P. Mosley and C. P. Pearson (editors). Floods and droughts: the New Zealand experience. New Zealand Hydrological Society Inc., Wellington

Minshall, G. W. 1988. Stream ecosystem theory: a global perspective. Journal of the North American Benthological Society 7: 263-288.

Mosley, M. P. 1992. Waters of New Zealand. The Caxton Press, Wellington.

Ormerod, S. J., and R. D. Edwards. 1987. The ordination and classification of macroinvertebrate assemblages in the catchment of the River Wye in relation to environmental factors. Freshwater Biology 17: 533-546.

Poff, N. L. 1996. A hydrogeography of unregulated streams in the United States and an examination of scale-dependence in some hydrological descriptors. Freshwater Biology 36: 71-91.

Poff, N. L., and J. D. Allan. 1995. Functional-organization of stream fish assemblages in relation to hydrological variability. Ecology 76: 606-627.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47: 769-784.

Poff, N. L., and J. V. Ward. 1989. Implications of stream flow variability and predictability for lotic community structure: a regional analysis of stream flow Patterns. Canadian Journal of Fisheries and Aquatic Sciences 46: 1805-1818.

Reckhow, K. H. 1999. Water quality prediction and probability network models. Canadian Journal Of Fish and Aquatic Science 56: 1150-1158.

Resh, V. H., A. V. Brown, A. P. Covich, M. E. Gurtz, H. W. Li, W. Minshall, S. R. Reice, A. L. Sheldon, B. Wallace, and R. C. Wissmar. 1988. The role of disturbance in stream ecology. Journal of the North American Benthological Society 7: 433-455.

Reynolds, C. S. 2000. Hydroecology of river plankton: the role of variability in channel flow. Hydrological Processes 14: 3119-3132.

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

Robinson, C. T., and G. W. Minshall. 1986. Effects of disturbance frequency on stream benthic community structure in relation to canopy cover. Journal of the North American Benthological Society 5: 237-248.

Rodgers, M. W. 2001. Hilltop System. Napier, New Zealand, Hilltop Software.

Sagar, P. M. 1986. The effects of floods on the invertebrate fauna of a large, unstable braided river. New Zealand Journal of Marine and Freshwater Research 20: 37-46.

SAS. 2000. SAS User's Guide: Statistics, Version 8. Cary, North Carolina, SAS Institute Inc.

Scarsbrook, M. R., and C. R. Townsend. 1993. Stream community structure in relation to spatial and temporal variation: a habitat templet study of two New Zealand streams. Freshwater Biology 29: 395-410.

Simpson, E. H. 1949. Measurement of diversity. Nature 163: 688.

Smakhtin, V. U. 2001. Low flow hydrology: a review. Journal of Hydrology 240: 147-186.

Stark, J. D. 1985. A macroinvertebrate community index of water quality for stony streams. Water and Soil miscellaneous publication 87, Wellington.

Stark, J. D. 1993. Performance of the macroinvertebrate community index: effects of sampling method, sampling replication, water depth, current velocity and substratum on index values. New Zealand Journal of Marine and Freshwater Research 27: 463-476.

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

Statzner, B., and B. Higler. 1986. Stream hydraulics as a major determinant of benthic invertebrate zonation patterns. Freshwater Biology 16: 127-139.

Thompson, W. R., and K. M. Walter. 1996. Tideda for Windows. Christchurch, National Institute of Water and Atmospheric Research.

Tonkin and Taylor Consultant Engineers. 1978. Water resources of the Wanganui River. Tonkin and Taylor Consultant Engineers, Auckland.

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

Townsend, C. R., S. Doledec, and M. R. Scarsbrook. 1997b. Species traits in relation to temporal and spatial heterogeneity in streams: A test of habitat templet theory. Freshwater Biology 37: 367-387.

Townsend, C. R., A. G. Hildrew, and J. Francis. 1983. Community structure in some southern English streams: the influence of physicochemical factors. Freshwater Biology 13: 521-544.

Townsend, C. R., M. R. Scarsbrook, and S. Doledec. 1997c. The intermediate disturbance hypothesis, refugia, and biodiversity in streams. Limnology and Oceanography 42: 938-949.

Townsend, C. R., M. R. Scarsbrook, and S. Doledec. 1997d. Quantifying disturbance in streams: alternative measures of disturbance in relation to macroinvertebrate species traits and species richness. Journal of the North American Benthological Society 16: 531-544.

Turak, E., L. K. Flack, R. H. Norris, J. Simpson, and N. Waddell. 1999. Assessment of river condition at a large spatial scale using predictive models. Freshwater Biology 41: 283-298

Winterbourn, M. J., and K. L. D. Gregson. 1981. Guide to the aquatic insects of New Zealand. Bulletin of the Entomological Society of New Zealand 5.

Wood, P. J., M. D. Agnew, and G. E. Petts. 2000. Flow variation variations and macroinvertebrate community responses in a small groundwater-dominated stream in south-east England. Hydrological Processes 14: 3133-3147.

3

Influence of landscape on stream invertebrate communities in tributaries of the Whanganui Catchment, New Zealand

Abstract

Catchment and riparian scale landscape measures of land cover and geology were used to examine patterns in benthic stream macroinvertebrate communities in rivers of the Whanganui Catchment, New Zealand. Landscape variables were derived from New Zealand's land classification database and land resource inventory. A number of diversity and enrichment indices as well as a detrended correspondence ordination were calculated from the invertebrate communities. Fifteen out of the 22 key taxa also had more than one strong link with landscape variables, with *Potamopyrgus antipodarum* have the strongest associations. This suggests that landscape variables have a strong influence on invertebrate community structure. Of the landscape variables percentage native vegetation showed strong positive links with Margalef's, EPT and MCI indices and *Deleatidium* sp. densities and percent ash had a strong positive link with the first ordination axis and a negative link with *Aphrophila neozelandica*. Canonical correlations showed catchment scale variables to be better predictors of community structure than riparian scale variables.

Key Words: Benthic invertebrate; canonical correlation; geographic information systems; geology; land cover; landscape; multiple regression; New Zealand; river; Whanganui

Introduction

There is currently a strong worldwide interest in the use of biological indicators, including benthic stream macroinvertebrates, as monitoring and assessment tools for management of water resources (Richards et al. 1997; Rothrock et al. 1998; Doledec et al. 1999; Charvat et al. 2000; Malmqvist 2002). To make effective use of biological communities it is important to have a good understanding of the mechanisms that lead to the occurrence and abundance of species within the environment (Richards et al. 1997; Manel et al. 2001). The application of indices in biomonitoring is widely accepted, but it is important to determine if indices perform with comparable precision across different environments i.e. between areas of different geology, river systems, and geographic locations.

Physical structure is one of the primary factors that determines the type of biological community present in a river (Winterbourn et al. 1981; Raven et al. 1998; Thompson & Townsend 2000). The distribution and abundance of benthic macroinvertebrates is influenced by a variety of physical and biological factors at multiple scales (Hynes 1970; Townsend et al. 1983; Harding & Winterbourn 1995; Richards et al. 1997). Land cover and geology are two of the major factors determining the physical nature of channel morphology and bed structure, as well as strongly influencing water chemistry and discharge (Hynes 1975; Osbourne & Wiley 1988; Close & Davies-Colley 1990; Church 2002). Thus, they can be important in determining invertebrate community composition (Maasdam & Smith 1994; Ometo et al. 2000).

At a local scale it is the stream habitat or riparian area that influences the structure and organization of biological communities (Harding et al. 1998; Scarsbrook & Halliday 1999). This area determines energy and matter inputs (Hynes 1975) as well as influencing habitat availability. In a pristine environment it has been suggested that local scale physical habitat will have a major influence over biotic assemblages at a site (Harding et al. 1998). It has been well established that riparian forests are effective in many stream environments as controllers of nutrient and sediment transport to streams from adjacent agricultural lands (Perry et al. 1999). Local geology has a large influence on the immediate environmental conditions at a site, influencing many aspects including flow regime, bed structure and channel shape (D'Angelo et al. 1997; Mosley & Jowett

1999; Newson & Newson 2000), which in turn influence communities (Huryn & Wallace 1987). Numerous studies have identified local scale variables as being important in influencing invertebrate communities (e.g. Huryn & Wallace 1987; Nerbonne & Vondracek 2001; Sponseller et al. 2001).

In considering larger spatial scales, such as ecoregions, much of the variation in species distribution can be attributed to patterns of variation within land cover and geology of a landscape (Richards et al. 1996; Allan et al. 1997; Allan & Johnson 1997). At a catchment scale, geology and land cover influence the hydrological regime (Hancock & Willgoose 2001; Church 2002) and water chemistry (Close & Davies-Colley 1990; Arheimer & Liden 2000; Hatch et al. 2001). A number of studies have shown surface geology within a catchment to account for a large proportion of variation in invertebrate distribution patterns (e.g. Corkum 1989; Richards et al. 1996). Large-scale human alterations, as measured by land cover, have major implications for aquatic systems (Rothrock et al. 1998; Harding et al. 1999). Changes in land cover can increase environmental variability, with areas of high anthropogenic use have more extremes i.e. a greater rang of temperatures, higher nutrient concentrations and low dissolved oxygen concentrations (Bargos et al. 1990; Richards et al. 1997).

The advancement in computing power over the last decade has made spatial analysis possible on a scale never considered before. The development of tools with in geographic information systems (GIS) allows calculations of both regional and local scale variables from digital information, reducing the need for full site surveys and allowing multiscale issues to be approached more efficiently (Allan & Johnson 1997). GIS tools have been widely applied to many ecological problems (e.g. Bellamy et al. 1998; Brown et al. 1998; Skinner et al. 2000; Fleishman et al. 2001; Gritzner et al. 2001; Jaberg & Guisan 2001; Wong 2002), with numerous studies now utilizing GIS to examine the riverine environment (e.g. Johnson & Gage 1997; Richards et al. 1997; Cooper et al. 1998; Doledec et al. 1999; Yang et al. 1999; Benke et al. 2000; Cresser et al. 2000; Manel et al. 2000; McDonnell 2000; Palik 2000; Sponseller et al. 2001). The recent availability of powerful statistical techniques has also led to a current expansion of predictive habitat distribution models in ecology (e.g. Guisan & Zimmermann 2000; Joy & Death 2000).

This chapter considers the application of digital information in identifying relationships between landscape variables, both anthropogenic and non-anthropologic, and biotic communities. An integration of both local and regional scales is used to examine their controls on community structure.

Study Site

This study was undertaken in the Whanganui Catchment (7100 km^2), in the North Island of New Zealand. The area was chosen because the component catchments cover a wide range of landscape classifications. The catchment varies in vegetation, from native bush and tussock through to exotic forest and intensive farming, with the lower altitude areas of the catchment more intensively developed particularly in terms of farming (Fig 1A). The dominant rock types include tertiary sandstone, mudstone and siltstone with streams around Mt Ruapehu flowing through bedrock of volcanic origin (Fig 1B).

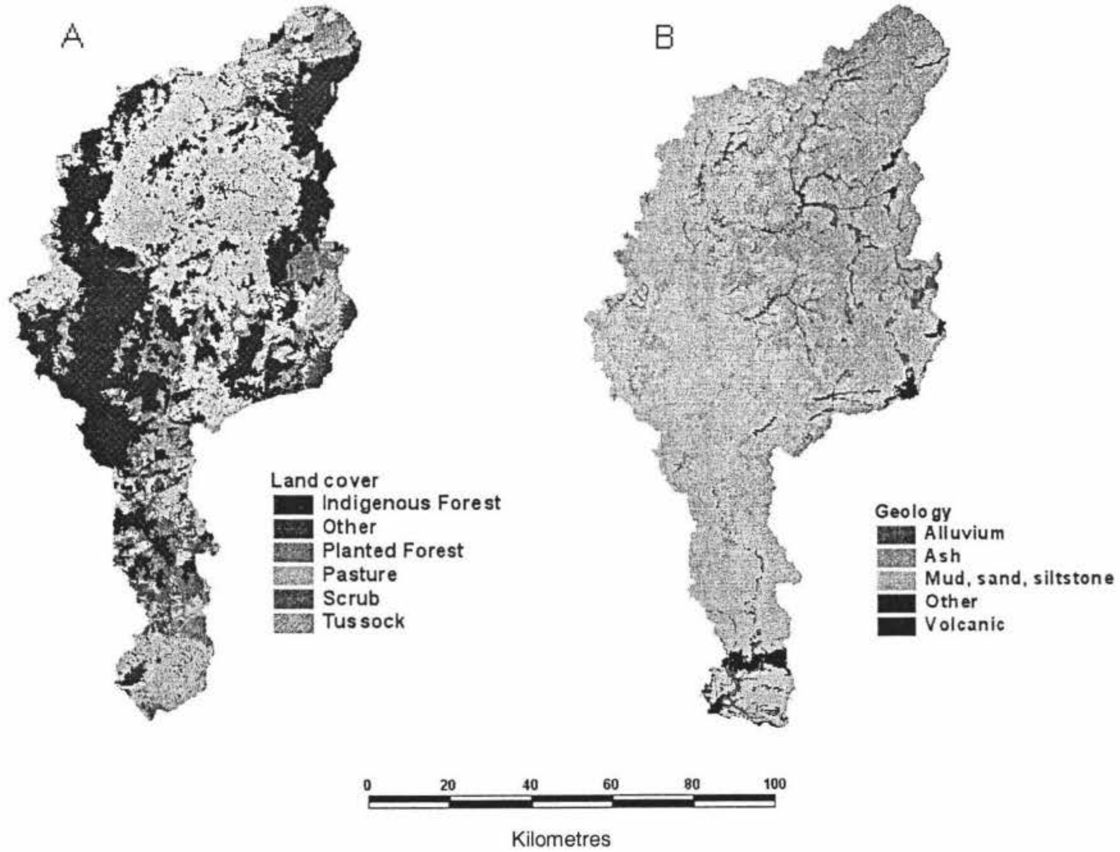


Figure 1: Land cover (A) and rock (B) of the Whanganui Catchment, New Zealand. Note urban land cover and Sods geology represent a very small proportion of the catchment, thus are not show in this map

Methods

Biological Methods

This study is based on invertebrate data from 6 different studies but with sampling and taxonomy methods comparable between studies (Department of Conservation 1988, Dunning 1998, Horrox 1998, Jowett et al. 2000, M.R. Scarsbrook National Institute of Water and Atmospheric Research unpublished data, J.D. Stark Cawthron Institute unpublished data). From these studies data from a total of 97 sites, in 3rd to 7th order rivers, was collected over 12 years, with 1 to 8 sampling occasions per site (a total of 138 sampling occasions) and each comprising at least 4 Surber samples. To standardise the data, information from only 4 Surber samples was randomly selected for each sampling occasion.

Community Composition

Variables used to assess community structure were total number of animals, species richness and the densities of those individual genera present at over 30% of the sites (*Aoteapsyche* sp., *Aphrophila neozelandica*, *Archichauliodes diversus*, *Austroclima* sp., *Austrosimulium* sp., *Coloburiscus humeralis*, *Deleatidium* sp., Eriopterini sp., *Hydrobiosis* sp., *Maoridiamesa* sp., *Nesameletus* sp. *Neurochorema confusum*, *Olinga feredayi*, *Oxyethira albiceps*, *Potamopyrgus antipodarum*, *Psilochorema* sp., *Pycnocentria* sp., *Pycnocentrodes* sp., *Tanytarsus vespertinus*, *Zelandobius* sp., *Zelandoperla* sp. and *Zephlebia* sp.). Individual taxa and total density were natural log transformed ($\log_e (x+1)$) to improve normality of the data. To examine overall community structure, detrended correspondence analysis (DECORANA) was carried out based on raw data. This ordination technique is designed to produce an objective arrangement of communities that reflect the similarities in their taxonomic composition (McCune and Mefford 1999). DECORANA analysis is widely used in stream ecology to examine community structure (e.g. Townsend et al. 1983, Furse et al. 1984, Harding 1994, Death 1996b, Gislason et al. 2001). In this study, two DECORANA axes (DEC1 and DEC2) of the full communities (as represented by taxa densities) are used.

Biotic Indices

An alternative way to characterise communities is through the use of biotic indices. Indices calculated (Chapter 2) included the water quality indices of Macroinvertebrate Community Index (MCI) (Stark 1985), Quantitative Macroinvertebrate Community Index (QMCI) (Stark 1993) and %EPT - *O. albiceps* (%EPT-Oxy). *O. albiceps* was removed from the EPT index calculation as it was relatively insensitive to changes in water quality. The diversity indices of Simpson's (Simpson 1949) for the whole community (Simp) and insect component (Isimp) and Margalef's (Marg) (Clifford & Stephenson 1975) were also derived (Appendix 1).

Land Quantification

Geographic Information Systems (GIS) were used to quantify land cover, rock and soil types in each basin. Digital land cover information was obtained from the land classification database (lcdb) (Terralink International Limited 2000) (Fig. 1A). Rock and soil data was taken from the land resource information (lri) (Ministry of Works and Development 1979) (Fig. 1B). Land cover was classified as plantation forest (PF), indigenous forest (IF), scrub (Scr), tussock (Tuss), primary pasture (PP), urban (Urb) and other land covers (OL), and broader groups of trees (indigenous and planted) and native vegetation (indigenous, scrub and tussock) (Nat). Rock was grouped into 5 classes; ash, alluvium (All), mud, sand and siltstone (MSS), volcanic (Vol), and other rock (OR). Soils were classified as pumice (Pum), loams, recent (Rec), mineral (Min), and other soils (OS). Rock and soil are discussed in combination as geology. Change in pasture (CiP) was the only temporal variables calculated. It was determined from the change in pasture cover between the land resources information (1979) and the land classification database (2000).

Catchments were delineated using a digital elevation grid created from 20 metre contours and spot height information in ArcView (Environmental Systems Research Institute 1999). From the elevation grid, flow direction and flow accumulation grids were generated, permitting the delineation of catchments and calculation of stream networks. This allowed the calculation of a polygon for each catchment, which included the entire drainage area upstream of the sample point. Landscape variables were calculated as catchment and riparian percent cover. To calculate the percentage land cover and geology type the catchment polygons and riparian polygons were overlaid on the lcdb and lri. The riparian area was defined by buffering the rivers by 100 metres either side, buffering the sample point by 800 meters and clipping the intersect. The intersect polygon was then clipped by the catchment area to give the riparian area (Fig. 2). This particular size was selected because of the limitations in the accuracy of the data, this was the smallest viable area.

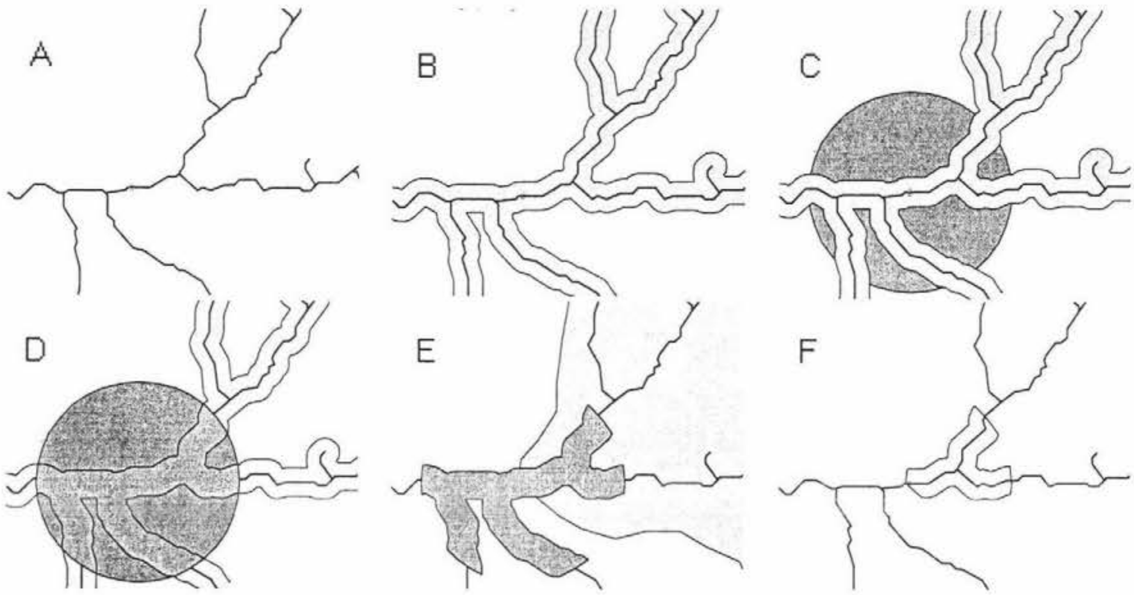


Figure 2: Derivation of riparian area. A – river and invertebrate sampling site, B- river buffered by 100m either side, C – site buffered by 800m, D – clip river buffer by site buffer, E – clipped area and catchment area, F – final clipped area.

Statistical Methods

Once all variables and biological ordinations had been calculated data was checked for univariate and multivariate normality and variables transformed if necessary. The initial stage of analysis considers variable reduction. Multicolinearity was examined in the landscape variables, retain only one variable of each highly correlated ($r>0.7$) group for further analysis. Standardised principal component analysis was carried out on the remaining landscape variables to determine if further reduction was useful.

The relationship between individual landscape and biological variables was explored with Pearson's correlations. Canonical correlation analyses is widely used in ecological data to explore links between two sets of information (Ohmann & Spies 1998; Zeilhofer & Schessl 2000; Griffith et al. 2001; Lods-Crozet et al. 2001) and in this study they are used to examine the relationship between all the landscape variables and two components of the biological communities (biotic indices and community structure). To examine the impact of catchment and riparian variables independently, two further canonical correlations were considered for each component; catchment landscape variables only and riparian landscape variables only.

Multiple regressions were utilized to examine the effect of all landscape variables on each biological variable (see Chapter 2 for further detail). Multiple regression was done using a stepwise method with p=0.05 this prevents autocorrelation within each of the models. Stepwise method was used as it was not practical to examine all possible combinations of variables in the regression models and the stepwise method gives then next best results.

Results

Summary Statistics

No one land cover dominated the data. Indigenous forests had the highest mean percent cover (28%) at both the riparian and catchment scale (Fig. 3). Percentage tussock and pasture showed the most variation between the catchment and riparian scale with more tussock cover at the catchment scale and more pasture at the riparian scale (Fig. 3).

Percent ash dominated the catchment scale rock (Fig. 4). At the riparian level alluvium and ash rock made up about the same percent cover (Fig. 4).

The main difference in soil types between the catchment and riparian scale was the presences of mineral soils at the catchment scale and not at the riparian scale (Fig. 5). Pumice soils had the highest mean percent cover at both scales (Fig. 5).

Multicolinearity in the data

Multicolinearity was examined as variables that are highly correlated will show very similar relationships, and in canonical correlation they may bias the axis and make multiple regression harder to interpret. A number of the landscape variables were correlated with each other (Appendix 9). Percentage riparian planted forest was highly positively correlated with percentage of total planted forest ($r=0.81$). Percentage bare ground land cover was positively correlated with percent mineral soils ($r=0.95$) and percentage volcanic rock ($r=0.91$). Percentage ash was negatively correlated with the percentage of other soils ($r=-0.87$) and percent mud, sand and siltstone ($r=-0.84$). Percentage of other soils was also negatively correlated with percent pumice ($r=-0.81$). Riparian tree cover positively correlated with riparian indigenous forest ($r=0.77$).

To reduce the number of variables used in further analysis one of each highly correlated ($r>\pm0.70$) pair was retained. Volcanic rock, mineral soils, riparian planted forest, other soils, mud, sand and siltstone rock and riparian tree cover were removed from further analysis.

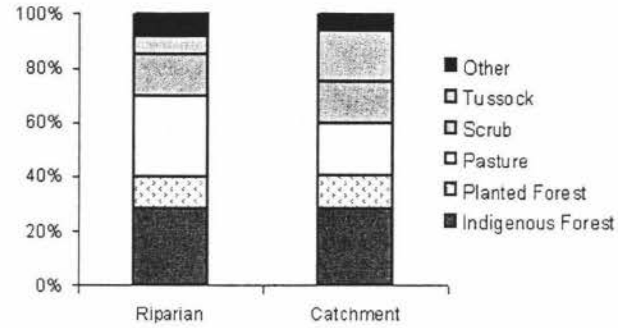


Figure 3: Average percentage land cover in riparian and catchment areas of sites sampled for invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand. (Other = Waterways, Wetland, Urban and Bare Ground).

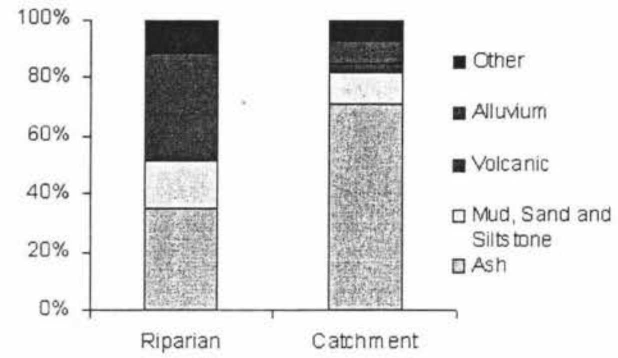


Figure 4: Average percentage rock base in riparian and catchment areas of sites sampled for invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand (Ash = ashes older than the Taupo eruption, Kaharoa and Taupo ashes and Ngauruhoe ash, Mud, sand and siltstone = sandstone or siltstone massive, mudstone or siltstone joined, mudstone or fine siltstone massive and mudstone or siltstone banded, Volcanic = welded volcanic rocks, Other = greywacke, peat, Taupo and Kohara breccia).

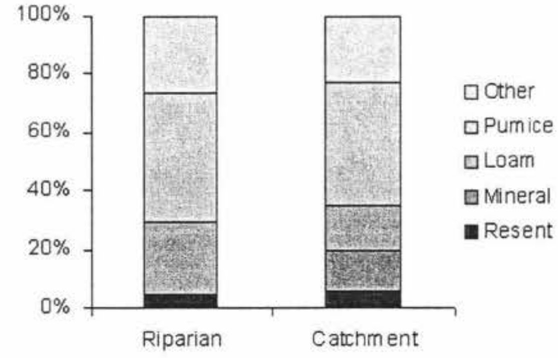


Figure 5: Average percentage soil cover in riparian and catchment areas of sites sampled for invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand (Other = bare rock, peaty, earths and podsol).

Principal component analysis on the 31 remaining variables yielded a new axis that only explained 19% of the variation in the data. Together the first 3 axes explain 45% of the variation in the landscape data. Axis 1 is a representation of volcanic areas, with percent pumice, ash, bare ground (volcanic) and tussock all having strong positive eigenvectors. Axis 2 and 3 represents anthropogenic use and riparian geology, with axis 2 relating to pastoral use and axis 3 forestry (Table 1).

Table 1: Eigenvectors for a principal component analysis on the landscape variables at 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand (shading indicates strongest links between landscape variables and principal components).

	PC1	PC2	PC3
CiP	-0.02	0.23	0.36
RCiP	-0.14	0.20	0.07
BG	0.27	0.06	0.03
IF	-0.08	-0.08	0.22
PF	-0.12	-0.15	-0.46
PP	-0.24	0.26	0.21
Scr	0.10	-0.15	0.16
Tuss	0.34	0.06	-0.11
Urb	0.02	0.25	0.05
RIF	0.16	-0.10	0.25
RPP	-0.22	0.33	0.06
RScr	0.10	-0.20	-0.03
RTuss	0.23	0.02	-0.07
Rash	0.16	0.04	0.19
Rall	0.06	0.30	-0.22
RMSS	-0.09	-0.24	0.19
Ash	0.27	0.27	0.05
All	0.18	0.10	-0.21
OR	-0.04	0.15	-0.06
Pum	0.31	0.09	-0.10
Loam	-0.15	0.24	0.19
Rec	0.20	0.07	0.07
RRes	-0.07	0.13	0.04
RLoam	0.12	-0.03	0.15
RPum	0.22	0.26	-0.11
RSO	-0.13	-0.21	0.25
Rnat	0.31	-0.22	0.16
nat	0.26	-0.13	0.21
Tree	-0.17	-0.20	-0.22

The plot of the first two principal components (PC1 and PC2) shows sites with more intensive human activity in the top of the figure and those to the right in the upper volcanic regions (Fig. 6).

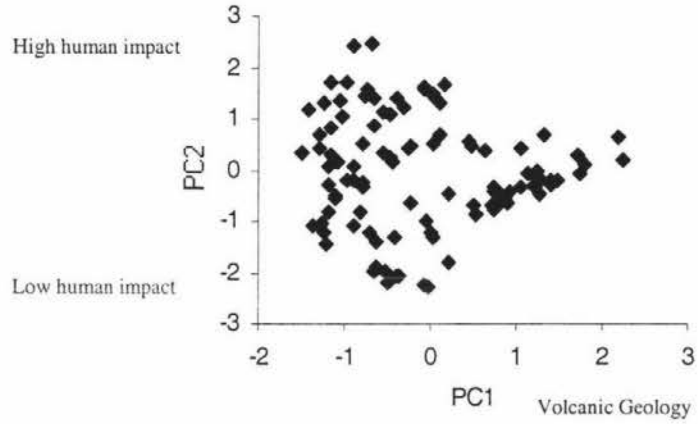


Figure 6: Plot of the first principal components derived from landscape characteristics of 173 sites in the Whanganui Catchment, New Zealand.

Univariate Correlations

Overall, 16 of the catchment variables and principal component scores were correlated with at least one biotic measure (Table 2). Margalef's index was the only diversity indices to be strongly correlated with any landscape variables. It was positively correlated with percent native vegetation, percent pumice soils, and PC1 and negatively correlated with percent planted forest. Percentage of native vegetation was positively linked with EPT, MCI and QMCI, and negatively linked with the first DECORANA axis. Nine of the land cover variables showed strong links with one or more of the indices with 3 geology variables also showing strong links.

Of the individual taxa, 10 of the 22 taxa were linked with more than one landscape variable (Table 2). *P. antipodarum* had a strongly positive relationship with percent primary pasture and a negative link with percent riparian indigenous forest, percent tussock, ash, native, pumice and the PC1. Percent native vegetation and percent ash showed the most links, being strongly correlated with 4 of the taxa. Nine land cover variables showed strong links with one or more of the taxa considered, 3 geology variables also showed strong links.

Table 2: Pearson's Correlation between landscape and biotic variables (only those with significant relationships are presented), for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand (+ or - significant) (all correlation values are given in Appendix 10).

	CiP	BG	IF	PF	PP	Tuss	Urb	nat	RIF	Rnat	Ash	Pum	RPum	PC1	PC2	PC3
Marg				-				+				+		+		
MCI					-			+		+				+		
QMCI							-	+								
EPT-Oxy							-	+		+						
DEC1		-		+		-	-	-			-	-	-	-	-	
Total number of animals	+															
Aoteapsyche sp.				-			+									
Aphrophila neozelandica											+	+	+	+		
Deleatidium sp.				-				+		+						+
Hydrobiosis sp.							+									
Maoridiamesa sp.											+					
Nesameletus sp.			+					+								
Potamopyrgus antipodarum		-			+	-		-	-	-	-	-	-	-		
Pycnocentria sp.			+													
Tanytarsus vespertinus							+									
Zelandoperla sp.					-	+		+		+	+	+	+	+		

Canonical Correlations

Canonical correlation analyses were used to determine a linear combination of landscape variables to predict a linear combination of the biotic variables. Thus, instead of using individual correlation as in the previous section this examines the data set as a whole.

Biotic Indices

Canonical correlation of the biotic indices with the landscape variables indicated 4 significant landscape axes explained 42% of the variation in the indices (Table 3). Overall the landscape variables explained 46% of the variation in the indices.

Table 3: Canonical correlation analysis of biotic indices and landscape variables for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

Canonical Variable	Canonical Correlation Co-efficient	Pr > F	Cumulative % explained of indices by indices canonical variants	Cumulative % explained of indices by landscape canonical variants
1	0.82	<0.0001	0.50	0.34
2	0.63	<0.0001	0.56	0.36
3	0.60	<0.0001	0.61	0.38
4	0.57	0.0060	0.74	0.42
5	0.42	0.6300	0.91	0.45
6	0.25	0.9700	1.00	0.46

Axis 1 was dominated by a weighting of Margalef's index, MCI, EPT and QMCI. The second is a contrast of Simpson's Indices and EPT with MCI, the third is dominantly QMCI with the fourth a contrast between Margalef's and the water quality indices (Appendix 11). The landscape canonical axes show axis 1 dominated by native vegetation, axis 2 a contrast between volcanic derived landscape and scrub, axis 3 is a weighting of urban and ash, with the fourth axis a weighting of alluvium and anthropogenic use (Appendix 12).

This suggests that richness (Margalef's) and water quality indices, increase with higher percentage of native vegetation. Dominance (Simpson's) increases with a higher percent of scrub cover or lower in percentage of ash. QMCI decreases with higher percentage of urban areas. The univariate analysis suggests water quality and Margalef's indices are well explained by the landscape variables but Simpson's indices are not (Table 4).

Table 4: Multiple regression statistics for predicting the biotic indices from the landscape canonical variables for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

	R-Square	Adjusted R-Square	F Value	Pr > F
Isimp	0.26	0.13	1.97	0.0060
Simp	0.27	0.14	2.03	0.0050
Imarg	0.57	0.49	7.47	<0.0001
MCI	0.58	0.51	7.86	<0.0001
QMCI	0.52	0.43	6.03	<0.0001
EPT-Oxy	0.53	0.45	6.38	<0.0001

Canonical correlation between the biological indices and the riparian and catchment variables was carried out separately. The combination of riparian and landscape variables explained 42% of the variation in the indices while catchment variables alone explained 34% (Appendix 13) and riparian variables alone explained 23%.(Appendix 14). From the correlations between the individual indices and landscape canonical variables it can be seen that the water quality indices produce stronger models with the catchment variables and the diversity indices produce stronger models with the riparian variables (Table 5).

Table 5: Multiple regression statistics for predicting the biotic indices from the landscape canonical variables for catchment and riparian variables for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

	Catchment				Riparian			
	R-Square	Adjusted R-Square	F Value	Pr > F	R-Square	Adjusted R-Square	F Value	Pr > F
Isimp	0.16	0.08	2.1	0.0150	0.18	0.12	2.98	0.0009
Simp	0.15	0.08	2.06	0.0170	0.19	0.13	3.21	0.0004
Imarg	0.42	0.37	8.3	<0.0001	0.41	0.36	9.17	<0.0001
MCI	0.48	0.44	10.49	<0.0001	0.37	0.33	7.94	<0.0001
QMCI	0.44	0.40	9.03	<0.0001	0.2	0.14	3.34	0.0002
EPT-Oxy	0.45	0.40	9.1	<0.0001	0.29	0.24	5.51	<0.0001

Community Structure

Canonical correlation of the invertebrate densities with the landscape variables showed the first 8 axes to be significant (P<0.01) with these 8 landscape axes explaining 32% of the variation in the community (Table 6). Overall, the landscape variables explained 38% of the variation in community structure. The first 3 axes explain the majority of this variation (24%) in the community structure and thus are considered in more detail.

Table 6: Canonical analysis of taxa density and landscape variables for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

Canonical Variable	Canonical Correlation Co-efficient	Pr > F	Cumulative % explained of the community by community canonical variants	Cumulative % explained of the community by landscape canonical variants
1	0.91	<0.0001	0.14	0.12
2	0.85	<0.0001	0.21	0.17
3	0.80	<0.0001	0.32	0.24
4	0.75	<0.0001	0.36	0.26
5	0.72	<0.0001	0.41	0.29
6	0.68	<0.0001	0.44	0.30
7	0.66	<0.0001	0.47	0.31
8	0.62	0.0052	0.50	0.32
9	0.60	0.0665	0.53	0.34
26	0.04	0.652	1.00	0.38

Community axis 1 is a contrast of *A. neozelandica*, *Deleatidium* sp. and *Zelandoperla* sp. with *P. antipodarum* and DEC1. Axis 2 is a contrast of *Aoteapsyche* sp., *P. antipodarum*, DEC2 and total number of animals with *Pycnocentria* sp. and *Zelandoperla* sp. Axis 3 is a weighting of *A. diversus*, *C. humeralis*, *Nesameletus* sp., *Pycnocentria* sp. and *Zephlebia* sp. (Appendix 15). The landscape axis 1 shows a contrast of native and planted vegetation, whereas axis 2 represents anthropogenic use and axis 3 increasing altitude (Appendix 16).

This suggest *A. neozelandica*, *Deleatidium* sp. and *Zelandoperla* sp. are more likely to be found in areas of native vegetation and *P. antipodarum* is more likely in areas of plantation forestry. *Aoteapsyche* sp., *P. antipodarum* and total number of animals are higher in areas with more human development. Overall, DEC1, *A. neozelandica*, *P. antipodarum* and *Zelandoperla* sp. have the strongest relationships with the landscape canonical variants (Table 7).

Table 7: Multiple regression statistics for predicting community structure from the landscape canonical variables for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

	R-Square	Adjusted R-Square	F Value	Pr > F
DEC1	0.70	0.64	12.84	<0.0001
DEC2	0.48	0.39	5.22	<0.0001
NoSp	0.37	0.26	3.33	<0.0001
density	0.35	0.23	3.01	<0.0001
Aoteapsyche sp.	0.42	0.31	4.02	<0.0001
Aphrophila neozelandica	0.38	0.27	3.43	<0.0001
Archichauliodes diversus	0.46	0.37	4.84	<0.0001
Austroclima sp.	0.32	0.19	2.59	0.0002
Austrosimulium sp.	0.21	0.07	1.50	0.0690
Coloburiscus humeralis	0.38	0.27	3.51	<0.0001
Deleatidium sp.	0.46	0.36	4.75	<0.0001
Eriopterini sp.	0.23	0.10	1.71	0.0254
Hydrobiosis sp.	0.39	0.28	3.62	<0.0001
Maoridiamesa sp.	0.30	0.17	2.37	0.0007
Nesameletus sp.	0.52	0.44	6.15	<0.0001
Neurochorema confusum	0.33	0.21	2.77	<0.0001
Olinga feredayi	0.45	0.35	4.53	<0.0001
Oxyethira albiceps	0.26	0.13	1.97	0.0063
Potamopyrgus antipodarum	0.54	0.46	6.54	<0.0001
Psilochorema sp.	0.24	0.10	1.77	0.0187
Pycnocentria sp.	0.45	0.35	4.53	<0.0001
Pycnocentrodes sp.	0.34	0.22	2.88	<0.0001
Tanytarsus vespertinus	0.31	0.18	2.48	0.0003
Zelandobius sp.	0.17	0.02	1.14	0.3094
Zelandoperla sp.	0.50	0.41	5.53	<0.0001
Zephlebia sp.	0.42	0.32	4.14	<0.0001

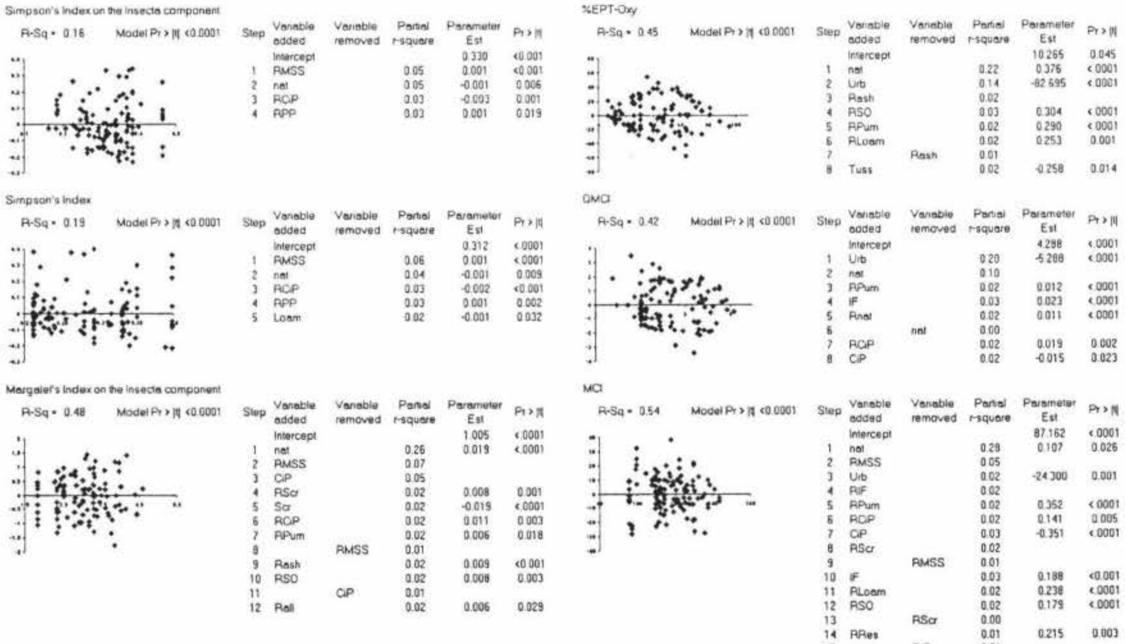
Canonical correlation between the community structure and the riparian and catchment variables was carried out separately. The combination of riparian and landscape variables explained 34% of the variation in the community while catchment variables alone explained 25% (Appendix 17) and riparian variables alone explained 17% (Appendix 18). All community measures have higher r-squared values with the catchment variables apart from *Zephlebia* sp. and *Austroclima* sp. (Table 8).

Table 8: Multiple regression statistics for predicting the community structure from the landscape canonical variables for catchment and riparian for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

	Catchment				Riparian			
	R-Square	Adjusted R-Square	F Value	Pr > F	R-Square	Adjusted R-Square	F Value	Pr > F
DEC1	0.59	0.56	16.41	<0.0001	0.44	0.40	10.64	<0.0001
DEC2	0.28	0.21	4.32	<0.0001	0.25	0.19	4.46	<0.0001
NoSp	0.24	0.17	3.53	<0.0001	0.24	0.19	4.3	<0.0001
density	0.28	0.22	4.49	<0.0001	0.24	0.18	4.14	<0.0001
Aoteapsyche sp.	0.33	0.28	5.67	<0.0001	0.27	0.22	4.98	<0.0001
Aphrophila neozelandica	0.33	0.27	5.5	<0.0001	0.25	0.19	4.34	<0.0001
Archichauliodes diversus	0.30	0.24	4.88	<0.0001	0.24	0.19	4.32	<0.0001
Austroclima sp.	0.14	0.06	1.77	0.0474	0.16	0.10	2.51	0.0048
Austrosimulium sp.	0.13	0.05	1.7	0.0610	0.11	0.04	1.64	0.0850
Coloburiscus humeralis	0.27	0.21	4.18	<0.0001	0.24	0.19	4.32	<0.0001
Deleatidium sp.	0.38	0.33	6.92	<0.0001	0.31	0.26	6.04	<0.0001
Eriopterini sp.	0.20	0.13	2.76	0.0011	0.05	0.00	0.66	0.7854
Hydrobiosis sp.	0.26	0.19	3.95	<0.0001	0.25	0.20	4.5	<0.0001
Maoridiamesa sp.	0.22	0.15	3.18	0.0002	0.18	0.12	2.94	0.0010
Nesameletus sp.	0.44	0.39	8.76	<0.0001	0.15	0.09	2.35	0.0084
Neurochorema confusum	0.19	0.12	2.71	0.0013	0.09	0.03	1.39	0.1779
Olinga feredayi	0.36	0.31	6.47	<0.0001	0.16	0.10	2.52	0.0046
Oxyethira albiceps	0.20	0.13	2.81	0.0009	0.10	0.03	1.44	0.1524
Potamopyrgus antipodarum	0.41	0.35	7.71	<0.0001	0.41	0.36	9.12	<0.0001
Psilochorema sp.	0.16	0.09	2.19	0.0101	0.13	0.06	1.97	0.0303
Pycnocentria sp.	0.27	0.20	4.14	<0.0001	0.22	0.16	3.71	<0.0001
Pycnocentrodes sp.	0.26	0.20	4.02	<0.0001	0.18	0.12	2.88	0.0013
Tanytarsus vespertinus	0.20	0.13	2.79	0.0010	0.12	0.06	1.9	0.0375
Zelandobius sp.	0.12	0.05	1.59	0.0885	0.11	0.05	1.71	0.0691
Zelandoperla sp.	0.43	0.38	8.66	<0.0001	0.36	0.31	7.45	<0.0001
Zephlebia sp.	0.22	0.15	3.17	0.0002	0.26	0.20	4.65	<0.0001

Multiple Regressions

To examine the relationships between the landscape variables and each of the biotic variables, stepwise multiple regressions were used. All index multivariate models were highly significant (P<0.001) with r-squared values ranging from 0.16 to 0.54 (Fig. 7). Insect Simpson's Index and Simpson's Index models were significant but the overall explanation power of these models was low, with the highest r-squared only 0.19. Margalef's index was one of the best models (r^2=0.48), showing richness to increase with percentage of catchment in native vegetation. MCI, QMCI and EPT all had strong models and were dominated by land cover, variables increasing with percent native vegetation and decreasing with percent urban areas (Fig. 7). Of the 16 land cover variables, 10 were used in at least one of the index models, with percentage urban, native cover and riparian change in pasture being incorporated in the majority of models. Of the 13 geology variables, 7 were used in the index models with percentage riparian pumice in 4 of the 6 models.



Simpson's Index on the Insecta component — R-Sq = 0.16, Model Pr > |t| <0.0001

Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			0.330	<0.001
1	RMSS		0.05	0.001	<0.001
2	nat		0.05	-0.001	0.006
3	RCiP		0.03	-0.003	0.001
4	RPP		0.03	0.001	0.019

Simpson's Index — R-Sq = 0.19, Model Pr > |t| <0.0001

Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			0.312	<.0001
1	RMSS		0.06	0.001	<.0001
2	nat		0.04	-0.001	0.009
3	RCiP		0.03	-0.002	<0.001
4	RPP		0.03	0.001	0.002
5	Loam		0.02	-0.001	0.032

Margalef's Index on the Insecta component — R-Sq = 0.48, Model Pr > |t| <0.0001

Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			1.005	<.0001
1	nat		0.26	0.019	<.0001
2	RMSS		0.07		
3	CiP		0.05		
4	RScr		0.02	0.008	0.001
5	Scr		0.02	-0.019	<.0001
6	RCiP		0.02	0.011	0.003
7	RPum		0.02	0.006	0.018
8		RMSS	0.01		
9	Rash		0.02	0.009	<0.001
10	RSO		0.02	0.008	0.003
11		CiP	0.01		
12	Rall		0.02	0.006	0.029

%EPT-Oxy — R-Sq = 0.45, Model Pr > |t| <0.0001

Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			10.265	0.045
1	nat		0.22	0.376	<.0001
2	Urb		0.14	-82.695	<.0001
3	Rash		0.02		
4	RSO		0.03	0.304	<.0001
5	RPum		0.02	0.290	<.0001
6	RLoam		0.02	0.253	0.001
7		Rash	0.01		
8	Tuss		0.02	-0.258	0.014

QMCI — R-Sq = 0.42, Model Pr > |t| <0.0001

Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			4.288	<.0001
1	Urb		0.20	-5.288	<.0001
2	nat		0.10		
3	RPum		0.02	0.012	<.0001
4	IF		0.03	0.023	<.0001
5	Rnat		0.02	0.011	<.0001
6		nat	0.00		
7	RCiP		0.02	0.019	0.002
8	CiP		0.02	-0.015	0.023

MCI — R-Sq = 0.54, Model Pr > |t| <0.0001

Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			87.162	<.0001
1	nat		0.28	0.107	0.026
2	RMSS		0.05		
3	Urb		0.02	-24.300	0.001
4	RIF		0.02		
5	RPum		0.02	0.352	<.0001
6	RCiP		0.02	0.141	0.005
7	CiP		0.03	-0.351	<.0001
8	RScr		0.02		
9		RMSS	0.01		
10	IF		0.03	0.188	<0.001
11	RLoam		0.02	0.238	<.0001
12	RSO		0.02	0.179	<.0001
13		RScr	0.00		
14	RRes		0.01	0.215	0.003
15		RIF	0.01		
16	Rall		0.02	-0.080	0.017

Figure 7: Stepwise multiple regressions of indices on full landscape variable set for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

The DECORANA axes had strong models with r-square values of 0.65 and 0.42. Overall the DECORANA axes, species richness and total number of animal models incorporated 9 land cover variables with the geology variables being predominately riparian not catchment orientated. Percentage urban and change in pasture were the most common variables incorporated in 3 out of the 4 models.

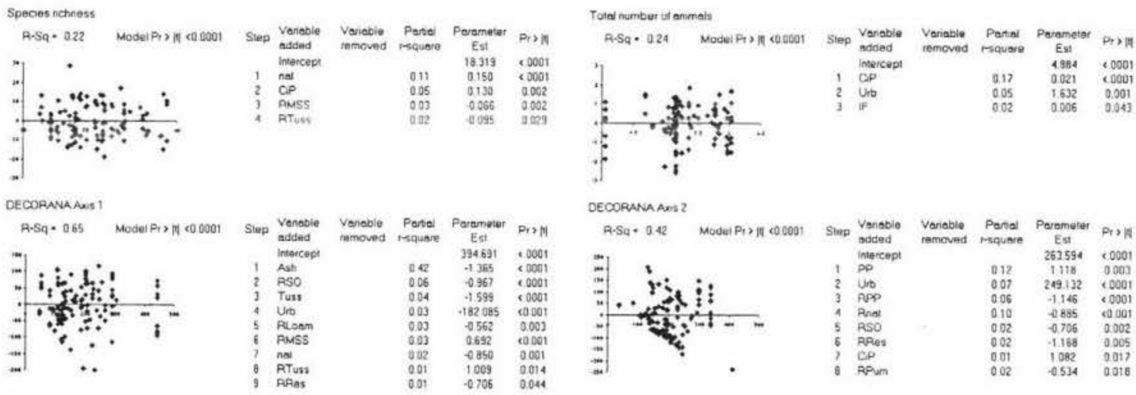
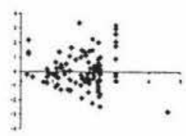


Figure 8: Stepwise multiple regressions of species richness, total number of animals and DECORANA axes on full landscape variable set for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

The densities of individual taxa showed less distinct patterns, often with the residuals lacking normality (Fig. 9). The r-squares of the models ranged from 0.09 to 0.46 with *Nesameletus* sp., *Zelandoperla* sp. and *P. antipodarum* having r-squares over 0.4. *Nesameletus* sp. increased with percentage indigenous forest and native vegetation cover, decreasing with percent urban, riparian mud, sand and siltstone and alluvium. *P. antipodarum* densities decrease with high percent native vegetation and pumice in the riparian areas and increased with percent pasture and scrub. *Zelandoperla* sp. densities increase with percent pumice, native vegetation and bare ground, decreasing with percent urban and riparian mud, sand and siltstone. Overall, the density models incorporated most of the geology variables in at least one model, and all of the land cover variables. Percent urban cover was the most widely used variable, being incorporated into 9 of the 22 models.

Aoteapsyche sp.

R-Sq = 0.34 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			1.746	<.0001
1	Urb		0.13	1.781	0.008
2	PF		0.09	-0.018	<.0001
3	Loam		0.04		
4	All		0.02	-0.030	0.003
5	RCiP		0.02	-0.020	0.001
6	RPP		0.03	0.011	<0.001
7		Loam	0.02		
8	RSO		0.03	-0.007	0.012

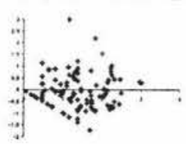
Aphrophila neozelandica

R-Sq = 0.25 Model Pr > |t| <0.0001

Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			0.300	0.003
1	Ash		0.25	0.011	<.0001

Archichauliodes diversus

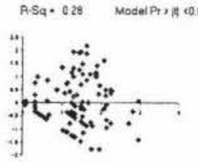
R-Sq = 0.26 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			0.866	<.0001
1	CiP		0.08	0.015	<.0001
2	Rec		0.07	-0.031	<.0001
3	nat		0.03	0.012	<.0001
4	Pum		0.04	-0.011	<.0001
5	RMSS		0.05	-0.007	0.001

Coloburiscus humeralis

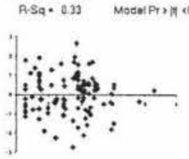
R-Sq = 0.28 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			-0.012	0.959
1	IF		0.11	0.014	<0.001
2	RPum		0.07	0.006	0.008
3	Rash		0.04	0.007	0.011
4	Rec		0.03	-0.027	0.006
5	RPP		0.02	0.009	0.004
6	Rnat		0.02	0.006	0.041

Deleatidium sp.

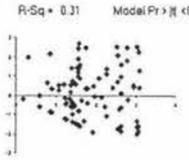
R-Sq = 0.33 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			0.215	0.488
1	nat		0.24	0.022	<.0001
2	PP		0.03		
3	Rnat		0.02	0.017	<.0001
4	RPP		0.02	0.013	<0.001
5		PP	0.01		
6	Pum		0.02	-0.008	0.030

Hydrobiosis sp.

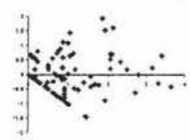
R-Sq = 0.31 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			1.019	<.0001
1	Urb		0.15	2.036	<.0001
2	PF		0.07	-0.009	<0.001
3	RMSS		0.06	-0.008	<.0001
4	RSO		0.03	0.005	0.007

Nesameletus sp.

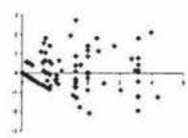
R-Sq = 0.42 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			-0.066	0.668
1	IF		0.28	0.016	<.0001
2	nat		0.05	0.011	<.0001
3	Urb		0.03	-1.327	0.001
4	RMSS		0.03	-0.004	0.029
5	All		0.02	-0.023	0.003
6	Rall		0.01	0.004	0.042

Olinga feredayi

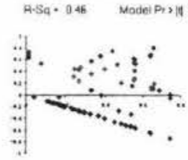
R-Sq = 0.34 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			0.162	0.308
1	RPum		0.08	0.009	0.001
2	IF		0.09	0.010	0.001
3	All		0.06	-0.034	<.0001
4	Urb		0.04	-2.672	<.0001
5	Ash		0.04	0.010	0.000
6	RTuss		0.04	-0.013	0.003

Potamopyrgus antipodarum

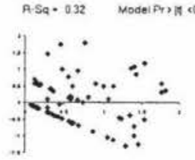
R-Sq = 0.46 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			2.391	<.0001
1	Rnat		0.22	-0.023	<.0001
2	RPum		0.11	-0.008	0.002
3	PP		0.03	0.023	<.0001
4	RPP		0.04	-0.014	<0.001
5	Scr		0.03	0.014	0.012
6	RScr		0.02	0.007	0.028

Pycnocentria sp.

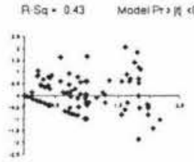
R-Sq = 0.32 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			-0.032	0.823
1	IF		0.15	0.016	<.0001
2	RSO		0.05	0.009	<0.001
3	Urb		0.04	-1.501	0.002
4	RPum		0.04	0.007	0.001
5	RCiP		0.03	0.010	0.004
6	OR		0.02	-0.101	0.044

Zelandoperla sp.

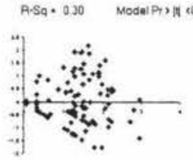
R-Sq = 0.43 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			0.151	0.366
1	Pum		0.22	0.007	0.011
2	nat		0.09	0.012	<.0001
3	BG		0.04	0.034	0.001
4	RMSS		0.04	-0.008	<0.001
5	Urb		0.03	-1.435	0.003
6	RIF		0.02	0.005	0.022

Zephlebia sp.

R-Sq = 0.30 Model Pr > |t| <0.0001



Step	Variable added	Variable removed	Partial r-square	Parameter Est	Pr > |t|
	Intercept			0.200	0.427
1	Ash		0.11	-0.009	<0.001
2	RPP		0.04	0.014	<.0001
3	RScr		0.06	0.010	<.0001
4	Rash		0.04	0.011	<.0001
5	RMSS		0.03	0.008	0.002
6	CiP		0.02	-0.011	0.025

Figure 9: Stepwise multiple regressions of taxon densities on full landscape variable set for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand – only models with r-squared values over 0.25 are shown.

Discussion

This study has explored 4 questions; 1) is land cover or geology more closely linked with the benthic biota, 2) what scale (catchment or riparian) is most appropriate when considering digitally derived landscape variables, 3) are biotic indictors of degradation applicable over a wide range of geological environments, and 4) how useful are statistical tools (variable reduction and multiple regression) and geographic information systems in understanding biotic systems.

Is land cover or geology more closely linked with benthic biota?

Richards et al. (1996) and other studies (e.g. Corkum 1989) identified catchment geology as an important controlling factor in benthic invertebrate communities. Many other studies have examined the effect of land cover on the benthic biota and found it to be important in structuring communities (Harding & Winterbourn 1995; Roth et al. 1996; Allan et al. 1997; Harding et al. 1998; Manel et al. 2000). The ordination analysis in this study suggested the biological communities were responding to a number of geological (rock and soil) and land cover variables. Of the individual correlations between the landscape and biotic variables, land cover showed stronger links than either rock or soil type. In particular, percentage native cover had the greatest number of links with the biotic variables. However, it was not possible to determine land cover or geology as having overall importance with the combination of both giving the strongest results.

What scale (catchment or riparian) is most appropriate when considering digitally derived landscape variables?

Hynes (1975) stated that the "valley rules the stream", suggesting the dominance of the catchment. A number of other studies have identified local habitat characteristics as being an important influence on biotic communities (Richards et al. 1997; Downes et al. 1998; Harding et al. 1998; Lammert & Allan 1999; Scarsbrook & Halliday 1999). This study explored both perspectives through integration of catchment and local scale variables. The use of catchment, riparian and a combination of these variables was used to determine what scale was most appropriate in the application of data sets applied in this study. The combination of both riparian and catchment data gave better predictions than either did individually, suggesting that biotic variables are influenced at both scales. However, of the two scales, catchment variables had more and stronger individual correlations with the biotic variables. Canonical correlations reinforced this showing catchment variables to be more strongly linked with the biota than the riparian scale, though identifying diversity indices to be more influenced by the local scale. This suggests that overall catchment variables may be more influential on community structure than riparian variables; the valley does rule the stream. Other studies have found similar results, with catchment characteristics being better predictors of biotic communities (Lenat & Crawford 1994; Quinn et al. 1997).

Although the dominance of catchment scale variables in this study contrasts with the findings of Richards et al. (1997), who identified stronger models with local scale variables. However, Richards only considered the occurrence of taxa, not abundance, and grouped them according to species traits; this study has utilized taxa densities, indices and ordinations. This suggests that some traits may be more closely linked to local scale variables, but community structure as a whole may not be. Richards measured riparian scale variables in the field where as this study used digitally derived data. It may be that local scale variables cannot be adequately derived from currently available digital data, and that finer detail is imperative for accurate assessment at a local scale. Catchment scale models developed in this study show a similar goodness of fit (r-squared) to Richards's riparian models, suggesting both scales may be equally important. Sponseller et al. (2001) also identified local conditions as the dominating influence on invertebrate community composition, finding temperature to be the best predictor variable, with 200m riparian areas also showing strong relationships with invertebrate communities as measured by diversity and water quality indices.

It may be that different forces control New Zealand streams invertebrate community to those in the rest of the world as New Zealand hydrology is considerable different. The relatively low amounts of allochthonous organic matter and thus importance to stream community in comparison to other countries may weaken the link with the riparian environment (Winterbourn et al 1981; Linklater & Winterbourn 1993). Therefore, in New Zealand it is likely that the catchment may be a more important influence on community structure than local area. The results from this and other studies suggest that it is important to consider the environment on many scales.

Are biotic indictors of degradation applicable over a range of geological environments?
Biomonitoring is now widely used to measure the health of river systems (e.g. Hannaford & Resh 1995; Fore et al. 1996; Reynoldson et al. 1997; Stark 1998; Doledec et al. 1999). It is important to understand how indices respond over a variety of landscapes before they are applied to management decisions. This study explored three indices widely used in New Zealand for monitoring river systems; MCI, QMCI and EPT. They were found to be higher with increasing percent cover of native vegetation and scrub, decreased with increased percent of pasture and urban land cover. They also had strong positive links with percent ash rock and pumice soils. Other measures of community structure including species richness and total number of animals, showed a negative correlation with high percent of plantation forestry and riparian mud, sand or siltstone and positive links with high percent of primary pasture, urban, ash, native and riparian pumice. As the biotic variables considered were influenced by a combination of geology and land cover, indices may be best comparable over areas of similar rock and soil bases.

Indices are a useful representation of community structure, reducing multiple variables to a simple figure. But are they an accurate representation of 'river health' assumed by numerous biomonitoring programs? In New Zealand, these indices are widely used in resource management to meet the requirements under the New Zealand Resource Management Act (1991) in assessing the 'life-supporting capacity' of streams and rivers. The indices are often applied by resource managers without an understanding of what they represent or how they should be applied. These indices are useful in identifying changes in a river system as they are comparable if taken at the same site over time, although resent hydrological history maybe influence community structure (Chapter 2). However, differences in index values over space need not be a result of degradation but may reflect other environmental differences such as geology. Indices have a place in the monitoring of our environment they need to be used with care and understanding of the biological information they represent.

How useful are statistical tools (variable reduction and multiple regression) and geographic information systems in understanding biotic systems?
Variable reduction was used in this study to condense environmental and biotic variables. Biotic information was reduced from a complete set of information on taxa densities to indices, ordination and a number of common taxa. This was important to

make the data manageable, although while some information on community structure is lost in this process, it allows the data to be analysed in such a way that the results are meaningful.

Reduction of landscape variables through principal component analysis was not particularly useful in this study, explaining only 47% of the original landscape data and yielding no correlations better than those with the individual landscape variables. Original variables are likely to be more meaningful than component scores, and are also easier to interpret and explain. This suggests it is more appropriate to use the original landscape variables than the reduced variables. Clausen & Biggs (1997) also found this in analysis of hydrological variables in New Zealand.

Nine of the multiple regressions explained greater than 40% of the variation in the data. These included Margalef's, MCI, QMCI, EPT, DEC1, DEC2, *Nesameletus* sp., *P. antipodarum* and *Zelandoperla* sp. This suggest that not only are land cover and geology useful in exploring benthic invertebrate data but that regressions are a useful tool in identifying linkages.

This study has shown that the application of GIS is a useful tool in understanding links between benthic biota and landscape. GIS allows powerful analysis of large data sets over wide spatial regions. Although analysis of landscape variables has been carried out for many years, improvements in computer power and development of GIS tools has made the analysis of large data sets possible. The utilisation of GIS has made it practical to consider different scales as well as display spatial information. It is a powerful tool that has many applications in the ecological environment (Wadsworth & Treweek 1999). By utilizing GIS and statistical models, processes can be developed in one study that can quickly be transferred to other regions, improving flexibility and integration of information. The GIS environment is becoming more user friendly, which should encourage its integration into many aspects of ecological work.

Conclusions

Overall, the results suggest that the comparison of indices in areas of differing rock and soil type should be conducted with care. Of the data used in this study, the biotic indices and taxa densities showed more links with catchment than riparian scale variables. Community structure could not be predicted by any particular grouping of variables (e.g. land cover or geology). This study has identified GIS tools as particular useful in dealing with multi scale issues.

Literature Cited

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

Allan, J. D., and L. B. Johnson. 1997. Catchment-scale analysis of aquatic ecosystems. Freshwater Biology 37: 107-111.

Arheimer, B., and R. Liden. 2000. Nitrogen and phosphorus concentrations from agricultural catchments - influence of spatial and temporal variables. Journal of Hydrology 227: 140-159.

Bargos, T., J. M. Mesanza, M. Basaguren, and E. Orive. 1990. Assessing river water quality by means of multifactorial methods using macroinvertebrates. A comparative study of main water courses of Biscay. Water Resources 24: 1-10.

Bellamy, P. E., N. J. Brown, B. Enoksson, L. G. Firbank, R. J. Fuller, S. A. Hinsley, and A. G. M. Schotman. 1998. The influences of habitat, landscape structure and climate on local distribution patterns of the nuthatch (Sitta europaea L.). Oecologia 115: 127-136.

Benke, A. C., I. Chaubey, G. M. Ward, and E. L. Dunn. 2000. Flood pulse dynamics of an unregulated river floodplain in the southeastern U.S. coastal plain. Ecology 81: 2730-2741.

Brown, N. J., R. D. Swetnam, J. R. Treweek, J. O. Mountford, R. W. G. Caldow, S. J. Manchester, T. R. Stamp, D. J. G. Gowing, D. R. Soloman, and A. C. Armstrong. 1998. Issues in GIS development: adapting to research and policy-needs for management of wet grasslands in an environmentally sensitive area. International Journal of Geographical Information Science 12: 465-478.

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

Church, M. 2002. Geomorphic thresholds in riverine landscapes. Freshwater Biology 47: 541-557.

Clausen, B., and B. J. F. Biggs. 1997. Relationships between benthic biota and hydrological indices in New Zealand streams. Freshwater Biology 38: 327-342.

Clifford, H. T., and W. Stephenson. 1975. An introduction to numerical classification. Academic Press, New York.

Close, M. E., and R. J. Davies-Colley. 1990. Baseflow water chemistry in New Zealand rivers 2. Influence of environmental factors. New Zealand Journal of Marine and Freshwater Research 24: 343-356.

Cooper, S. D., S. Diehl, K. Kratz, and O. Sarnelle. 1998. Implications of scale for patterns and processes in stream ecology. Australian Journal of Ecology 23: 27-40.

Corkum, L. D. 1989. Patterns of benthic invertebrate assemblages in rivers in northwestern North America. Freshwater Biology 21: 191-205.

Cresser, M. S., R. Smart, M. F. Billett, G. Soulsby, C. Neal, A. Wade, S. Langan, and A. C. Edwards. 2000. Modelling water chemistry for a major Scottish river from catchment attributes. Journal of Applied Ecology 37: 171-184.

D'Angelo, D. J., S. V. Gregory, L. R. Ashkenas, and J. L. Meyer. 1997. Physical and biological linkages within a stream geomorphic hierarchy - a modelling approach. Journal of the North American Benthological Society 16: 480-502.

Department of Conservation. 1988. Wanganui River minimum flows submission : for consideration by the Rangitikei-Wanganui Catchment Board. Department of Conservation, Wellington.

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

Downes, B. J., P. S. Lake, E. S. G. Schreiber, and A. Glaister. 1998. Habitat structure and regulation of local species diversity in a stony, upland stream. Ecological Monographs 68: 237-257.

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

Environmental Systems Research Institute. 1999. ArcView Version 3.2. Redland, California.

Fleishman, E., R. MacNally, J. P. Fay, and D. D. Murphy. 2001. Modelling and predicting species occurrence using broad-scale environmental variables: an example with butterflies of Great Basin. Conservation Biology 15: 1674-1685.

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

Griffith, M. B., P. R. Kaufmann, A. T. Herlihy, and B. H. Hill. 2001. Analysis of macroinvertebrate assemblages in relation to environmental gradients in Rocky Mountain streams. Ecological Applications 11: 489-505.

Gritzner, M. L., W. A. Marcus, R. Aspinall, and S. G. Custer. 2001. Assessing landslide potential using GIS, soil wetness modelling and topographic attributes, Payette River, Idaho. Geomorphology 37: 149-165.

Guisan, A., and N. E. Zimmermann. 2000. Predictive habitat distribution models in ecology. Ecological Modelling 135: 147-186.

Hancock, G., and G. Willgoose. 2001. The interaction between hydrology and geomorphology in a landscape simulator experiment. Hydrological Processes 15: 115-133.

Hannaford, M. J., and V. H. Resh. 1995. Variability in macroinvertebrate rapid-bioassessment surveys and habitat assessments in a Northern California stream. Journal of the North American Benthological Society 14: 430-439.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones. 1998. Stream biodiversity: the ghost of land use past. Proceedings of the National Academy of Sciences of the United States of America 95: 14843-14847.

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

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

Hatch, L. K., A. Mallawatantri, D. Wheeler, A. Gleason, D. Mulla, J. Perry, K. W. Easter, R. Smith, L. Gerlach, and P. Brezonik. 2001. Land management at the major watershed - agroecoregion intersection. Journal of Soil and Water Conservation 56: 44-51.

Horrox, J. V. 1998. Benthic communities of the Whanganui River catchment: the effects of land use and geology. M.Sc. Thesis. Massey University, Palmerston North.

Huryn, A. D., and J. B. Wallace. 1987. Local geomorphology as a determinant of macrofaunal production in a mountain stream. Ecology 68: 1932-1942.

Hynes, H. B. N. 1970. The ecology of running waters. Liverpool University Press, Liverpool.

Hynes, H. B. N. 1975. The stream and its valley. Verhandlungen der internationalen Vereinigung fur theoretische und angewandte Limnologie 19: 1-15.

Jaberg, C., and A. Guisan. 2001. Modelling the distribution of bats in relation to landscape structure in a temperate mountain environment. Journal of Applied Ecology 38: 1169-1181.

Johnson, L. B., and S. H. Gage. 1997. Landscape approaches to the analysis of aquatic ecosystems. Freshwater Biology 37: 113-132.

Jowett, I., J. Boubee, and B. J. F. Biggs. 2000. Fish, benthic invertebrate, periphyton ans instream habitat in the upper Whanganui River catchment above and below the Western Diversions. Consultantly report ELE90243. National Institue of Water and Atmospher Research Ltd, Hamilton.

Joy, M. K., and R. G. Death. 2000. Development and application of a predictive model of riverine fish community assemblages in the Taranaki region of the North Island, New Zealand. New Zealand Journal of Marine and Freshwater Research 34: 241-252.

Lammert, M., and J. D. Allan. 1999. Assessing biotic integrity of streams: effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. Environmental Management 23: 257-270.

Land Information New Zealand. 2000. NZMS 1:50000 Series LINZ, Wellington.

Lenat, D. R. 1988. Water quality assessment of streams using a qualitative collection method for benthic invertebrates. Journal of the North American Benthological Society 7: 22-233.

Lenat, D. R., and J. K. Crawford. 1994. Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. Hydrobiologia 294: 185-199.

Linklater, W. L., and M. J. Winterbourn. 1993. Life histories and production of two trichopteran shredders in New Zealand streams with different riparian vegetation. New Zealand Journal of Marine and Freshwater Research 27: 61-70.

Lods-Crozet, B., E. Castella, D. Cambin, C. Ilg, S. Knispel, and H. Mayor-Simeant. 2001. Macroinvertebrate community structure in relation to environmental variables in a Swiss glacial stream. Freshwater Biology 46: 1641-1661.

Maasdam, R., and D. G. Smith. 1994. New Zealand's National Water Quality Network: relationships between physico-chemical data and environmental factors. New Zealand Journal of Marine and Freshwater Research 28: 37-54.

Malmqvist, B. 2002. Aquatic invertebrates in riverine landscapes. Freshwater Biology 47: 679-694.

Manel, S., S. T. Buckton, and S. J. Ormerod. 2000. Testing large-scale hypotheses using surveys: the effects of land use on the habitats, invertebrates and birds of Himalayan rivers. Journal of Applied Ecology 37: 756-770.

Manel, S., H. C. Williams, and S. J. Ormerod. 2001. Evaluating presence-absence models in ecology: the need to account for prevalence. Journal of Applied Ecology 38: 921-931.

McDonnell, R. A. 2000. Hierarchical modelling of the environmental impacts of river impoundment based on a GIS. Hydrological Processes 14: 2123-2142.

Ministry of Works and Development. 1978. Land Resource Inventory, Wellington.

Mosley, P., and I. Jowett. 1999. River morphology and management in New Zealand. Progress in Physical Geography 23: 541-565.

Nerbonne, B. A., and B. Vondracek. 2001. Effects of local land use in physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, USA. Environmental Management 28: 87-99.

Newson, M. D., and C. L. Newson. 2000. Geomorphology, ecology and river channel habitat: meaoscale approaches to basin-scale challenges. Progress in Physical Geography 24: 195-217.

Ohmann, J. L., and T. A. Spies. 1998. Regional gradient analysis and spatial pattern of woody plant communities of Oregon forests. Ecological Monographs 62: 151-182.

Ometo, J. P. H. B., L. A. Martinelli, M. V. Ballester, A. Gessner, A. V. Krusche, R. L. Victoria, and M. Williams. 2000. Effects of land use on water chemistry and macroinvertebrates in two streams of the Piracicana river basin, south-east Brazil. Freshwater Biology 44: 327-337.

Osbourne, L. L., and M. J. Wiley. 1988. Empirical relationship between land use/cover and stream water quality in an agricultural watershed. Journal of Environmental Management 26: 9-27.

Palik, B. J. G., P. C. Kirkman, L. K. West, L. 2000. Using landscape hierarchies to guide restoration of disturbed ecosystems. Ecological Applications 10: 189-202.

Perry, C. D., G. Vellidis, R. Lowrance, and D. L. Thomas. 1999. Watershed-scale water quality Impacts of riparian forest management. Journal of Water Resources Planning and Management 125: 117-125.

Quinn, J. M., A. B. Cooper, R. J. Davies-Colley, J. C. Rutherford, and R. B. Williamson. 1997. Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand hill country streams. New Zealand Journal of Marine and Freshwater Research 31: 569-577.

Raven, P. J., H. N.T.H., F. H. Dawson, and M. Everars. 1998. Quality assessment using river habitat survey data. Aquatic Conservation: Marine and Freshwater Ecosystems 8: 477-499.

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

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

Richards, C., L. B. Johnson, and G. E. Host. 1996. Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53: 295-311.

Roth, N. E., J. D. Allan, and D. L. Erickson. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. Landscape Ecology 11: 141-156.

Rothrock, J. A., P. K. Barten, and G. L. Ingman. 1998. Land use and aquatic biointegrity in the Blackfoot River watershed, Montana. Journal of the American Water Resources Association 34: 565-581.

Scarsbrook, M. R., and J. Halliday. 1999. Transition from pasture to native forest land-use along stream continua: effects on stream ecosystems and implications for restoration. New Zealand Journal of Marine and Freshwater Research 33: 293-310.

Simpson, E. H. 1949. Measurement of Diversity. Nature 163: 688.

Skinner, K. M., W. P. Kemp, and J. P. Wilson. 2000. GIS-based indicators of Montana grasshopper communities. Transactions in GIS 4: 113-128.

Sponseller, R. A., E. F. Benfield, and H. M. Valett. 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology 46: 1409-1424.

Stark, J. D. (1985). A macroinvertebrate community index of water quality for stony streams. Wellington, National Water and Soil Conservation Authority.

Stark, J. D. 1993. Performance of the macroinvertebrate community index: effects of sampling method, sampling replication, water depth, current velocity and substratum on index values. New Zealand Journal of Marine and Freshwater Research 27: 463-476.

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

Terralink International Limited. 2000. New Zealand Land Cover Database, Wellington.

Thompson, R. M., and C. R. Townsend. 2000. Is resolution the solution?: the effect of taxonomic resolution on the calculated properties of three stream food webs. Freshwater Biology 44: 413-422.

Townsend, C. R., A. G. Hildrew, and J. Francis. 1983. Community structure in some southern English streams: the influence of physicochemical factors. Freshwater Biology 13: 521-544.

Wadsworth, R., and J. Treweek. 1999. GIS for ecology: an introduction. Addison Wesley Longman Limited, Harlow.

Winterbourn, M. J., J. S. Rounick, and B. Cowie. 1981. Are New Zealand stream ecosystems really different? New Zealand Journal of Marine and Freshwater Research 15: 321-328.

Wong, M. D. 2002. Spacing and ecology of the Australasian Harrier (Circus approximans) in the Manawatu-Rangitikei sand country. M.Sc. Thesis. Massey University, Palmerston North.

Yang, M., C. J. Merry, and R. M. Sykes. 1999. Integration of water quality modelling, remote sensing and GIS. Journal of the American Water Resources Association 35: 253-263.

Zeilhofer, P.; Schessl, M. 2000: Relationship between vegetation and environmental conditions in the northern Pantanal of Mato Grosso, Brazil. Journal of Biogeography 27: 159-168.

4

Predicting the occurrence of stream invertebrate taxa using classification trees and logistic regression

Abstract

Classification trees and logistic regression were used to determine the occurrence of 22 benthic macroinvertebrate taxa in rivers of the Whanganui Catchment, New Zealand. Geographic Information Systems were used to calculate 21 variables from hydrological, land cover and geological databases. Twelve of the 22 taxa explored showed classification trees and/or stepwise logistic regression to give stronger models than the majority rule (the most common response, presence or absence, is applied to the full data set). Seven taxa showed classification trees to be significantly ($p < 0.05$) better than all other model types, with no taxa showing logistic regression to be the best at correctly predicting occurrence. The presence of extremely common taxa is not predicted well by any of these techniques. Classification trees gave equal or better predictions than other models and tended to produce simpler models, suggesting they may be a better modelling technique.

Keywords: Benthic invertebrates, classification trees, hydrology, landscape, logistic regression, New Zealand, river, Whanganui.

Introduction

Ecological data is often complex, lacks normality, includes large numbers of response variables, is strongly inter-correlated and often non-linear (Fowler and Cohen 1990, De'ath and Fabricius 2000, De'ath 2002). There are many methods that can be used to explore and analyse ecological data, all of which have their advantages and disadvantages. The aims of the analysis must be clearly understood before choosing a suitable method. This paper explores classification trees and stepwise logistic regression as two alternative approaches for examining the presence of stream invertebrate taxa at 138 sampling occasions in Whanganui Catchment Rivers, New Zealand.

Classification trees build a series of rules by which data can be grouped according to one response i.e. presence. Rules are selected based on what yields the best mutually exclusive groups. The objective is to split the data based on rules to give homogenous groups but at the same time keep the tree relatively small, simple and meaningful. The benefits of using classification trees are that the explanatory variables can be categorical or numerical, data does not have to be normal, and linear relationships are not assumed. It has been suggested that classification trees are more appropriate, give stronger results and are simpler to understand than other modelling techniques (Franklin 1998, De'ath and Fabricius 2000, Vayssieres et al. 2000).

Logistic regression is based on a linear model of variable combinations, in this case, a binary response (absence/presence). It has been specifically designed for modelling binomial data (Hastie and Pregibon 1993) and is widely used in modelling species occurrence (e.g. Owen 1990, Hill 1991, Osborne 1992, Compton et al. 2002).

Many studies have shown strong linkages between benthic stream invertebrates and hydrological and landscape variables. Resh et al. (1988) identified five key components of the hydrological regime as important to determining benthic invertebrate communities. Richards et al (1996) showed geology accounted for a large proportion of the variation in stream invertebrate communities. Land cover has also been identified as an important component in explaining variation in stream invertebrate communities in New Zealand and elsewhere (Harding and Winterbourn 1995, Barton 1996, Townsend et al. 1997, Rothrock et al. 1998, Ometo et al. 2000). Based on these assertions, the occurrence of 22 invertebrate taxon were explored through the two modelling techniques using hydrological, geological and land cover information as possible predictors.

Methods

Biological measures

Benthic invertebrate data from six previous studies undertaken in the Whanganui Catchment, New Zealand were analysed using similar sampling and taxonomy methods (Department of Conservation 1988, Dunning 1998, Horrox 1998, Jowett et al. 2000, M.R. Scarsbrook National Institute of Water and Atmospheric Research unpublished data, J.D. Stark Cawthron Institute unpublished data). From these studies, data from a total of 97 sites, in third to seventh order rivers, was collected over 12 years, with 1 to 8 sampling occasions per site (a total of 138 sampling occasions) and each comprising at least 4 Surber samples. To standardise the data, information from only 4 Surber samples was randomly selected for each sampling occasion.

In this analysis only the densities of those individual genera present at over 30% of the sites are used (*Aoteapsyche* sp., *Aphrophila neozelandica*, *Archichauliodes diversus*, *Austroclima* sp., *Austrosimulium* sp., *Coloburiscus humeralis*, *Deleatidium* sp., Eriopterini sp., *Hydrobiosis* sp., *Maoridiamesa* sp., *Nesameletus* sp., *Neurochorema confusum*, *Olinga feredayi*, *Oxyethira albiceps*, *Potamopyrgus antipodarum*, *Psilochorema* sp., *Pycnocentria* sp., *Pycnocentrodes* sp., *Tanytarsus vespertinus*, *Zelandobius* sp., *Zelandoperla* sp. and *Zephlebia* sp.).

Hydrological measures

To determine hydrological characteristics, information from 18 long-term flow sites and 15 rainfall sites (record length from 6 to 40 years), as well as instantaneous gauging, regional hydrological maps and topographic maps were used (Tonkin and Taylor Consultant Engineers 1978, Land Information New Zealand 2000, National Institute of Water and Atmospheric Research unpublished data, horizons.mw unpublished data, Genesis Power unpublished data).

There are a multitude of hydrological variables that can be calculated from long term records many of which represent similar hydrological characteristics. A range of commonly used variables that represent differing aspects of the hydrological regime that were identified as being related to the invertebrate community structure in chapter 2 were selected. Mean annual flow (Q), specific discharge (q) and the number of events three times the medium flow (FRE3) (Clausen and Biggs 1997) were calculated from long-term flow records using TIDEDA (Thompson and Walter 1996). Drought (DF) and flood frequencies (FF) (100 year return period event / annual return period event) (Mosley 1992) were calculated using HILLTOP (Rodgers 2001) and based on frequency analysis (McKerchar et al. 1997) of 7-day low flows and instantaneous high flows using the Gumbel distribution (Gumbel 1958).

Information was exported to a GIS package, ArcView (Environmental Systems Research Institute 1999), where contours were drawn based on hydrological site values, knowledge of topography, rainfall and instantaneous gauging information. From the contour information, a grid was interpolated to cover the whole catchment. Interpolation was done using the inverse distance weighted method (Environmental Systems Research Institute 1999). The 12 nearest neighbours were used with no barriers and power of two weighting. This method was applied as it gave smoother terrain (Environmental Systems Research Institute 1999) with the points closest to a location more likely to have the same rainfall thus given stronger weighting.

Catchments were defined using a digital elevation grid created from 20-metre contours and spot height information in ArcView. From the elevation grid, flow direction and flow accumulation grids were generated allowing the delineation of catchments and calculation of stream networks. A catchment polygon was created for each site sampled, and this overlaid on the hydrological grids to calculate mean and total interpolated variables. The digital elevation grid was also used to calculate river slope and catchment area.

The final set of hydrological variables were used to quantify variation in flow before sampling. Four temporal parameters were derived: amount of rain in the last 7 days

(T7), number of days since a total of 20 and 60 mm of rain had fallen (DT20, DT60) and the number of days since 40 mm of rain had fallen in a single day (D40).

Six of the rainfall recorder sites in the catchment had data for the whole period of interest. These 6 sites were used to interpolate the given values for each invertebrate sampling point. The 4 rainfall variables were calculated for the 6 rainfall sites and for any given location at any given time, the data was interpolated from the closest sites to give an average. This procedure, although not exact, gave an estimate of the variables at each of the sampling sites.

This gave a total of 11 hydrological variables; Mean annual flow (Q), specific discharge (q), number of events 3 times the medium flow (FRE3), drought frequency (DF), flood frequency (FF), slope, area, amount of rain in the last 7 days (T7), number of days since a total of 20 and 60 mm of rain had fallen (DT20, DT60) and the number of days since 40 mm of rain had fallen in a single day (D40).

Land quantification

Geographic Information Systems (GIS) was used to quantify land cover, rock and soil types in the area upstream of each stream invertebrate sampling site. Digital land cover information was obtained from the land classification database (lcdb) (Terralink International Limited 2000). Rock and soil data was taken from land resource information (lri) (Ministry of Works and Development 1979). Land cover was classified as plantation forest (PF), indigenous forest (IF), scrub (Scr), tussock (Tuss), bare ground (BG), urban (Urb) and primary pasture (PP). Dominant cover was calculated for both the riparian (R) and total catchment. Land cover was also defined into broader groups of trees (indigenous and planted) and native vegetation (indigenous, scrub and tussock) (Nat) as a percentage of total cover. Rock was grouped into 4 categories; ash, alluvium (All), mud, sand and siltstone (MSS), and volcanic (Vol). Soils were classified as pumice (Pum), loams, recent (Rec), mineral (Min), other (OS). Both soil and rock were each re-categorised into one nominal variable relating to the dominant factor. This was done for both the riparian (R) and catchment scale.

The catchment polygons derived for the hydrological analysis were used to calculate catchment cover. To determine the percentage land cover and geology type catchment polygons were overlaid on the lcdb and lri. The riparian area was defined by buffering the rivers by 100m either side, buffering the sample point by 800m and clipping the intersect. The intersect polygon was then clipped by the catchment area to give the riparian area (Chapter 3). These polygons were then overlaid on the lcdb and lri to determine dominate coverings.

Statistics

When comparing models it is important to assess how well they will predict new data i.e. data validation. To allow assessment of this, 30 models were run with 90% training data and 10% assessment data, randomly grouped in each run for each taxa. This meant over 400 test values were run through the models. By separating data into training and assessment data an estimate of the validation of the model can be made. This is important in the application of models and is also useful for comparing model types. Because of the small data set, it was not feasible to remove a large proportion to use as independent data in the model validation, thus the cross validation technique was used (Guisan and Zimmermann 2000). Classification trees were calculated in SAS using the Enterprise Miner extension (SAS 2000). They were built using only variables that were significant ($p < 0.05$) and selected based on the smallest number of rules. Misclassification error was determined on test data runs with the final model being derived from the full data set.

Logistic regression was also carried out in SAS Enterprise Miner (SAS 2000). Stepwise regression was used to stop inter-correlation with a significance level of $p < 0.05$. Misclassification errors, based on the 30 cross validation runs, were used to assess the accuracy of each model with the final model being determined from the full data set.

Misclassification rates were also calculated for the majority rule based on the same 30 subsets of data. The majority rule is the idea that the more common response, presence or absence, is applied to the full data set (De'ath and Fabricius 2000). For example if 60% of the sites have *P. antipodarum* present than it is assumed to be present at all sites and will give a misclassification rate of 40%.

Analysis of Variance (ANOVA) was used to determine if there were any significant differences in misclassification rates between classification trees, logistic regression and majority rule. If classification trees or logistic regressions were significantly different from the majority rule, T-tests were used to determine if differences were significant between the classification tree and logistic regression models.

Five taxa, that showed significant differences between majority rule and the other models, were examined further. These taxa were selected to represent a range of orders. The taxa *O. feredayi*, *Zelandoperla* sp., *Zephlebia* sp., *T. vespertinus* and *P. antipodarum* were used. To examine the effect of different data sets one taxon, *P. antipodarum*, was modelled based on 4 different data sets; the total data set, hydrology, land cover, and geology. These models were assessed through the same process, 30 runs of partitioned test data.

Results

All models

Models were only considered useful if they explained more of the variation in the data than the majority rule. Of the 22 taxa analysed, 12 had models were significantly different from the majority rule (Table 1). These 12 models are explored further to examine the differences between stepwise logistic regression and the classification trees. Of the 12 taxa that had differences between models, 5 showed classification trees and stepwise logistic regression to be the same but better than the majority rule. Seven taxa identified classification trees to be better at explaining the data. The models that did not differ from the majority rule were mainly those with high presence counts i.e. very common (no rare taxa were modelled).

Table 1: Percentage of misclassification for 22 stream invertebrate taxa sampled at 138 occasions in the Whanganui Catchment based on 3 modelling options; majority rule, classification trees and logistic regression. ANOVA results testing for significant differences in misclassification rates between model types are also given (highlighting shows taxa with significantly different models).

	Majority Rule	Classification Tree	Logistic Regression	P valve for a one-way ANOVA
Aoteapsyche sp.	14 ± 3	10 ± 1	10 ± 2	0.18
Aphrophila neozelandica	20 ± 2	18 ± 2	20 ± 2	0.83
Archichauliodes diversus	26 ± 2	24 ± 2	26 ± 2	0.71
Austroclima sp.	49 ± 2	40 ± 2	49 ± 2	<0.01
Austrosimulium sp.	56 ± 3	28 ± 2	37 ± 2	<0.01
Coloburiscus humeralis	17 ± 2	11 ± 2	16 ± 2	0.12
Deleatidium sp.	9 ± 2	5 ± 1	6 ± 1	0.18
Eriopterini sp.	39 ± 2	23 ± 2	31 ± 3	<0.01
Hydrobiosis sp.	16 ± 2	16 ± 2	16 ± 2	0.95
Maoridiamesa sp.	28 ± 2	20 ± 2	17 ± 2	<0.01
Nesameletus sp.	37 ± 2	26 ± 2	32 ± 2	<0.01
Neurochorema confusum	43 ± 3	29 ± 2	36 ± 2	<0.01
Olinga feredayi	61 ± 2	46 ± 3	43 ± 3	<0.01
Oxyethira albiceps	41 ± 3	36 ± 3	39 ± 2	0.45
Potamopyrgus antipodarum	43 ± 3	18 ± 2	24 ± 2	<0.01
Psilochorema sp.	38 ± 2	42 ± 3	38 ± 2	0.50
Pycnocentria sp.	43 ± 3	51 ± 3	42 ± 3	0.06
Pycnocentrodes sp.	41 ± 3	36 ± 2	37 ± 2	0.26
Tanytarsus vespertinus	40 ± 3	14 ± 2	23 ± 2	<0.01
Zelandobius sp.	37 ± 4	26 ± 2	27 ± 3	0.01
Zelandoperla sp.	28 ± 2	22 ± 2	20 ± 2	0.02
Zephlebia sp.	45 ± 3	37 ± 2	34 ± 2	0.02

Exploration of five taxa

To explore the differences in the modelling techniques further, 5 taxa were selected for more detailed examination. The selected taxa represent each of the main orders and include; *O. feredayi*, *Zelandoperla* sp., *Zephlebia* sp., *T. vespertinus* and *P. antipodarum*. *P. antipodarum* was used to assess the use of different data sets.

Olinga feredayi

The classification tree for *O. feredayi* only makes use of hydrological variables (Fig. 1A). The tree shows *O. feredayi* is more likely to be present with a high number of flood events per year. Stepwise logistic regression uses the riparian geology variables (Table 2). Both models were significantly ($p<0.0001$) different from the majority rule. The regression gave the lowest (43%) misclassification rate but was not significantly different in misclassification rate from the classification tree model.

Tanytarsus vespertinus

Both the stepwise logistic regression and classification trees show a large improvement from majority rule and use numerous variables, with the classification tree having the lower misclassification rate (Table 1). The best classification tree only used hydrological variables and showed *T. vespertinus* presence to be more likely the longer the time since the last flood event and the higher the overall frequency of events per year (Fig. 1B). Therefore, rivers that have regular freshes but have not recently had one are most likely to have *T. vespertinus*. The best stepwise logistic regression incorporates D40, FRE3, native, soils and riparian land cover (Table 2). This suggests that *T. vespertinus* will be more likely to be present the longer the time since the last event, higher FRE3, greater the amount of catchment in native vegetation, loam soils, riparian indigenous forest, riparian pasture and riparian scrub.

Zelandoperla sp.

Both the classification tree and stepwise logistic regression models for *Zelandoperla* sp. used only number of flood events per year (Fig. 1C and Table 2). They showed

Zelandoperla sp. densities to increase at higher FRE3. The logistic regression model had a slightly lower misclassification rate than the classification tree model, but the difference was not significant.

Zephlebia sp.

The classification tree shows *Zephlebia* sp. occurrence more likely at high drought frequency or at low drought frequency in areas with large amounts of riparian pasture and riparian scrub (Fig. 1D). The stepwise logistic regression suggests *Zephlebia* sp. will be present in streams with planted forest and pasture (Table 2). The regression model had a slightly lower misclassification rate than the tree model, but the difference was not significant. This genera grouping is made up of species that respond differently thus maybe confuse the models.

Potamopyrgus antipodarum

Classification tree models for *P. antipodarum* based on the full data set suggested it was more likely to occur if the amount of riparian native land cover was low and in areas of high drought frequency (Fig. 2A). Stepwise logistic regression incorporated 9 different variables, dominated by drought frequency and riparian native vegetation both of which occurred in over 20 of the 30 models created for validating the data. The logistic regression models also suggest *P. antipodarum* will be present with higher drought frequency and lower amounts of native vegetation (Table 2). Although this is similar to the classification tree model in the use of variables, the tree model gave better overall predictions i.e. the tree model was significantly better than the logistic regression at predicting taxa occurrence.

Variable selection - Potamopyrgus antipodarum

Using a limited data set, with only hydrological variables, constructed models showed no significant difference with those built from the full data set. There was still a significant difference between the model types, with the classification tree giving better results. The classification tree model showed *P. antipodarum* was more likely to occur in areas with high drought frequency or if an area had low specific discharge and low slope (Fig. 2B). Logistic regressions only used three of the hydrological variables, with FRE3 in all 30 models and flow in 21 of the validation model runs. The base model suggests *P. antipodarum* presence will be more likely with decreased FRE3 and flow (Table 2).

Models built from the geology (rock and soils) data set also showed no significant difference between those constructed from the complete set. Both classification trees and logistic regressions only used riparian rock and catchment soils (Fig. 2C and Table 2) but regression validation gave 6 runs with no variables selected. *P. antipodarum*, based on the classification tree models, were more likely to be present in catchments dominated by loamy and mineral soils and other catchments that have a riparian area dominated by alluvium rock. The logistic regression suggested *P. antipodarum* occurrence will increase with riparian alluvium, catchment loams, mineral soils and the other soils group, and decrease with riparian ash and catchment pumice soils.

There was no significant difference between the classification tree and logistic regression models produced using vegetation data. The classification tree suggests *P. antipodarum* is more likely to be present when riparian native vegetation is low and the native vegetation in the whole catchment is also low, or if catchment native vegetation is high but riparian area is predominately pasture (Fig. 2D). The logistic regression only uses one variable, percentage of riparian native vegetation (Table 2). As riparian native vegetation decreases, *P. antipodarum* occurrence is more likely. These models (land cover) were significantly less predictive than those created with the other variable groups (geology, hydrology and full data set).

Table 2: Best Logistic Regression models for *Olinga feredayi*, *Tanytarsus vespertinus*, *Zelandoperla* sp., *Zephlebia* sp. and *Potamopyrgus antipodarum* based on the full data set (hydrological, geology and land cover) unless otherwise stated, collected in the Whanganui Catchment over 138 sampling occasions (1988 to 2000). See methods text for explanation of abbreviations.

Taxa	Logistic Regression
Olinga	y = 0.141 + 0.406 RLoam + 0.707 ROS + 0.637 RPum
Tanytarsus	y = -24.9 + 3.64 D40 + 0.31 FRE3 + 0.043 Nat + 3.06 Loam - 2.34 Min - 0.53 OS - 5.51 Pum + 3.85 RIF - 2.73 RPF + 4.76 RPP + 3.01 RScr
Zelandoperla	y = -4.84 + 0.188 FRE3
Zephlebia	y = -0.254 - 0.169 IF + 0.947 PF + 1.58 PP - 1.36 Scr
Potamopyrgus – all variables	y = -2.78 + 2.99 DF - 0.028Rnat
Potamopyrgus – Hydrology	y = 18.1 - 0.339 FRE3 - 0.402 Q
Potamopyrgus – Geology	y = 1.08 + 1.17 Rall - 0.854 Rash + 12.3 Loam + 6.12 Min + 2.63 OS - 1.54 Pum
Potamopyrgus - Land cover	y = 1.73 - 0.034 Rnat

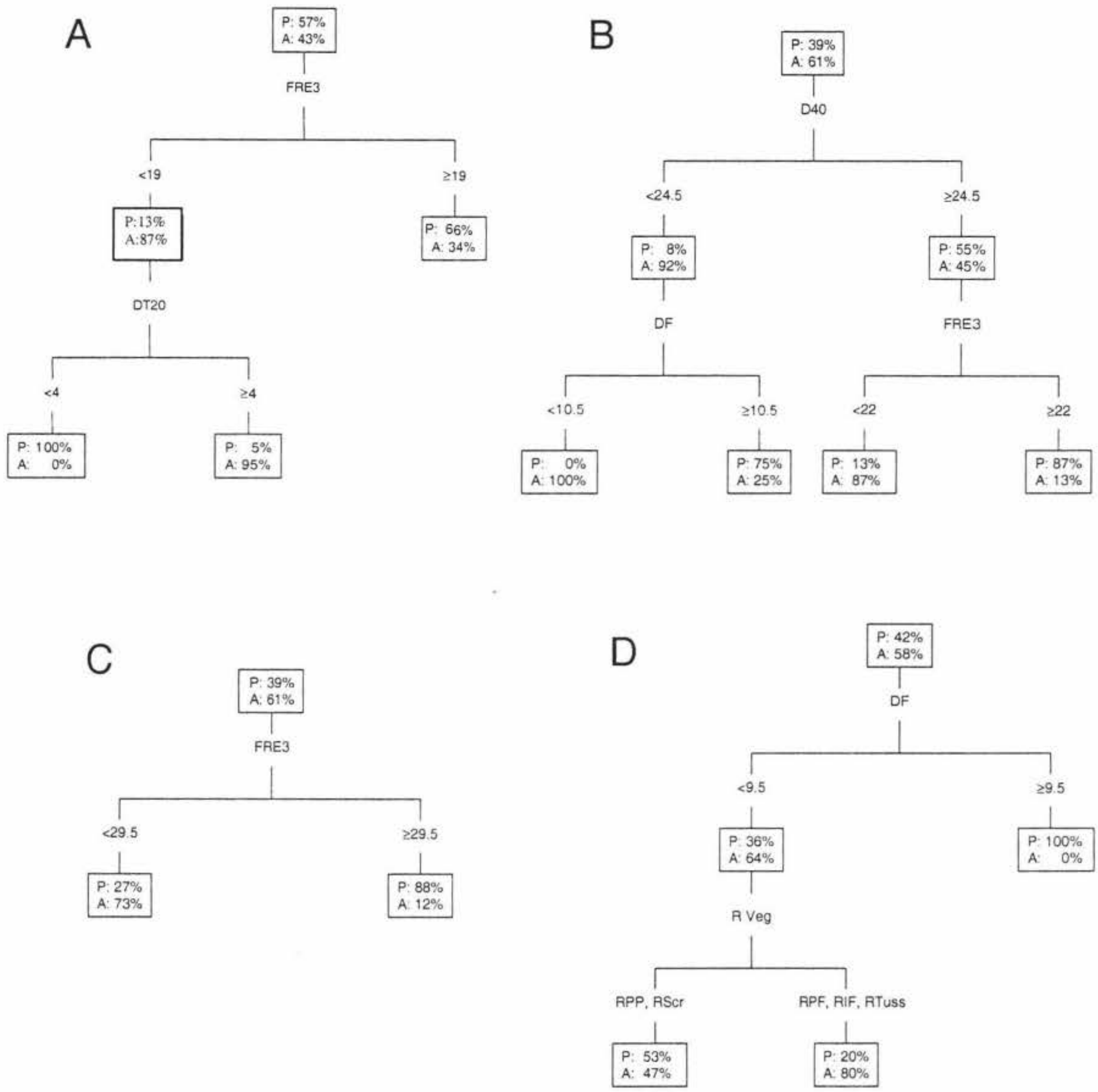


Figure 1: Classification trees based on hydrology, land cover and geology in the Whanganui Catchment: A - Tree for *Olinga feredayi*, B - Tree for *Tanytarsus vespertinus*, C- Tree for *Zelandoperla* sp., D - Tree for *Zephlebia* sp. (taxa collected over 138 sampling occasions 1988 to 2000). (P = Presents, A = Absents) Variables defined in Appendix 1.

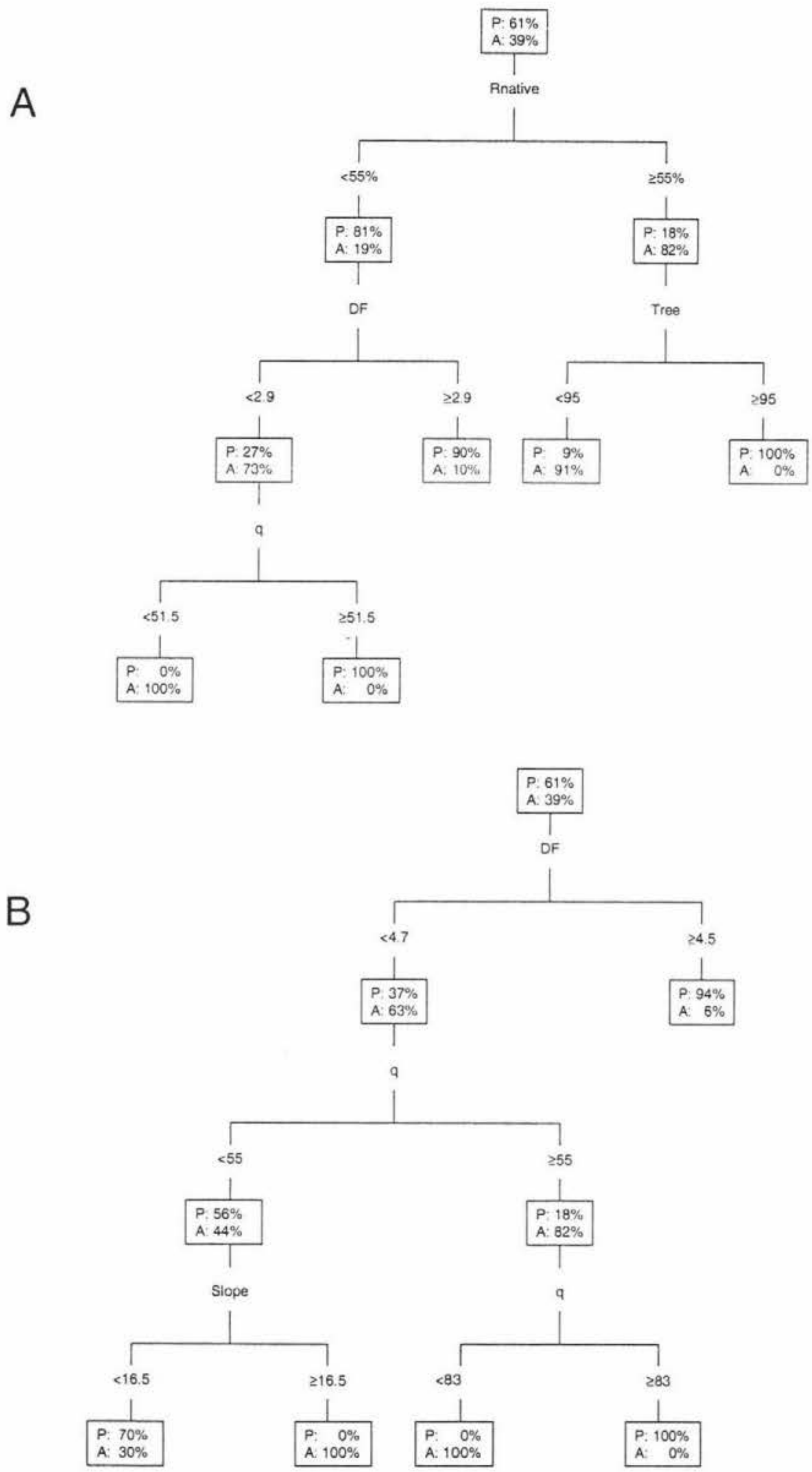


Figure 2: Classification tree for *Potamopyrgus antipodarum* collected in the Whanganui Catchment over 138 sampling occasions (1988 to 2000). A - based on hydrological, land cover and geological variables, B - based on hydrological variables. (P = Presents, A = Absents) Variables defined in Appendix 1.

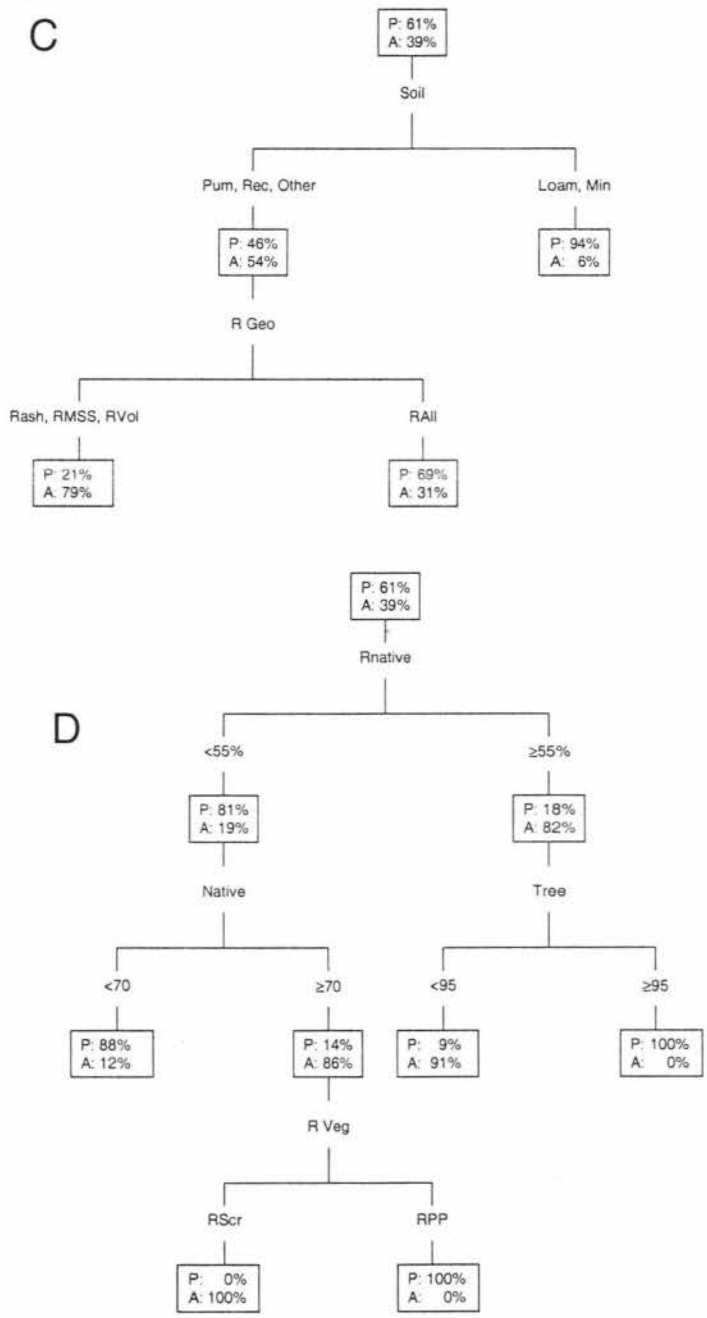


Figure 2 continued: Classification tree for *Potamopyrgus antipodarum* collected in the Whanganui Catchment over 138 sampling occasions (1988 to 2000). C – based on geological variables, D - based on land cover variables. (P = Presents, A = Absents) Variables defined in Appendix 1.

Discussion

As in a number of other studies, classification trees gave a lower misclassification rate than logistic regression (Franklin 1998, De'ath and Fabricius 2000, Vayssieres et al. 2000). Although in many cases (45% of the taxa examined) the results showed the use of either technique was no better than the majority rule. Low misclassification rate with the majority rule occurred when taxa were extremely common and the alternative modelling techniques tended not to further reduce the misclassification rate. This suggests that the occurrence of extremely common will not be well predicted by any of these techniques, at least for stream invertebrates in the Whanganui catchment based on the hydrological and landscape variables that have been considered. Although rare taxa were not explored they would be expected to have similar results to the very common taxa in terms of misclassification rate.

Of the 12 taxa that showed significant differences in the misclassification rates between the 3 modelling techniques, 7 revealed classification trees to be significantly better, and the remaining 5 showed no significant difference between classification trees and logistic regressions. This suggests that classification trees will give as good or better predictions, and are therefore probably a more accurate modelling technique. Of the models explored using the full data set, 4 out of 5 taxa showed classification trees require fewer variables than logistic regression. Simpler models, although not always better, are often easier to understand and therefore used more appropriately. The classification tree models were found to be more intuitive, easier to develop and interpret (Vayssieres et al. 2000).

When considering differences in variable selection between logistic regression and classification trees there was a wide variation in variables used. In some taxa very similar variables were selected using the two techniques, in others completely different variables were used. This occurs because variables may be highly correlated or due to differences in how the modelling techniques select variables. In logistic regression it is assumed there is a sigmoidal relationship between independent variables (Hastie and Pregibon 1993); this is not required with classification trees. Classification trees also build rules based on previous rules, thus taking into account the relationships between variables.

When considering the biological usefulness of the modelling process the logic and explanatory value of the models need to be considered along with the predictive power. The *T. vespertinus* classification tree model only used hydrological variables and showed rivers that have regular freshes, but have not recently had any, are more likely to have *T. vespertinus*. Suggesting *T. vespertinus* may be a slow coloniser thus is unlikely to be found directly after a flow event, alternatively it maybe food supply disturbed. In the logistic regression model *T. vespertinus* occurrence was more likely the longer since the last event, greater the number of events per year, higher the amount of catchment in native vegetation, loam soils, riparian indigenous forest, riparian pasture and riparian scrub; a far more complex model but with a similar misclassification rate. The classification tree model would therefore seem to be the more useful. *O. feredayi* models, in contrast, use a similar number of variables in each model but they are completely different, with classification trees suggesting occurrence to be more likely with a high number of events per year and regression using the riparian geological variables. *O. feredayi* has been suggested to prefer unstable streams (Death submitted). Collier and Scarsbrook (2000) suggest *O. feredayi* utilize the hyporheic zone, and this may reflect the influence of local geology. Both models give useful information and misclassification rates are not significantly different, therefore the use of both may be worthwhile. The difference in variable selection is probably due to the way the two techniques bring in and combine variables.

Zelandoperla sp. and *P. antipodarum* both used similar variables in the classification trees and logistic regressions. *Zelandoperla* sp. models showed occurrence to be more likely with a greater number of flood events per year. *Zelandoperla* sp. prefers water that is cool fast flowing and with stony substrates (Winterbourn and Gregson 1981), which tend to be second or third order streams in the upper catchments of the Whanganui. These are also sites that have high numbers of high flow events per year. *P. antipodarum* models suggest it was more likely to be present if the amount of riparian native land cover was low in areas of high drought frequency. This fits with other studies that have found much higher densities of *P. antipodarum* in pasture areas, lowland streams and streams which have low annual low flows (Holomuzki and Biggs 1999).

When comparing the models that limited the number of variables available for selection there was no difference in misclassification rate between the full set, hydrological set and geological set, with the vegetation group the only models that were significantly less predictive than the full data set. This suggests there is high inter correlation in the full data set, i.e. the variation in geology is explaining the same variation in the *P. antipodarum* data set as the hydrological variation.

Overall, this study has identified classification trees as a useful modelling technique giving as good or better predictions in all cases than the other modelling techniques (logistic regression and the majority rule). It has shown pre-section of variables an important component of the modelling process and that cross validation gives a useful measure of a models performance. The occurrence of extremely common taxa was not be well predicted by any of these techniques. In summary, classification trees are a valuable statistical tool for the exploration and prediction of species occurrence.

Literature Cited

Barton, D. R. 1996. The use of percent model affinity to assess the effects of agriculture on benthic invertebrate communities in headwater streams of Southern Ontario, Canada. Freshwater Biology 36: 397-410.

Clausen, B., and B. J. F. Biggs. 1997. Relationships between benthic biota and hydrological indices in New Zealand streams. Freshwater Biology 38: 327-342.

Collier, K. J., and M. R. Scarsbrook. 2000. Use of riparian and hyporheic habitats. Pages 178-208 in K. J. Collier and M. J. Winterbourn (editors). New Zealand stream invertebrates: ecology and implications for management. New Zealand Limnological Society, Christchurch.

Compton, B. W., J. M. Rhymer, and M. McCollough. 2002. Habitat selection by wood turtles (Clemmys insculpta): an application of paired logistic regression. Ecology 83: 833-843.

Death, R. G. Submitted. Spatial patterns in benthic invertebrate community composition: is disturbance actually important? Canadian Journal of Fisheries and Aquatic Sciences

De'ath, G. 2002. Multivariate regression trees: a new technique for modelling species-environment relationships. Ecology 83: 1105-1117.

De'ath, G., and K. E. Fabricius. 2000. Classification and regression trees: a powerful yet simple technique for ecological data analysis. Ecology 81: 3178-3192.

Department of Conservation. 1988. Wanganui River minimum flows submission : for consideration by the Rangitikei-Wanganui Catchment Board. Department of Conservation, Wellington.

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

Environmental Systems Research Institute. 1999. ArcView Version 3.2. Redland, California.

Fowler, J., and L. Cohen. 1990. Practical statistics for field biology. John Wiley and Sons Ltd, Chichester.

Franklin, J. 1998. Predicting the distribution of shrub species in southern California from climate and terrain-derived variables. Journal of Vegetation Science 9: 733-748.

Guisan, A., and N. E. Zimmermann. 2000. Predictive habitat distribution models in ecology. Ecological Modelling 135: 147-186.

Gumbel, E. J. 1958. Statistics of Extremes. Columbia University Press, New York.

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

Hastie, T. J., and D. Pregibon. 1993. Generalized linear models. Pages 195 - 247 in J. M. Chambers and T. J. Hastie (editors). Statistical Models in S. Chapman and Hall, London.

Hill, M. O. 1991. Patterns of species distribution in Britain elucidated by canonical correspondence analysis. Journal of Biogeography 18: 247-255.

Holomuzki, J. R., and B. J. F. Biggs. 1999. Distributional responses to flow disturbance by a stream- dwelling snail. Oikos 87: 36-47.

Horrox, J. V. 1998. Benthic communities of the Whanganui River catchment: the effects of land use and geology. M.Sc. Thesis. Massey University, Palmerston North.

Jowett, I., J. Boubee, and B. J. F. Biggs. 2000. Fish, benthic invertebrate, periphyton ans instream habitat in the upper Whanganui River catchment above and below the Western Diversions. Consultantly report ELE90243. National Institue of Water and Atmospher Research Ltd, Hamilton.

Land Information New Zealand. 2000. NZMS 1:50000 Series LINZ, Wellington.

McKerchar, A., R. Ibbitt, and R. Woods. 1997. Analysis and estimation of extreme events: deterministic methods. Pages 51-63 in M. P. Mosley and C. P. Pearson (editors). Floods and droughts: the New Zealand experience. New Zealand Hydrological Society Inc., Wellington

Ministry of Works and Development. 1978. Land Resource Inventory, Wellington.

Mosley, M. P. 1992. Waters of New Zealand. The Caxton Press, Wellington.

Ometo, J. P. H. B., L. A. Martinelli, M. V. Ballester, A. Gessner, A. V. Krusche, R. L. Victoria, and M. Williams. 2000. Effects of land use on water chemistry and macroinvertebrates in two streams of the Piracicana river basin, south-east Brazil. Freshwater Biology 44: 327-337.

Osborne, P. E. 1992. Interpreting bird atlas data using logistic models: an example from Lesotho, Southern Africa. Journal of Applied Ecology 29: 55-62.

Owen, J. 1990. An analysis of the spatial structure of mammalian distribution patterns in Texas. Ecology 71: 1823-1832.

Rodgers, M. W. 2001. Hilltop System. Napier, New Zealand, Hilltop Software.

Rothrock, J. A., P. K. Barten, and G. L. Ingman. 1998. Land use and aquatic biointegrity in the Blackfoot River watershed, Montana. Journal of the American Water Resources Association 34: 565-581.

SAS. 2000. SAS User's Guide: Statistics, Version 8. Cary, North Carolina, SAS Institute Inc.

Terralink International Limited. 2000. New Zealand Land Cover Database, Wellington.

Thompson, W. R., and K. M. Walter. 1996. Tideda for Windows. Christchurch, National Institute of Water and Atmospheric Research.

Tonkin and Taylor Consultant Engineers. 1978. Water resources of the Wanganui River. Tonkin and Taylor Consultant Engineers, Auckland.

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

Vayssieres, M. P., R. E. Plant, and B. H. Allen-Diaz. 2000. Classification trees: an alternative non-parametric approach for predicting species distributions. Journal of Vegetation Science 11: 679-694.

Winterbourn, M. J., and K. L. D. Gregson. 1981. Guide to the Aquatic Insects of New Zealand. Bulletin of the Entomological Society of New Zealand 5.

5

Effect of hydrology and landscape on densities of stream invertebrates in the Whanganui Catchment, New Zealand

Abstract

This study examines the effect of the hydrological regime and landscape characteristics on benthic stream macroinvertebrate communities in the Whanganui Catchment, New Zealand. Ten hydrological variables were calculated from hydrological records and topographic information to represent spatial and temporal variation in the hydrological regime. Catchment geology and land cover variables were derived with GIS from New Zealand's land classification database and land resource inventory. Community structure was characterised using ordination, species richness, total number of animals and the densities of 22 common taxa. Biotic data was also explored through diversity and water quality indices. Overall, FRE3 (number of flood events per year over 3 times the medium flow) had the greatest number and strongest links with the biotic variables. Hydrological characteristics dominate the explanatory variables in many of the index models suggesting that indices that measure water quality may only be comparable over areas with similar hydrological regimes and that diversity may be similarly strongly influenced by hydrology.

Keywords: benthic invertebrate, geographic information systems, geology, hydrology, landscape, land use, multiple regression, New Zealand, river, Whanganui.

Introduction

This study examines the relationship between benthic invertebrate community structure and hydrological and landscape characteristics of river in the Whanganui Catchment, New Zealand. It builds on the work presented in chapter 2, which examined the relation between the hydrological regime and benthic biota and chapter 3 where links with landscape and benthic biota were explored. A better understanding of the interactions between the hydrological, landscape and biotic components is important as it may allow more careful management of riverine environments (Richards et al. 1996, Allan et al. 1997, Meyer et al. 1999, Cresser et al. 2000, Smakhtin 2001, Wang 2001).

Five key components of the hydrological regime that can affect benthic invertebrate communities have been identified (Resh et al. 1988, Poff and Ward 1989, Poff et al. 1997, Chapter 2): magnitude of flow, frequency of high and low flow events, predictability of flow, duration and timing of events. In New Zealand predictability of flow is generally low and duration of events, particularly high flow events short by world standards (Duncan 1987). These hydrological characteristics affect different components of the invertebrate fauna to differing extents and at different times (Clausen and Biggs 1997, Chapter 2). Consequently, the hydrological regime cannot be summarised in a single variable when considering the link between hydrology and the benthic biota.

Catchment scale geology and land cover have also been found to influence the hydrological regime and water chemistry. Several studies (Richards et al. 1996, Johnson et al. 1997, Cresser et al. 2000, Chapter 3) have shown geology to account for a large proportion of the variation in invertebrate communities and strongly influence water chemistry. Land use practices have also been identified as an important component in explaining variation in invertebrate communities (Harding and Winterbourn 1995, Barton 1996, Townsend et al. 1997a, Rothrock et al. 1998, Ometo et al. 2000, Chapter 3).

Based on the assertion that hydrology, land cover and geology may be linked to riverine biotic communities, 22 invertebrate taxa, diversity indices and water quality measures are examined through multiple regressions using hydrological, geological and land cover data in the Whanganui Catchment, New Zealand.

Study Site

The Whanganui River, located in the North Island of New Zealand, has a catchment of 7100 km^2. The catchment varies in vegetation, from native bush and tussock through to exotic forest and intensive farming, with the lower altitude areas of the catchment most intensively farmed. The dominant geology types include tertiary sandstone, mudstone and siltstone with streams around Mt Ruapehu flowing through bedrock of volcanic origin. The climate varies from high rainfall areas in the upper reaches, to areas in the mid and lower catchment that experience periods of prolonged dry weather.

Methods

Biological measures

This study is based on invertebrate data sourced from a number of previous studies undertaken in the area. Samples were collected as part of 6 different studies but with sampling and taxonomy methods consistent between studies (Department of Conservation 1988, Dunning 1998, Horrox 1998, Jowett et al. 2000, M.R. Scarsbrook National Institute of Water and Atmospheric Research unpublished data, J.D. Stark Cawthron Institute unpublished data). From these studies data from a total of 97 sites, in third to seventh order rivers, was collected over 12 years, with 1 to 8 sampling occasions per site (a total of 138 sampling occasions) and each comprising at least 4 Surber samples. To standardise the data, information from only 4 Surber samples was randomly selected for each sampling occasion.

Invertebrate community structure

Variables used as measures of community structure were; total number of animals, species richness and the densities of those individual genera present at over 30% of the sites (*Aoteapsyche* sp., *Aphrophila neozelandica*, *Archichauliodes diversus*, *Austroclima* sp., *Austrosimulium* sp., *Coloburiscus humeralis*, *Deleatidium* sp., Eriopterini sp., *Hydrobiosis* sp., *Maoridiamesa* sp., *Nesameletus* sp. *Neurochorema confusum*, *Olinga feredayi*, *Oxyethira albiceps*, *Potamopyrgus antipodarum*, *Psilochorema* sp., Pycnocentria sp., *Pycnocentrodes* sp., *Tanytarsus vespertinus*, *Zelandobius* sp., *Zelandoperla* sp. and *Zephlebia* sp.). Taxa and total density were log transformed (log (x+1)) to improve normality of the data. Two DECORANA axes (DEC1 and DEC2) of the full communities (as represented by taxa densities) were also considered. They were calculated using Detrended Correspondence Analysis (DECORANA) in PC-Ord (McCune and Mefford 1999).

Biotic indices

An alternative way to characterise communities is to use biological indices. Indices calculated included water quality indices of Macroinvertebrate Community Index (MCI) (Stark 1985), Quantitative Macroinvertebrate Community Index (QMCI) (Stark 1993), %EPT – *O. albiceps* (%EPT-Oxy) (Lenat 1988). *O. albiceps* was removed from the EPT indices, as it is insensitive to changes in water quality. The diversity indices of Simpson's (Simp) (Simpson 1949) and Margalef's (Marg) (Clifford and Stephenson 1975) were also derived (Appendix 1).

Hydrological measures

To determine hydrological characteristics information from 18 long-term flow sites and 15 rainfall sites (record length from 6 to 40 years), as well as instantaneous gauging, regional hydrological maps and topographic maps were used (Tonkin and Taylor Consultant Engineers 1978, Land Information New Zealand 2000, National Institute of Water and Atmospheric Research unpublished data, horizons.mw unpublished data, Genesis Power unpublished data).

There are a multitude of hydrological variables that can be calculated from long term records many of which represent similar hydrological characteristics. A range of commonly used variables were selected to represent differing aspects of the hydrological regime that had been identified in the previous chapter as having strong links with benthic biota (Appendix 1). Mean annual flow ($\bar{Q}$), specific discharge (q) and number of events 3 times the medium flow (FRE3) (Clausen and Biggs 1997) were calculated from long-term records using TIDEDA (Thompson and Walter 1996). Drought (DF) and flood frequencies (FF) (100 year return period event / annual return period event) (Mosley 1992) were calculated using HILLTOP (Rodgers 2001) and based on frequency analysis (McKerchar et al. 1997) of 7-day low flows and instantaneous high flows using the Gumbel distribution (Gumbel 1958).

Information was exported to a GIS package, ArcView (Environmental Systems Research Institute 1999), where contours were drawn based on hydrological site values, knowledge of topography, rainfall and instantaneous gauging information. From the contour information, a grid was interpolated to cover the whole catchment. Interpolation was done using inverse distance weighted method (Environmental Systems Research Institute 1999). The 12 nearest neighbours were used with no barriers and power of two weighting. This method was applied as it gave smoother terrain (Environmental Systems Research Institute 1999) with the points closest to a location more likely to have the same rainfall thus given stronger weighting.

Catchments were defined using a digital elevation grid created from 20-metre contours and spot height information in ArcView. From the elevation grid, flow direction and flow accumulation grids were generated allowing the delineation of catchments and calculation of stream networks. A catchment polygon was created for each site sampled, and this overlaid on the hydrological grids to calculate mean and total interpolated variables. The digital elevation grid was also used to calculate river slope.

The final set of hydrological variables was used to quantify variation in flow before sampling. Three temporal parameters were derived: amount of rain in the last 7 days (T7), number of days since a total of 20 mm of rain had fallen (DT20) and the number of days since 40 mm of rain had fallen in a single day (D40).

Six of the rainfall recorder sites in the catchment had data for the whole period of interest. These six sites were used to interpolate the given values for each invertebrate sampling point. The 3 rainfall variables were calculated for the 6 rainfall sites and for any given location at any given time, the data was interpolated from the closest sites to give an average. This procedure, although not exact, gave an estimate of the variables at each of the sampling sites.

This gave a total of 9 hydrological variables; Mean annual flow ($\bar{Q}$), specific discharge (q), number of events 3 times the medium flow (FRE3), drought frequency (DF), flood frequency (FF), slope, amount of rain in the last 7 days (T7), number of days since a total of 20 mm of rain had fallen (DT20) and the number of days since 40 mm of rain had fallen in a single day (D40).

Land cover and geology quantification

Geographic Information Systems (GIS) was used to quantify land cover and geology in the area up stream of each invertebrate site. Digital land cover information was obtained from the land classification database (lcdb) (Terralink International Limited 2000). Geology data was taken from land resource information (lri) (Ministry of Works and Development 1979).

The catchment polygons calculated for the hydrological analysis were used to calculate catchment cover. To determine the percentage land cover and geology type the catchment polygons were overlaid on the lcdb and lri. This gave 11 landscape variables; plantation forest (PF), indigenous forest (IF), scrub (Scr), tussock (Tuss), pasture (PP), urban (Urb), bare ground (BG), ash (Ash), alluvium (All), mud sand and siltstone (MSS) and pumice (Pum).

Statistics

Once all hydrological variables, indices and DECORANA axes had been calculated, data was checked for univariate and multivariate normality and variables transformed if necessary. As a result all taxa densities, total density and a number of hydrological parameters were log transformed.

Data was explored to determine if relationships were linear, logistic, exponential or polynomial by plotting data points (Chapter 2 and 3). The majority of variables showed linear or near linear relationships apart from those of FRE3 and river size ($\bar{Q}$) with total number of animals and species richness. To examine the relationship between individual hydrological variables and landscape variables and individual biological variables Pearson's correlation coefficients were used for those that showed linear relationships and 2nd order polynomial regressions for the non-linear relationships.

Multiple regression is widely used in ecology for predictive modelling and data exploration (e.g. Hawkins et al. 1997, Jowett and Biggs 1997). Multiple regressions considered the effect of all hydrological and landscape variables on each biological variable. Multiple regression was done using a stepwise method with p=0.05, which prevented autocorrelation within each of the models.

Significance level was set at p = 0.0001, with Bonferroni corrections this gave an overall experiment wise error of 0.15. DECORANA ordination was carried out in PC-Ord (McCune and Mefford 1999), all other statistical analyses were done using the SAS system (SAS 2000).

Results

Univariate correlations

Species richness and Simpson's indices were not strongly correlated with any measured environmental variables. DECORANA axis 1 was correlated with 9 environmental variables (FRE3, q, DF, PF, Tuss, Ash, Urban, Pumice and Slope). Of the environmental variables, FRE3, D40, percent pasture and percent peat showed the most significant links with the biotic variables. Margalef's index, EPT, MCI and QMCI decreased with greater time since the last event, higher percentage cover of pasture and urban area, and increased with higher FRE3 and specific discharge. Overall, there were a similar number of correlations between biotic indices and DECORANA axes with hydrological variables (21) and landscape variables (18) (Table 1).

Twelve out of the 22 taxa examined show a significant link with 1 or more of the environmental variables considered. *P. antipodarum* showed the highest number of individual correlations with the environmental variables (positive with FRE3, q, Tuss, Ash and Pumice, negative with DF, Slope, pasture and MSS) Once again, FRE3 was the most correlated environmental variable, with specific discharge, slope and ash also having a number of significant correlations. Hydrological and landscape variables again showed a similar number of significant links with taxa densities (18 and 19 respectively) (Table 1).

Table 1: Correlations between indices and DECORANA axes of taxa and environmental variables from 138 sites in the Whanganui Basin, New Zealand (only significantly correlated variables are shown, full correlation tables in Appendix 4,7 and 10).

	Q	FRE3	q	DF	Slope	D40	IF	PF	PP	Tuss	Urb	Ash	MSS	Pum
Marg		+	+		-			-				+		+
EPT-Oxy	-	+				-			-		-			
MCI	-	+	+		-	-						+		+
QMCI	-	+				-			-		-			
Density								-	+		+			
DEC1		-	-	+	+			+		-	-	-		-
DEC2	+	-				+			+					
Aoteapsyche sp.	+							-	+		+			
Aphrophila neozelandica						-						+		
Deleatidium sp.		+	+					-						
Hydrobiosis sp.								-			+			
Maoridiamesa sp.		+	+		-							+		
Nesameletus sp.						-	+							
Olinga feredayi						-								
Potamopyrgus antipodarum		-	-	+	+				+	-		-	+	-
Psilochorema sp.		+												
Pycnocentrodes sp.												+		
Tanytarsus vespertinus											+			
Zelandoperla sp.		+	+	-	-				-	+				+

Multiple regression

Of the diversity indices, only the Margalef's index model explained a large proportion of the data variation (Fig. 1). This is similar to univariate correlations that revealed no strong links for Simpson's Indices with any environmental variables. FRE3 was the most dominant variable in the Margalef model, explaining 36% of the variation in the data, with a higher FRE3 associated with an increase in Margalef's Index. Five of the 8 variables used in this model were hydrological (making up 52% of the 68% explained by the model). Of the other non-hydrological variables, percentage indigenous and planted forest added the most to the model (0.07 r^2 each), indigenous forest had a positive link and planted forest a negative link with increased Margalef values.

All three water quality indices, EPT, MCI and QMCI, gave high r-squared values for the overall models (>0.55) (Fig. 1). EPT and QMCI models utilize similar variables, with D40 and mean annual flow the top three predictor variables in both cases. D40 and mean annual flow both had negative relationships with water quality indices, index values increasing with lower mean annual flow and few days since the last 40 mm event. FRE3 dominated the MCI model, explaining 42% of the data variation by itself.

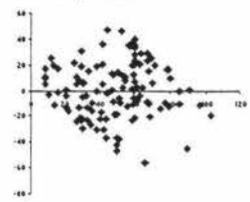
All three water quality indices models utilize mainly hydrological variables with indigenous forest, urban and bare ground the only non-hydrological variable incorporated in any of the models.

The species richness model had an r-squared of 0.54, with FRE3, DT20, D40 and specific discharge adding most to the model. Specific discharge showed a negative link with species richness, the other three variables had a positive link. The total number of animals model was weaker with an r-squared of 0.39 but only utilized four variables; ash, D40, MSS and urban, with all showing a positive link with total number of animals. Both DECORANA axes had strong models (r^2>0.5) and use only a small number of variables. The first DECORANA axis model, incorporated percent urban, drought frequency and percent planted forest. The second used D40, mean annual flow and percent pasture.

Of the 22 taxa models explored all were significant, with r-square values ranging from 0.05 to 0.65. All 20 variables were incorporated into at least one of the models, with FRE3 being the most common variable, and used in 12 of the models. Seven variables were incorporated in only 1 or 2 of the models. FRE3 was the first variable entered in five of the taxa models, explaining a high proportion of the data variability in three. *Deleatidium* sp. (0.3), and *Zelandoperla* sp. (0.4) had strong positive links; *P. antipodarum* (0.51) had a strong negative link with FRE3.

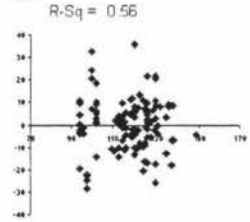
Only models that explained a large amount of overall variation (r^2>0.5) in the data are discussed further. These were the model for *A. diversus*, *Deleatidium* sp., *Maoridiamesa* sp. and *P. antipodarum*. *A. diversus* model had an r-squared of 0.56 and used seven variables, showing negative links with specific discharge and percent scrub, and positive links with drought frequency, FRE3, and percent ash, MSS, and bare ground cover. The *Deleatidium* sp. model had an r-squared of 0.54 and used six variables. FRE3 added the most to the model (r^2=0.3) with drought frequency being the next strongest (0.12) both with a positive links, DT20, D40 and flood frequency also had positive links, percent planted forest had a negative link. The *Maoridiamesa* sp. model had an r-squared of 0.5 and used six variables, with FRE3 again adding the most to the model (0.2). FRE3, D40, percent pasture, percent urban and T7 had positive links, alluvium had a negative link. The *P. antipodarum* model had the highest r-squared of all the taxa models, at 0.65, and only used three variables. The first entered into the model, FRE3, had a very high partial r-squared of 0.51 and showed a negative link with the densities. Percent urban area also showed a negative link and percent pasture had a positive link.

%EPT-Oxy
R-Sq = 0.58



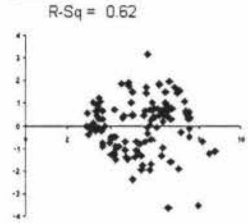
Step	Variable Added	Variable Removed	Model Pr > |t| Partial r-squared	< 0.0001 Final Parameter Est	Final Pr > |t|
	Intercept			144.62	<.0001
1	Q		0.26	-5.33	<.0001
2	DT20		0.17	0.94	0.0048
3	D40		0.06	-11.87	<.0001
4	IF		0.04	0.39	0.0002
5	Urb		0.03	-31.87	0.0286
6	BG		0.02	1.10	0.0063

MCI
R-Sq = 0.56



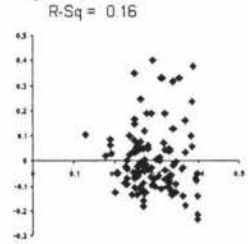
Step	Variable Added	Variable Removed	Model Pr > |t| Partial r-squared	< 0.0001 Final Parameter Est	Final Pr > |t|
	Intercept			149.46	<.0001
1	FRE3		0.42	1.28	<.0001
2	FF		0.06	14.91	0.0018
3	Q		0.04	-1.93	0.0001
4	IF		0.04	0.18	0.0021

QMCI
R-Sq = 0.62



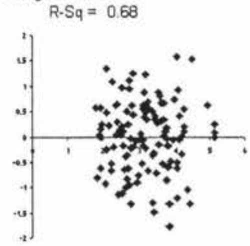
Step	Variable Added	Variable Removed	Model Pr > |t| Partial r-squared	< 0.0001 Final Parameter Est	Final Pr > |t|
	Intercept			14.35	<.0001
1	Q		0.29	-0.30	<.0001
2	D40		0.17	-0.56	<.0001
3	Urb		0.04	-2.11	0.0165
4	IF		0.03	0.01	0.0141
5	FRE3		0.04	0.06	0.0002
6	T7		0.03	-0.01	0.0034
7	FF		0.02	1.27	0.0124

Simpson's Index
R-Sq = 0.16



Step	Variable Added	Variable Removed	Model Pr > |t| Partial r-squared	< 0.0001 Final Parameter Est	Final Pr > |t|
	Intercept			-0.07	0.3648
1	Q		0.12	0.02	0.0004
2	D40		0.04	0.02	0.022

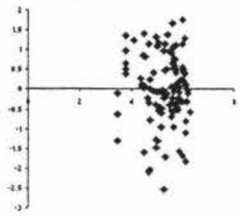
Margalef's Index
R-Sq = 0.68



Step	Variable Added	Variable Removed	Model Pr > |t| Partial r-squared	< 0.0001 Final Parameter Est	Final Pr > |t|
	Intercept			1.10	0.2312
1	FRE3		0.36	0.16	<.0001
2	IF		0.07	0.01	0.0016
3	DT20		0.07	0.04	0.0003
4	q		0.04	-0.05	<.0001
5	PF		0.07	-0.02	<.0001
6	D40		0.04	0.29	<.0001
7	FF		0.02	0.74	0.0084
8	All		0.02	-0.02	0.0242

Figure 1: Stepwise multiple regressions for taxa from the Whanganui Basin, NZ. Residual plot (residuals vs. predicted), r-squared, partial r-squares, parameter estimates and p-values of explanatory variables are presented.

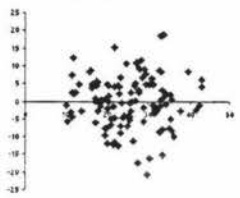
Total number of animals
R-Sq = 0.39



Model Pr > |t| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			1.52	0.0044
1	PP		0.15		
2	Ash		0.07	0.03	<.0001
3	D40		0.08	0.38	<.0001
4	MSS		0.08	0.02	<.0001
5		PP	0.02		
6	Urb		0.02	1.13	0.0424

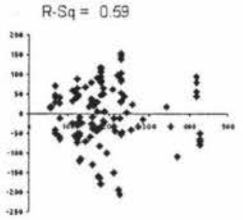
Species richness
R-Sq = 0.54



Model Pr > |t| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			-17.46	0.0043
1	FRE3		0.11	1.18	<.0001
2	DT20		0.11	0.64	<.0001
3	PF		0.05	-0.23	<.0001
4	D40		0.08	5.29	<.0001
5	q		0.07	-0.43	<.0001
6	All		0.04	-0.26	0.0023
7	T7		0.03	0.10	0.0092
8	IF		0.03	0.10	0.0146

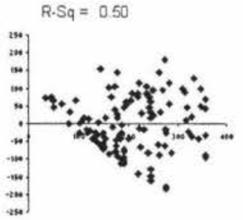
DECORANA axis 1
R-Sq = 0.59



Model Pr > |t| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			196.43	0.0042
1	Slope		0.37		
2	PF		0.09	1.47	<.0001
3	DF		0.08	59.91	0.0006
4		Slope	0.00		
5	Urb		0.03	-186.05	0.0005
6	FRE3		0.02	-3.21	0.0135

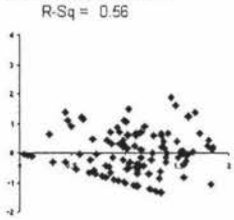
DECORANA axis 2
R-Sq = 0.50



Model Pr > |t| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			-187.76	0.0003
1	D40		0.29	42.88	<.0001
2	Q		0.17	13.97	<.0001
3	PP		0.05	1.22	0.0011

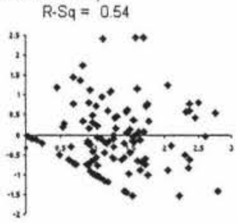
Archichauliodes diversus
R-Sq = 0.56



Model Pr > |t| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			-3.23	<.0001
1	D40		0.05		
2	DF		0.07	0.87	<.0001
3	FRE3		0.17	0.09	<.0001
4		D40	0.00		
5	Scu		0.05	-0.01	0.0038
6	Ash		0.07	0.02	<.0001
7	q		0.06	-0.04	<.0001
8	MSS		0.04	0.02	0.0002
9	BG		0.04	0.06	0.0012

Deleatidium sp.
R-Sq = 0.54

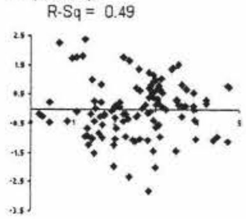


Model Pr > |t| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			-2.71	0.0805
1	FRE3		0.30	0.16	<.0001
2	DF		0.12	1.23	<.0001
3	DT20		0.04	0.04	0.0268
4	D40		0.04	0.24	0.028
5	PF		0.02	-0.01	0.0058
6	FF		0.02	0.87	0.037

Figure 1 continued: Stepwise multiple regressions for taxa from the Whanganui Basin, NZ. Residual plot (residuals vs. predicted), r-squared, partial r-squares, parameter estimates and p-values of explanatory variables are presented.

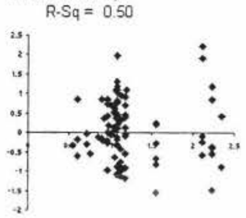
Eriopterini sp.
R-Sq = 0.49



Model Pr > \|t\| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			-2.94	<.0001
1	PP		0.09	0.02	<.0001
2	FRE3		0.11	0.09	<.0001
3	D40		0.17	0.31	<.0001
4	Slope		0.08	0.05	<.0001
5	BG		0.02	-0.02	0.0171
6	FF		0.02	0.40	0.023

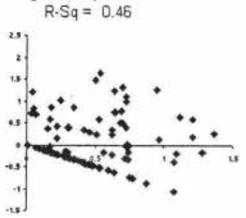
Maoridiamesa sp.
R-Sq = 0.50



Model Pr > \|t\| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			1.83	0.0008
	Intercept			-6.49	<.0001
1	FRE3		0.20	0.16	<.0001
2	Urb		0.10	1.67	0.031
3	D40		0.08	0.68	<.0001
4	All		0.05	-0.04	0.001
5	PP		0.05	0.02	0.0009
6	T7		0.02	0.01	0.038

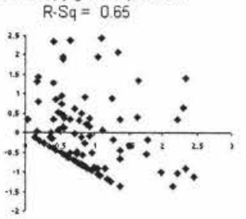
Olinga feredayi
R-Sq = 0.46



Model Pr > \|t\| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			-1.75	0.1449
1	D40		0.15		
2	IF		0.07	0.02	<.0001
3	Q		0.08	-0.14	0.0007
4	Ash		0.08	0.03	<.0001
5		D40	0.02		
6	Slope		0.06	0.11	<.0001
7	FRE3		0.03	0.04	0.0252

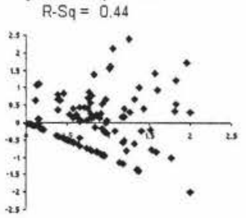
Potamopyrgus antipodarum
R-Sq = 0.65



Model Pr > \|t\| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			3.77	<.0001
1	FRE3		0.51	-0.09	<.0001
2	PP		0.05	0.03	<.0001
3	Urb		0.08	-2.93	<.0001

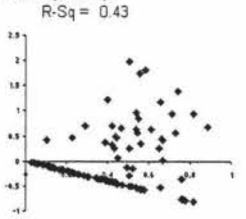
Tanytarsus vespertinus
R-Sq = 0.44



Model Pr > \|t\| < 0.0001

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			-9.14	<.0001
1	Urb		0.18	5.34	<.0001
2	DT20		0.05	0.10	0.0002
3	T7		0.03	0.03	0.0001
4	D40		0.06	0.75	<.0001
5	FRE3		0.05	0.11	<.0001
6	DF		0.06	1.02	0.0009

Zelandoperla sp.
R-Sq = 0.43



Model Pr > \|t\| 0.0169

Step	Variable Added	Variable Removed	Partial r-squared	Final Parameter Est	Final Pr > |t|
	Intercept			-1.83	<.0001
1	FRE3		0.40	0.09	<.0001
2	DT20		0.03	0.03	0.0141

Figure 1 continued: Stepwise multiple regressions for taxa from the Whanganui Basin, NZ. Residual plot (residuals vs. predicted), r-squared, partial r-squares, parameter estimates and p-values of explanatory variables are presented.

Discussion

The biotic aspects of river systems are influenced by numerous environmental variables (Resh et al. 1988, Richards et al. 1997). In this study both hydrological and landscape influences on invertebrate community have been explored to determine which is the most important factor in determining what the differing components of invertebrate communities might be.

The first stage of this exploration was to define both landscape and hydrological variables that represent different spatial and temporal components of the environment that may also influence the benthic biota. Numerous variables have been identified as having links with benthic biota in particular vegetation cover, geology and flow (Allan et al. 1997, Clausen and Biggs 1997, Richards et al. 1997, Rothrock et al. 1998, Harding et al. 1999, Chapters 2 and 3). Variables were selected based on the results of Chapters 2 and 3 both on their links with the biotic components and independence from each other. This yielded nine hydrological and 11 landscape variables.

Of all the individual environmental variables, FRE3 was found to be the most useful in explaining the biota. Chapter 2 discusses these relationships in more detail. This has also been identified in a number of other studies (Clausen and Biggs 1997, Clausen and Biggs 2000). Three other hydrological variables, mean annual flow, specific discharge and number of days since 40mm of rain had fallen, also show a number of correlations. These variables relate to river size, rainfall and time since the last rainfall event. River size has been suggested to be an important component of the hydrological regime in relation to stream communities in other studies (Ormerod and Edwards 1987, Death et al. 1999, Chapter 2). Specific discharge influences how extreme events are, in areas of low specific discharge low flow events are likely to be more extreme, high specific discharge often means there is a higher base flow. Time since the last flood event can determine what taxa are present, with the faster colonisers dominating the community soon after the last event and total numbers being severely reduced directly after an event (Hendricks et al. 1995, Chapter 2). Of the landscape variables, percentage planted forest, pasture, ash and peat had a number of strong links with biotic variables, suggesting some aspects of geology and land cover are important in determining invertebrate community structure (Harding et al. 1999, Chapter 3). FRE3 had the highest partial r-squared in five of the models; MCI (0.42), Margalef's Index (0.36), *Deleatidium* sp. (0.3), and *Zelandoperla* sp. (0.4) had positive links with FRE3, *P. antipodarum* (0.51) had a strong negative link (Chapter 2). This suggests these taxa are strongly influenced by the frequency of events, either requiring stable flows (*P. antipodarum*) (Holomuzki and Biggs 1999) or colonising unstable streams (*Deleatidium* sp. and *Zelandoperla* sp.) (Mackay 1992, Townsend et al. 1997b). Thus, although both landscape and hydrological components are important if only a limited number of variables are required FRE3 showed the strongest links.

Margalef's index, a measure of richness, and species richness were the only diversity indices strongly linked with landscape and hydrological variables. In particular, they related positively to frequency of flow events and time since the last event and negatively specific discharge. The models showed more diverse communities were richer the longer since the last event, in areas that experience regular events. This suggests that the greater the time since the last event the more time species will have had to recolonise the area. Clausen and Biggs (1997) also found links between richness and hydrology, the strongest were negative relationships between species richness and mean and medium flows; higher flows having lower richness. They also identified FRE3 to have a week positive relationship with richness, but did not consider specific discharge or time since the last event.

Water quality indices also showed good relationships with the hydrological variables. In particular, the indices increase with high FRE3, flood frequency and low mean annual flow and few days since the last event. This may be the result of two things; taxa that increase the index scores have a preference for more hydraulically disturbed sites and smaller streams, or streams in these areas have better water quality, with more regular flushing reducing contamination. This suggests that more work needs to be done in identifying the response of individual taxa and communities to water degradation with respect to similar hydrological regimes.

Four taxa had particularly strong predictive models; *A. diversus*, *Deleatidium* sp., *Maoridiamesa* sp. and *P. antipodarum*. *P. antipodarum* gave the strongest model (0.65) that indicated low FRE3 and high percent pasture sites yield the greatest densities. *P. antipodarum* is a freshwater snail that prefers slow flowing, stable environments

(Holomuzki and Biggs 1999), hence the negative correlation with FRE3. It also has a relatively high tolerance to organic enrichment, hence the positive correlation with percent pasture.

Overall, hydrological variables have been shown to be more closely linked with invertebrate community structure than the landscape variables. Hydrological variables are often not fully utilized in stream biological studies, this work has shown the importance of the hydrological regime, in particular the frequency of events in relation to stream invertebrates. The dominance of hydrological variables has major ramifications in the long-term management of our river systems. Under the current forecast for climate change not only will temperature rise but climatic patterns will change, in particular rainfall patterns (Mpelasoka et al. 2001). It has been suggested that extreme events will occur more often at more extreme levels. This study has shown frequency of events to be a major determinant of community structure, and thus climate change is likely to have a major detrimental impact on benthic stream macroinvertebrate communities. The importance of the hydrological variables also needs to be considered in the management of rivers, not just setting minimum flows but allowing for a 'natural' flow regime.

Literature Cited

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

Barton, D. R. 1996. The use of percent model affinity to assess the effects of agriculture on benthic invertebrate communities in headwater streams of southern Ontario, Canada. Freshwater Biology 36: 397-410.

Berger, W. H., and F. L. Parker. 1970. Diversity of plankton Formininfera in deep sea sediments. Science 168: 1345-1347.

Clausen, B., and B. J. F. Biggs. 1997. Relationships between benthic biota and hydrological indices in New Zealand streams. Freshwater Biology 38: 327-342.

Clausen, B., and B. J. F. Biggs. 2000. Flow variables for ecological studies in temperate streams: groupings based on covariance. Journal of Hydrology 237: 184-197.

Clifford, H. T., and W. Stephenson. 1975. An introduction to numerical classification. Academic Press, New York.

Cresser, M. S., R. Smart, M. F. Billett, G. Soulsby, C. Neal, A. Wade, S. Langan, and A. C. Edwards. 2000. Modelling water chemistry for a major Scottish river from catchment attributes. Journal of Applied Ecology 37: 171-184.

Death, R. G., H. McWilliam, and M. Rodway. 1999. Macroinvertebrate monitoring: lowland streams and order non-shingle river ecotypes. Pages 33-41 in Ministory for the Environment (editors). The use of macroinvertebrates in water management. Ministory for the Environment , Wellington.

Department of Conservation. 1988. Wanganui River minimum flows submission : for consideration by the Rangitikei-Wanganui Catchment Board. Department of Conservation, Wellington.

Duncan, M. J. 1987. River hydrology and sediment transport. Pages 113-137 in A. B. Viner (editors). Inland waters of New Zealand. DSIR, Wellington.

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

Environmental Systems Research Institute. 1999. ArcView Version 3.2. Redland, California.

Gumbel, E. J. 1958. Statistics of Extremes. Columbia University Press, New York.

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

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

Hendricks, A. C., L. D. Willis, and C. Snyder. 1995. Impact of flooding on the densities of selected aquatic insects. Hydrobiologia 299: 241-247.

Holomuzki, J. R., and B. J. F. Biggs. 1999. Distributional responses to flow disturbance by a stream- dwelling snail. Oikos 87: 36-47.

Horrox, J. V. 1998. Benthic communities of the Whanganui River catchment: the effects of land use and geology. M.Sc. Thesis. Massey University, Palmerston North.

Johnson, L. B., C. Richards, G. E. Host, and J. W. Arthur. 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology 37: 193-210.

Jowett, I., J. Boubee, and B. J. F. Biggs. 2000. Fish, benthic invertebrate, periphyton ans instream habitat in the upper Whanganui River catchment above and below the Western Diversions. Consultantly report ELE90243. National Institue of Water and Atmospher Research Ltd, Hamilton.

Land Information New Zealand. 2000. NZMS 1:50000 Series LINZ, Wellington.

Lenat, D. R. 1988. Water quality assessment of streams using a qualitative collection method for benthic invertebrates. Journal of the North American Benthological Society 7: 22-233.

Mackay, R. J. 1992. Colonization by lotic macroinvertebrates: a review of processes and patterns. Canadian Journal of Fisheries and Aquatic Sciences 49: 617-628.

McCune, B., and M. J. Mefford 1999. PC-ORD for Windows - Multivariate Analysis of Ecological Data. Gleneden Beach, Oregon, MjM Software.

McKerchar, A., R. Ibbitt, and R. Woods. 1997. Analysis and estimation of extreme events: deterministic methods. Pages 51-63 in M. P. Mosley and C. P. Pearson (editors). Floods and droughts: the New Zealand experience. New Zealand Hydrological Society Inc., Wellington

Meyer, J. L., M. J. Sale, P. J. Mulholland, and N. L. Poff. 1999. Impacts of climate change on aquatic ecosystem functioning and health. Journal of the American Water Resources Association 35: 1373-1386.

Ministry of Works and Development. 1978. Land Resource Inventory, Wellington.

Mosley, M. P. 1992. Waters of New Zealand. The Caxton Press, Wellington.

Mpelasoka, F.S., A.B. Mullan, and R.G.Heerdegen. 2001. New Zealand climate change information derived by multivariate statistical and artificial neural networks approaches. International Journal of Climatology 21: 1415-1433.

Ometo, J. P. H. B., L. A. Martinelli, M. V. Ballester, A. Gessner, A. V. Krusche, R. L. Victoria, and M. Williams. 2000. Effects of land use on water chemistry and macroinvertebrates in two streams of the Piracicana river basin, south-east Brazil. Freshwater Biology 44: 327-337.

Ormerod, S. J., and R. D. Edwards. 1987. The ordination and classification of macroinvertebrate assemblages in the catchment of the River Wye in relation to environmental factors. Freshwater Biology 17: 533-546.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47: 769-784.

Poff, N. L., and J. V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46: 1805-1818.

Resh, V. H., A. V. Brown, A. P. Covich, M. E. Gurtz, H. W. Li, W. Minshall, S. R. Reice, A. L. Sheldon, B. Wallace, and R. C. Wissmar. 1988. The role of disturbance in stream ecology. Journal of the North American Benthological Society 7: 433-455.

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

Richards, C., L. B. Johnson, and G. E. Host. 1996. Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53: 295-311.

Rodgers, M. W. 2001. Hilltop System. Napier, New Zealand, Hilltop Software.

Rothrock, J. A., P. K. Barten, and G. L. Ingman. 1998. Land use and aquatic biointegrity in the Blackfoot River watershed, Montana. Journal of the American Water Resources Association 34: 565-581.

SAS. 2000. SAS User's Guide: Statistics, Version 8. Cary, North Carolina, SAS Institute Inc.

Simpson, E. H. 1949. Measurement of diversity. Nature 163: 688.

Smakhtin, V. U. 2001. Low flow hydrology: a review. Journal of Hydrology 240: 147-186.

Stark, J. D. 1985. A Macroinvertebrate community index of water quality for stony streams. Wellington, National Water and Soil Conservation Authority.

Stark, J. D. 1993. Performance of the macroinvertebrate community index: effects of sampling method, sampling replication, water depth, current velocity and substratum on index values. New Zealand Journal of Marine and Freshwater Research 27: 463-476.

Thompson, W. R., and K. M. Walter. 1996. Tideda for Windows. Christchurch, National Institute of Water and Atmospheric Research.

Terralink International Limited. 2000. New Zealand Land Cover Database, Wellington.

Tonkin and Taylor Consultant Engineers. 1978. Water resources of the Wanganui River. Tonkin and Taylor Consultant Engineers, Auckland.

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

Townsend, C. R., S. Doledec, and M. R. Scarsbrook. 1997b. Species traits in relation to temporal and spatial heterogeneity in streams: A test of habitat templet theory. Freshwater Biology 37: 367-387.

Wang, X. 2001. Integrating water-quality management and land-use planning in a watershed context. Journal of Environmental Management 61: 25-36.

6

Synthesis

This study has found a number of strong relationships between environmental and biotic variables some of which have been identified in other studies (e.g. Resh et al. 1988, Poff and Ward 1989, Harding and Winterbourn 1995, Barton 1996, Richards et al. 1996, Clausen and Biggs 1997, Johnson et al. 1997, Poff et al. 1997, Richards et al. 1997, Townsend et al. 1997, Rothrock et al. 1998, Cresser et al. 2000, Ometo et al. 2000). The strongest individual links in the Whanganui catchment were between FRE3 and the biotic variables. Other hydrological variables that showed strong relationships with community structure were river size, specific discharge and time since the last event. Landscape characteristics, as measured by this study, also showed links with invertebrate community structure, though fewer than the hydrological variables. All indices and ordinations had significant multiple regressions when only landscape variables were incorporated. Canonical correlations of landscape variables showed catchment scale variables to be better predictors of community structure than reach scale variables. There was no clear pattern identified as to the dominance of land cover or geology variables in predicting community structure. The hydrological variables were in general better predictors of community than the landscape variables. Hydrological variables are often not fully utilized in an ecological sense, although this work has shown the importance of the hydrological regime, in particular the frequency of flood events.

The analytical techniques used in this study indicate Geographic Information Systems tools are useful in ecological studies, particularly with the integration of spatial data and scale analysis (Wadsworth and Treweek 1999). The interpolation process undertaken on the spatially variable hydrological components was particularly useful, allowing estimation for many otherwise incalculable parameters. The utilisation of GIS also made multi-scale analysis possible, incorporating catchment and riparian data into the models. Exploration of invertebrate taxa modelling techniques showed classification trees to be the most useful tool at an absence/presence level (De'ath and Fabricius 2000).

The dominance of hydrological variables in relation to invertebrate community structure has important ramifications in the long-term management of our river systems. This study has shown frequency of events to be a major controlling factor of community structure suggesting that it is important to not only set minimum flows but to ensure that a 'natural' flow regime is also considered in the management of river systems.

Literature Cited

Barton, D. R. 1996. The use of percent model affinity to assess the effects of agriculture on benthic invertebrate communities in headwater streams of Southern Ontario, Canada. Freshwater Biology 36: 397-410.

Clausen, B., and B. J. F. Biggs. 1997. Relationships between benthic biota and hydrological indices in New Zealand streams. Freshwater Biology 38: 327-342.

Cresser, M. S., R. Smart, M. F. Billett, G. Soulsby, C. Neal, A. Wade, S. Langan, and A. C. Edwards. 2000. Modelling water chemistry for a major Scottish river from catchment attributes. Journal of Applied Ecology 37: 171-184.

De'ath, G., and K. E. Fabricius. 2000. Classification and regression trees: a powerful yet simple technique for ecological data analysis. Ecology 81: 3178-3192.

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

Johnson, L. B., C. Richards, G. E. Host, and J. W. Arthur. 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology 37: 193-210.

Ometo, J. P. H. B., L. A. Martinelli, M. V. Ballester, A. Gessner, A. V. Krusche, R. L. Victoria, and M. Williams. 2000. Effects of land use on water chemistry and macroinvertebrates in two streams of the Piracicana river basin, south-east Brazil. Freshwater Biology 44: 327-337.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47: 769-784.

Poff, N. L., and J. V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow Patterns. Canadian Journal of Fisheries and Aquatic Sciences 46: 1805-1818.

Resh, V. H., A. V. Brown, A. P. Covich, M. E. Gurtz, H. W. Li, W. Minshall, S. R. Reice, A. L. Sheldon, B. Wallace, and R. C. Wissmar. 1988. The role of disturbance in stream ecology. Journal of the North American Benthological Society 7: 433-455.

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

Richards, C., L. B. Johnson, and G. E. Host. 1996. Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53: 295-311.

Rothrock, J. A., P. K. Barten, and G. L. Ingman. 1998. Land use and aquatic biointegrity in the Blackfoot River watershed, Montana. Journal of the American Water Resources Association 34: 565-581.

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

Wadsworth, R., and J. Treweek. 1999. GIS for ecology: an introduction. Addison Wesley Longman Limited, Harlow.

Appendix 1: Variable abbreviations

Abbreviation	Description of Variables
%EPT	% Ephemeroptera, Plecoptera and Tricoptera in the total sample
%EPT-Oxy	% Ephemeroptera, Plecoptera and Tricoptera not including *Oxyethira albiceps* in the total sample
%IEPT	% Ephemeroptera, Plecoptera and Tricoptera in the Insecta group
%IEPT-Oxy	% Ephemeroptera, Plecoptera and Tricoptera not including *Oxyethira albiceps* of the Insecta group
Area	Catchment area
Area/Peri	Catchment area divided by the Catchment Perimeter
BP	Berger-Parker diversity index for the total sample
CS	Catchment shape
D20	Number of days since 20 mm of rain has fallen in a single day
D40	Number of days since 40 mm of rain has fallen in a single day
Dd	River density / catchment area
DEC1	DECORANA axis 1
DEC2	DECORANA axis 2
density	Total number of animals
DF	Drought Frequency
DLO	Density of large order streams
DMO	Density of medium order streams
DSO	Density of small order streams
DT20	Number of days over which the last 20 mm of rain has fallen
DT40	Number of days over which the last 40 mm of rain has fallen
DT60	Number of days over which the last 20 mm of rain has fallen
Evapo	Evapotranspiration
FF	Flood Frequency
FRE12	Number of time the flow goes over 12 times the median flow per year
FRE3	Number of time the flow goes over 3 times the median flow per year
FRE6	Number of time the flow goes over 6 times the median flow per year
H10	10 year flood event
H100	100 year flood event
H1	mean annual flood
H20	20 year flood event
H5	5 year flood event
H50	50 year flood event
IBP	Berger-Parker diversity index for Insecta group
Imarg	Margalef's index for Insecta group
IPC1	Interpolated principle component 1
IPC2	Interpolated principle component 2
IPC3	Interpolated principle component 3
Isimp	Simpson's index for Insecta group
L10	10 year 7-day low flow

Abbreviation	Description of Variables
L100	100 year 7-day low flow
L1	mean annual 7-day low flow
L20	20 year 7-day low flow
L5	5 year 7-day low flow
L50	50 year 7-day low flow
LQ	Lower quartile of flow
MAR	Mean annual rainfall
Marg	Margalef's index for total sample
MCI	Macro invertebrate Community Index
Mm/day	mm of rain during day of sampling
MPC1	Map principle component 1
MPC2	Map principle component 2
MPC3	Map principle component 3
Nosp	Species Richness
Q	Mean Annul Flow
Order	Stream order
Peri	Catchment Perimeter
q	Mean annual flow / catchment area - specific discharge
Q10	flow 10% exceedance
Q20	flow 20% exceedance
Q50	Median Flow
Q90	flow 90% exceedance
QMCI	Quantitative Macro invertebrate Community Index
Rain24	10 year 24 hour rainfall intensity
RD	Road Density
RMAR	Local Mean annual rainfall
Rslope	Local slope
SD	Standard deviation of flow
Simp	Simpson's Index for total sample
Slope	Catchment slope
SQMCI	Semi Quantitative Macro invertebrate Community Index
StdH10	10 year event / catchment area
StdL10	10 year 7 day low flow / catchment area
T1	amount of rain fallen in the last day
T14	amount of rain fallen in the last 14 days
T21	amount of rain fallen in the last 21 days
T28	amount of rain fallen in the last 28 days
T7	amount of rain fallen in the last 7 days
TPC1	Temporal principle component 1
TPC2	Temporal principle component 2
UQ	Upper quartile of flow

Appendix 2: Pearson's correlations between hydrological variables calculated from long-term flow recorder information from 138 sampling occasions in the Whanganui Catchment, New Zealand (+++ r>0.9, + r>0.7, --- r<-0.9, - r<-0.7).

	$\bar{Q}$	SD	CV	Q90	Q75	Q50	Q25	Q10	FRE3	FRE6	FRE12	DF	L100	L50	L20	L10	L5	L1	FF	H100	H50	H20	H10	H5	H1
$\bar{Q}$		+++		+++	+++	+++	+++	+++					+	+++	+	+	+	+		+	+	+	+	+++	+++
SD				+	+++	+++	+++	+++					+	+		+				+++	+++	+++	+++	+++	+++
CV													-	-											
Q90					+++	+++	+++	+++				+	+++	+++	+	+++	+	+		+	+	+	+	+	+++
Q75						+++	+++	+++					+++	+++		+				+++	+++	+++	+++	+++	+++
Q50							+++	+++					+++	+++	+	+++	+	+		+++	+++	+++	+++	+++	+++
Q25								+++					+++	+++	+	+++	+	+		+	+	+++	+++	+++	+++
Q10													+	+		+				+++	+++	+++	+++	+++	+++
FRE3										+	+														
FRE6											+						-	-							
FRE12																	-	-							
DF													+	+	+	+	+								
L100														+++	+++	+++	+++	+++		+	+	+	+	+	+
L50															+++	+++	+++	+++		+	+	+	+	+	+
L20																+++	+	+							
L10																	+++	+++		+	+	+	+	+	+
L5																		+++							
L1																									
FF																									
H100																					+++	+++	+++	+++	+++
H50																						+++	+++	+++	+++
H20																							+++	+++	+++
H10																								+++	+++
H5																									+++
H1																									

Appendix 3: Pearson's correlations between interpolated, map and temporal principal components from 138 sampling occasions in the Whanganui Basin, New Zealand.

	IPC2	IPC3	MPC1	MPC2	MPC3	TPC1	TPC2
IPC1	0	0	-0.12	0.73	-0.23	-0.26	-0.08
IPC2		0	0.98	0.14	0.03	0.18	0.20
IPC3			-0.13	0.25	0.11	-0.22	0.01
MPC1				0	0	0.24	0.19
MPC2					0	-0.22	-0.02
MPC3						-0.01	0.00
TPC1							0

Appendix 5: Pearson's correlations between hydrological principal components and community structure from 138 sampling occasions in the Whanganui Catchment, New Zealand.

	MPC1	MPC2	MPC3	TPC1	TPC2	IPC1	IPC2	IPC3
DEC1	0.32	0.18	0.02	0.21	0.01	0.03	0.31	0.11
DEC2	-0.05	0.29	0.01	0.10	-0.05	0.24	-0.03	0.10
NoSp	-0.17	-0.59	0.06	0.26	-0.14	-0.63	-0.23	0.02
density	0.53	-0.11	0.10	0.45	0.10	-0.28	0.51	-0.11
Aoteapsyche sp.	0.41	0.07	0.04	0.12	-0.03	0.04	0.42	-0.05
Aphrophila neozelandica	0.03	0.33	-0.18	0.02	-0.14	0.33	0.06	0.00
Archichauliodes diversus	-0.24	-0.01	0.09	-0.24	-0.27	-0.07	-0.25	0.14
Austroclima sp.	-0.08	-0.14	0.19	-0.02	-0.09	-0.19	-0.09	0.18
Austrosimulium sp.	0.17	0.18	0.06	0.00	0.06	-0.02	0.18	0.10
Coloburiscus humeralis	-0.10	0.14	0.06	-0.04	-0.14	0.06	-0.10	0.12
Deleatidium sp.	-0.29	0.23	-0.07	-0.16	-0.18	0.41	-0.25	0.21
Eriopterini sp.	-0.08	0.08	0.08	0.11	-0.03	0.06	-0.09	0.07
Hydrobiosis sp.	0.14	0.26	0.08	-0.08	-0.01	0.26	0.13	-0.16
Maoridiamesa sp.	-0.10	0.36	-0.06	-0.09	0.02	0.43	-0.07	-0.11
Nesameletus sp.	-0.22	-0.01	-0.08	-0.41	-0.16	0.13	-0.21	0.19
Neurochorema confusum	0.09	0.21	0.01	0.02	0.17	0.30	0.10	-0.13
Olinga feredayi	-0.29	-0.01	-0.07	-0.36	-0.13	0.08	-0.26	0.30
Oxyethira albiceps	0.29	0.16	0.00	-0.03	0.39	0.06	0.28	-0.15
Potamopyrgus antipodarum	0.35	-0.45	0.14	0.25	0.15	-0.67	0.27	0.02
Psilochorema sp.	-0.29	0.13	-0.10	-0.03	-0.05	0.35	-0.26	0.01
Pycnocentria sp.	-0.20	0.14	-0.01	-0.06	-0.23	0.04	-0.18	0.31
Pycnocentrodes sp.	-0.22	0.23	-0.23	-0.21	-0.14	0.21	-0.20	0.19
Tanytarsus vespertinus	0.21	0.11	-0.01	0.24	0.11	0.10	0.18	-0.22
Zelandobius sp.	-0.07	0.14	0.02	-0.04	0.08	0.13	-0.03	0.16
Zelandoperla sp.	-0.36	0.41	-0.14	-0.18	-0.14	0.54	-0.29	0.07
Zephlebia sp.	-0.15	-0.29	-0.12	0.15	-0.04	-0.31	-0.21	0.18

Appendix 6: Pearson's correlations between biotic indices from 138 sampling occasions in the Whanganui Catchment, New Zealand.

	Isimp	Simp	IBP	BP	Imarg	Marg	EPT	EPT-Oxy	IEPT	IEPT-Oxy	MCI	QMCI	SQMCI
Isimp		0.81	0.97	0.75	-0.46	-0.38	-0.53	-0.51	-0.51	-0.49	-0.39	-0.46	-0.39
Simp	0.81		0.74	0.97	-0.28	-0.31	-0.47	-0.44	-0.45	-0.41	-0.21	-0.39	-0.30
IBP	0.97	0.74		0.74	-0.39	-0.31	-0.49	-0.45	-0.48	-0.44	-0.34	-0.39	-0.32
BP	0.75	0.97	0.74		-0.24	-0.28	-0.41	-0.36	-0.40	-0.34	-0.15	-0.32	-0.23
Imarg	-0.46	-0.28	-0.39	-0.24		0.96	0.38	0.39	0.21	0.24	0.48	0.34	0.37
Marg	-0.38	-0.31	-0.31	-0.28	0.96		0.28	0.30	0.14	0.17	0.35	0.27	0.30
EPT	-0.53	-0.47	-0.49	-0.41	0.38	0.28		0.95	0.92	0.88	0.73	0.84	0.81
EPT-Oxy	-0.51	-0.44	-0.45	-0.36	0.39	0.30	0.95		0.88	0.94	0.76	0.90	0.88
IEPT	-0.51	-0.45	-0.48	-0.40	0.21	0.14	0.92	0.88		0.94	0.68	0.82	0.79
IEPT-Oxy	-0.49	-0.41	-0.44	-0.34	0.24	0.17	0.88	0.94	0.94		0.73	0.90	0.87
MCI	-0.39	-0.21	-0.34	-0.15	0.48	0.35	0.73	0.76	0.68	0.73		0.78	0.83
QMCI	-0.46	-0.39	-0.39	-0.32	0.34	0.27	0.84	0.90	0.82	0.90	0.78		0.97
SQMCI	-0.39	-0.30	-0.32	-0.23	0.37	0.30	0.81	0.88	0.79	0.87	0.83	0.97	

Appendix 4: Pearson's correlations between hydrological variables and community structure from 138 sampling occasions in the Whanganui Catchment, New Zealand.

	Q	FRE3	L10	H10	Evapo	Rain24	MAR	RMAR	q	StdL10	StdH10	DF	FF	Area	peri	RD	Slope	Order	Rslope	CS	DSO	DMO	DLO	Dd	T1	T7	T14	T21	T28	DT20	DT40	DT60	D20	D40
DEC1	0.33	-0.04	0.31	0.32	-0.07	-0.01	0.02	-0.05	0.12	-0.04	0.05	0.04	0.17	0.30	0.30	0.31	-0.16	0.31	-0.29	0.30	0.00	-0.08	0.31	0.07	-0.03	-0.17	-0.13	-0.26	-0.22	0.26	0.09	0.15	0.10	0.19
DEC2	0.01	0.34	0.05	0.01	-0.27	0.12	0.24	0.21	0.22	0.18	0.21	-0.17	0.00	-0.05	-0.04	-0.05	-0.30	-0.03	-0.16	-0.07	0.20	-0.03	0.00	0.23	-0.03	-0.06	-0.01	-0.17	-0.10	0.29	0.02	0.00	0.04	0.12
NoSp	-0.30	-0.53	-0.41	-0.32	0.65	-0.45	-0.60	-0.38	-0.59	-0.58	-0.62	0.57	0.19	-0.17	-0.18	-0.19	0.61	-0.23	0.29	-0.15	-0.32	-0.07	-0.21	-0.50	-0.19	-0.28	-0.07	-0.17	-0.22	0.04	0.32	0.26	0.34	0.25
density	0.47	-0.52	0.41	0.47	0.27	-0.19	-0.29	-0.41	-0.24	-0.26	-0.22	0.17	0.10	0.53	0.53	0.52	0.18	0.49	-0.09	0.52	-0.18	-0.12	0.47	-0.17	-0.02	-0.30	-0.32	-0.41	-0.49	0.27	0.31	0.36	0.38	0.53
Aoteapsyche sp.	0.43	-0.14	0.40	0.44	-0.07	0.08	0.02	-0.09	0.11	-0.02	0.12	0.05	0.04	0.41	0.40	0.41	-0.01	0.40	-0.21	0.41	-0.07	-0.07	0.35	0.00	-0.03	-0.14	-0.13	-0.14	-0.16	0.16	0.07	0.11	0.05	-0.03
Aphrophila neozelandica	0.10	0.30	0.16	0.11	-0.36	0.26	0.32	0.27	0.30	0.26	0.31	-0.28	-0.11	0.03	0.04	0.04	-0.39	0.06	-0.15	0.02	0.12	0.20	0.04	0.30	0.03	-0.10	-0.15	-0.03	0.11	0.22	0.07	-0.05	-0.05	-0.23
Archichauliodes diversus	-0.22	0.22	-0.26	-0.23	0.03	-0.15	-0.06	0.06	-0.03	-0.14	-0.05	0.23	0.12	-0.24	-0.25	-0.25	0.02	-0.26	-0.07	-0.24	0.10	-0.14	-0.19	-0.04	-0.19	0.10	0.39	0.20	0.22	-0.02	-0.22	-0.29	-0.12	-0.23
Austroclima sp.	-0.09	-0.12	-0.14	-0.12	0.15	-0.19	-0.16	-0.06	-0.19	-0.15	-0.23	0.22	0.22	-0.06	-0.08	-0.07	0.20	-0.12	0.13	-0.04	0.04	-0.19	-0.09	-0.11	-0.03	-0.09	0.11	0.01	0.04	0.00	-0.08	-0.08	0.07	0.06
Austrosimulium sp.	0.19	-0.03	0.17	0.18	-0.02	-0.11	0.00	-0.06	-0.02	0.02	-0.03	0.04	0.14	0.19	0.18	0.19	-0.08	0.16	-0.15	0.18	0.12	-0.03	0.15	0.15	0.25	-0.07	-0.02	0.07	0.11	0.08	-0.02	-0.02	0.06	0.08
Coloburiscus humeralis	-0.07	0.22	-0.09	-0.08	-0.11	0.00	0.07	0.08	0.13	-0.06	0.07	0.07	0.08	-0.12	-0.11	-0.12	-0.16	-0.13	-0.16	-0.13	0.13	-0.10	0.03	0.07	-0.09	0.04	0.09	-0.01	0.04	0.17	-0.04	-0.05	-0.05	-0.16
Deleatidium sp.	-0.17	0.55	-0.14	-0.18	-0.42	0.33	0.44	0.56	0.41	0.31	0.36	-0.11	0.02	-0.27	-0.28	-0.27	-0.34	-0.24	0.00	-0.27	0.20	0.06	-0.32	0.21	-0.09	0.05	0.18	0.09	0.19	0.09	-0.13	-0.20	-0.18	-0.23
Eriopterini sp.	-0.07	0.15	-0.07	-0.07	-0.06	0.03	0.08	0.07	0.06	0.08	0.09	0.05	0.06	-0.09	-0.10	-0.09	-0.09	-0.04	0.01	-0.08	0.12	-0.08	-0.03	0.08	-0.14	-0.10	0.09	-0.16	-0.20	0.12	-0.01	0.05	0.05	0.23
Hydrobiosis sp.	0.17	0.24	0.20	0.20	-0.27	0.17	0.26	0.12	0.27	0.20	0.36	-0.15	-0.19	0.11	0.11	0.12	-0.21	0.19	-0.16	0.11	0.17	-0.07	0.26	0.22	0.05	0.04	0.11	0.04	0.07	0.04	-0.17	-0.07	-0.05	-0.09
Maoridiamesa sp.	-0.03	0.45	0.05	0.00	-0.43	0.29	0.40	0.30	0.37	0.42	0.47	-0.34	-0.25	-0.11	-0.11	-0.10	-0.38	-0.02	-0.12	-0.12	0.24	0.07	0.00	0.34	-0.01	0.08	0.14	0.06	0.04	0.03	-0.16	-0.12	-0.14	0.04
Nesameletus sp.	-0.16	0.32	-0.17	-0.17	-0.13	0.11	0.14	0.26	0.20	0.02	0.10	0.09	0.10	-0.21	-0.22	-0.21	0.00	-0.22	-0.03	-0.20	0.02	0.06	-0.30	0.00	-0.01	0.25	0.38	0.36	0.43	-0.19	-0.35	-0.40	-0.28	-0.47
Neurochorema confusum	0.13	0.26	0.18	0.15	-0.30	0.23	0.29	0.10	0.26	0.31	0.34	-0.26	-0.19	0.08	0.08	0.09	-0.17	0.17	-0.06	0.08	0.12	0.02	0.15	0.23	0.09	0.03	-0.02	-0.13	-0.17	0.10	-0.16	-0.05	-0.10	0.10
Olinga feredayi	-0.23	0.30	-0.23	-0.26	-0.13	0.03	0.05	0.22	0.16	-0.02	0.01	0.04	0.20	-0.28	-0.29	-0.28	-0.07	-0.30	-0.03	-0.27	0.02	0.01	-0.35	-0.02	-0.06	0.27	0.42	0.27	0.32	-0.13	-0.27	-0.33	-0.29	-0.39
Oxyethira albiceps	0.28	0.00	0.31	0.30	-0.07	-0.04	0.04	-0.12	0.03	0.07	0.10	-0.17	-0.12	0.28	0.28	0.29	-0.12	0.31	-0.13	0.28	0.02	0.02	0.34	0.13	0.27	0.17	-0.07	-0.02	-0.07	-0.11	-0.15	0.08	-0.18	0.11
Potamopyrgus antipodarum	0.21	-0.72	0.06	0.18	0.64	-0.53	-0.64	-0.60	-0.57	-0.66	-0.61	0.63	0.31	0.33	0.31	0.31	0.55	0.24	-0.11	0.35	-0.35	-0.23	0.26	-0.51	0.13	-0.22	-0.16	-0.21	-0.28	0.07	0.22	0.24	0.23	0.31
Psilochorema sp.	-0.20	0.41	-0.16	-0.19	-0.32	0.34	0.36	0.46	0.31	0.31	0.33	-0.14	-0.15	-0.27	-0.28	-0.27	-0.17	-0.23	0.07	-0.27	0.13	0.11	-0.31	0.18	-0.03	-0.04	0.05	-0.03	0.04	0.05	-0.04	-0.04	-0.09	-0.10
Pycnocentria sp.	-0.16	0.18	-0.16	-0.18	-0.08	0.03	0.06	0.18	0.10	-0.01	-0.05	0.03	0.23	-0.19	-0.18	-0.18	-0.11	-0.22	-0.10	-0.19	0.16	-0.01	-0.22	0.14	-0.13	-0.04	0.15	0.00	0.16	0.08	-0.08	-0.12	0.04	-0.08
Pycnocentrodes sp.	-0.18	0.34	-0.13	-0.19	-0.23	0.17	0.18	0.26	0.25	0.12	0.15	-0.16	0.06	-0.24	-0.23	-0.24	-0.28	-0.22	-0.25	-0.26	0.08	0.19	-0.18	0.19	-0.01	0.17	0.20	0.15	0.26	0.06	-0.19	-0.24	-0.16	-0.29
Tanytarsus vespertinus	0.18	0.06	0.20	0.21	-0.09	0.03	0.10	-0.09	0.10	0.06	0.21	-0.09	-0.18	0.16	0.17	0.17	-0.13	0.24	-0.14	0.15	-0.03	-0.01	0.37	0.05	0.00	-0.05	-0.19	-0.28	-0.29	0.25	0.11	0.22	0.13	0.27
Zelandobius sp.	-0.02	0.16	0.02	-0.03	-0.17	0.06	0.12	0.18	0.12	0.11	0.05	-0.15	0.08	-0.05	-0.04	-0.04	-0.18	-0.06	-0.03	-0.06	0.11	-0.04	-0.12	0.10	0.05	0.15	0.08	-0.04	-0.03	0.08	-0.11	-0.18	-0.07	0.12
Zelandoperla sp.	-0.24	0.63	-0.13	-0.22	-0.54	0.44	0.54	0.56	0.48	0.47	0.46	-0.40	-0.19	-0.34	-0.32	-0.33	-0.43	-0.29	-0.03	-0.36	0.34	0.20	-0.32	0.45	-0.05	0.16	0.17	0.09	0.18	0.10	-0.16	-0.27	-0.15	-0.23
Zephlebia sp.	-0.22	-0.16	-0.29	-0.25	0.27	-0.30	-0.27	-0.13	-0.30	-0.29	-0.33	0.32	0.23	-0.17	-0.17	-0.18	0.18	-0.20	0.01	-0.17	-0.27	0.04	-0.14	-0.31	-0.09	-0.14	-0.03	-0.16	-0.15	0.11	0.13	0.11	0.11	0.18

Appendix 8: Pearson's correlations between hydrological principal components and community structure from 138 sampling occasions in the Whanganui Catchment, New Zealand.

	MPC1	MPC2	MPC3	TPC1	TPC2	IPC1	IPC2	IPC3
Isimp	0.30	-0.18	0.04	0.32	0.19	-0.21	0.26	-0.18
Simp	0.36	0.05	0.11	0.23	0.09	0.00	0.35	-0.10
IBP	0.27	-0.14	0.04	0.33	0.17	-0.14	0.23	-0.20
BP	0.33	0.09	0.11	0.24	0.04	0.08	0.33	-0.13
Imarg	-0.24	0.46	0.04	-0.12	-0.17	0.42	-0.18	0.19
Marg	-0.24	0.31	0.03	-0.01	-0.11	0.27	-0.21	0.13
EPT	-0.54	0.14	0.02	-0.38	-0.31	0.28	-0.48	0.28
EPT-Oxy	-0.58	0.09	0.01	-0.32	-0.40	0.26	-0.52	0.29
IEPT	-0.55	-0.03	0.05	-0.34	-0.29	0.12	-0.52	0.23
IEPT-Oxy	-0.59	-0.07	0.04	-0.27	-0.38	0.12	-0.56	0.24
MCI	-0.52	0.28	0.00	-0.33	-0.27	0.45	-0.44	0.30
QMCI	-0.62	0.04	-0.05	-0.30	-0.35	0.22	-0.58	0.28
SQMCI	-0.62	0.07	-0.05	-0.28	-0.35	0.25	-0.58	0.27

Appendix 7: Pearson's correlations between hydrological variables and biotic indices from 138 sampling occasions in the Whanganui Catchment, New Zealand.

	Q	FRE3	L10	H10	Evapo	Rain24	MAR	RMAR	q	StdL10	StdH10	DF	FF	Area	peri	RD	Slope	Order	Rslope	CS	DSO	DMO	DLO	Dd	T1	T7	T14	T21	T28	DT20	DT40	DT60	D20	D40
Isimp	0.22	-0.39	0.17	0.22	0.26	-0.10	-0.20	-0.34	-0.16	-0.16	-0.13	0.14	-0.03	0.27	0.26	0.26	0.22	0.30	-0.05	0.28	-0.21	-0.07	0.30	-0.19	0.03	-0.21	-0.33	-0.33	-0.37	0.09	0.22	0.30	0.18	0.38
Simp	0.34	-0.20	0.32	0.35	0.03	0.08	0.02	-0.19	0.07	0.01	0.07	0.01	0.01	0.34	0.34	0.34	-0.02	0.37	-0.17	0.34	-0.05	-0.16	0.38	-0.04	-0.03	-0.19	-0.24	-0.26	-0.28	0.09	0.11	0.19	0.11	0.24
IBP	0.20	-0.32	0.17	0.21	0.19	-0.05	-0.13	-0.28	-0.11	-0.10	-0.07	0.09	-0.07	0.24	0.24	0.24	0.16	0.29	-0.02	0.25	-0.18	-0.06	0.28	-0.14	0.03	-0.22	-0.34	-0.34	-0.36	0.14	0.22	0.29	0.20	0.38
BP	0.32	-0.13	0.32	0.33	-0.04	0.14	0.09	-0.12	0.13	0.08	0.15	-0.06	-0.06	0.31	0.31	0.31	-0.07	0.36	-0.14	0.30	-0.01	-0.15	0.37	0.01	-0.06	-0.21	-0.24	-0.27	-0.26	0.14	0.11	0.19	0.13	0.23
Imarg	-0.12	0.60	-0.04	-0.12	-0.47	0.25	0.42	0.43	0.38	0.35	0.34	-0.32	-0.01	-0.23	-0.21	-0.22	-0.49	-0.22	-0.14	-0.25	0.40	-0.03	-0.16	0.40	-0.06	0.08	0.17	0.06	0.18	0.17	-0.14	-0.17	-0.11	-0.16
Marg	-0.17	0.46	-0.11	-0.17	-0.30	0.12	0.27	0.30	0.21	0.22	0.20	-0.22	-0.02	-0.24	-0.22	-0.23	-0.32	-0.23	-0.08	-0.25	0.30	-0.03	-0.16	0.29	-0.02	0.02	0.09	-0.04	0.07	0.23	-0.05	-0.08	-0.01	-0.01
EPT	-0.43	0.52	-0.39	-0.45	-0.29	0.23	0.29	0.53	0.29	0.21	0.17	-0.10	0.06	-0.51	-0.51	-0.51	-0.21	-0.53	0.15	-0.51	0.28	-0.01	-0.54	0.17	-0.14	0.24	0.32	0.34	0.49	-0.12	-0.27	-0.34	-0.23	-0.55
EPT-Oxy	-0.47	0.49	-0.44	-0.49	-0.26	0.24	0.28	0.54	0.27	0.18	0.15	-0.06	0.07	-0.55	-0.54	-0.54	-0.17	-0.57	0.18	-0.55	0.25	0.00	-0.58	0.13	-0.20	0.16	0.29	0.30	0.46	-0.05	-0.18	-0.32	-0.13	-0.51
IEPT	-0.49	0.37	-0.47	-0.50	-0.12	0.10	0.15	0.39	0.12	0.08	0.04	0.04	0.07	-0.53	-0.53	-0.53	-0.05	-0.55	0.24	-0.52	0.18	-0.04	-0.53	0.02	-0.11	0.17	0.30	0.32	0.46	-0.15	-0.22	-0.30	-0.16	-0.48
IEPT-Oxy	-0.52	0.36	-0.51	-0.54	-0.11	0.12	0.15	0.41	0.12	0.07	0.04	0.07	0.07	-0.56	-0.56	-0.57	-0.02	-0.59	0.26	-0.56	0.15	-0.04	-0.57	-0.01	-0.18	0.09	0.27	0.28	0.42	-0.07	-0.13	-0.28	-0.07	-0.45
MCI	-0.38	0.65	-0.30	-0.39	-0.46	0.35	0.45	0.61	0.44	0.37	0.34	-0.26	0.04	-0.49	-0.48	-0.48	-0.39	-0.49	0.07	-0.50	0.33	-0.01	-0.48	0.27	-0.16	0.17	0.29	0.27	0.40	-0.10	-0.23	-0.32	-0.26	-0.44
QMCI	-0.54	0.49	-0.50	-0.55	-0.21	0.19	0.24	0.49	0.22	0.16	0.11	-0.05	0.06	-0.60	-0.59	-0.60	-0.15	-0.61	0.18	-0.60	0.18	0.05	-0.62	0.08	-0.19	0.14	0.29	0.29	0.42	-0.06	-0.14	-0.30	-0.17	-0.48
SQMCI	-0.53	0.52	-0.49	-0.55	-0.25	0.21	0.27	0.50	0.25	0.18	0.15	-0.08	0.04	-0.60	-0.59	-0.60	-0.19	-0.60	0.17	-0.60	0.19	0.05	-0.58	0.10	-0.21	0.15	0.27	0.27	0.39	-0.03	-0.14	-0.29	-0.17	-0.45

Appendix 9: Pearson's correlations between landscape varables from 173 sites in the Whanganui Catchment, New Zealand.

	RCiP	BG	IF	PF	PP	Scr	Tuss	Urb	RIF	RPF	RPP	RScr	RTuss	Rash	Rall	RMSS	Ash	All	MSS	Vol	OR	Pum	Loam	Rec	Min	OS	RRes	RLoam	RPum	RSO	Rnat	nat	RTree	Tree
CiP	0.26	-0.02	0.19	-0.62	0.54	-0.08	-0.06	0.14	0.05	-0.52	0.38	-0.11	-0.06	0.18	0.07	0.00	0.23	0.01	0.15	-0.03	0.03	0.02	0.44	0.04	-0.03	-0.24	0.12	0.15	0.13	0.06	-0.07	0.07	-0.30	-0.39
RCiP		-0.17	-0.03	-0.10	0.33	-0.02	-0.16	0.03	-0.19	-0.19	0.61	-0.26	-0.09	-0.09	0.16	-0.05	-0.01	-0.01	0.09	-0.17	0.11	-0.10	0.19	-0.09	-0.17	0.05	0.20	-0.05	-0.03	0.07	-0.40	-0.17	-0.31	-0.12
BG			-0.26	-0.25	-0.29	-0.10	0.68	0.24	0.34	-0.12	-0.26	-0.14	0.32	0.10	0.04	-0.01	0.42	0.06	-0.43	0.91	-0.05	0.24	-0.28	0.64	0.95	-0.47	-0.04	0.16	0.31	-0.21	0.33	0.23	0.24	-0.43
IF				-0.31	-0.14	-0.12	-0.41	0.01	0.26	-0.31	0.05	-0.02	-0.30	0.03	-0.05	0.11	-0.04	-0.13	0.26	-0.17	0.18	-0.11	0.17	-0.26	-0.25	0.15	0.06	-0.01	-0.10	0.28	0.09	0.47	0.05	0.56
PF					-0.29	-0.26	-0.18	-0.20	-0.23	0.81	-0.20	-0.03	-0.11	-0.20	-0.10	-0.11	-0.33	-0.03	-0.14	-0.21	-0.01	-0.12	-0.24	-0.20	-0.25	0.34	-0.10	-0.14	-0.20	-0.23	-0.27	-0.56	0.32	0.62
PP						-0.25	-0.40	0.20	-0.22	-0.21	0.62	-0.28	-0.27	-0.04	0.02	-0.01	-0.12	-0.20	0.30	-0.28	0.03	-0.41	0.60	-0.11	-0.29	0.14	0.20	-0.03	-0.17	0.10	-0.51	-0.59	-0.35	-0.37
Scr							-0.08	-0.21	0.04	-0.23	-0.17	0.38	0.07	0.16	-0.11	0.19	0.01	-0.12	0.15	-0.06	-0.15	0.19	-0.20	-0.13	-0.11	0.00	-0.12	-0.01	0.04	0.22	0.34	0.50	-0.12	-0.33
Tuss								0.05	0.10	-0.12	-0.34	0.13	0.63	0.08	0.23	-0.15	0.42	0.51	-0.49	0.54	-0.09	0.52	-0.36	0.56	0.69	-0.58	-0.09	0.18	0.41	-0.28	0.45	0.36	0.02	-0.49
Urb									0.06	-0.18	0.31	-0.28	-0.18	0.02	0.23	-0.09	0.41	0.02	-0.33	0.13	0.20	0.16	0.13	0.26	0.21	-0.32	0.24	-0.10	0.20	-0.12	-0.23	-0.08	-0.06	-0.16
RIF										-0.26	-0.39	-0.30	-0.12	0.33	-0.30	0.10	0.26	-0.08	-0.16	0.30	-0.05	0.25	-0.09	0.22	0.28	-0.29	0.02	0.21	0.01	0.07	0.60	0.33	0.77	0.02
RPF											-0.32	-0.17	-0.07	-0.31	-0.19	0.09	-0.36	-0.10	-0.08	-0.07	-0.09	-0.20	-0.19	-0.17	-0.13	0.34	-0.11	-0.26	-0.23	-0.22	-0.38	-0.51	0.41	0.46
RPP												-0.33	-0.25	-0.15	0.42	-0.25	-0.01	-0.11	0.16	-0.26	0.26	-0.28	0.41	-0.13	-0.26	0.13	0.23	-0.16	0.13	0.04	-0.68	-0.33	-0.58	-0.14
RScr													0.00	0.04	0.05	0.03	-0.11	0.27	0.13	-0.06	-0.16	0.22	-0.22	-0.23	-0.11	0.01	-0.16	0.24	-0.08	0.21	0.47	0.33	-0.39	-0.04
RTuss														0.15	0.05	-0.18	0.23	0.13	-0.22	0.16	-0.08	0.17	-0.21	0.55	0.43	-0.28	-0.08	0.00	0.31	-0.16	0.32	0.26	-0.16	-0.34
Rash															-0.31	-0.37	0.45	-0.04	-0.30	0.12	-0.08	0.33	0.16	0.11	0.10	-0.41	-0.09	0.42	0.21	-0.21	0.38	0.19	0.11	-0.15
Rall																-0.48	0.36	0.49	-0.38	0.08	0.29	0.35	0.11	-0.11	0.00	-0.34	0.23	-0.18	0.54	-0.32	-0.20	0.05	-0.41	-0.13
RMSS																	-0.40	-0.26	0.51	-0.06	-0.13	-0.37	-0.14	0.03	-0.02	0.38	-0.16	-0.08	-0.48	0.39	0.03	0.11	0.15	0.00
Ash																		0.24	-0.84	0.33	0.20	0.66	0.20	0.35	0.43	-0.86	0.00	0.16	0.68	-0.51	0.24	0.29	0.01	-0.32
All																			-0.37	0.07	0.01	0.63	-0.15	-0.06	0.01	-0.45	0.01	0.08	0.34	-0.28	0.18	0.20	-0.14	-0.13
MSS																				-0.37	-0.18	-0.67	-0.01	-0.28	-0.42	0.76	0.03	-0.07	-0.63	0.66	-0.13	-0.04	-0.20	0.10
Vol																					-0.06	0.26	-0.22	0.31	0.79	-0.40	-0.05	0.15	0.28	-0.19	0.28	0.22	0.24	-0.33
OR																						-0.04	0.14	-0.02	-0.05	-0.02	0.07	-0.12	0.19	-0.11	-0.19	-0.01	-0.10	0.14
Pum																							-0.20	0.03	0.19	-0.81	-0.06	0.13	0.58	-0.37	0.44	0.42	0.10	-0.19
Loam																								-0.22	-0.29	-0.20	0.05	0.13	0.06	-0.16	-0.33	-0.26	-0.21	-0.08
Rec																									0.73	-0.35	0.05	0.14	0.16	-0.14	0.26	0.12	0.10	-0.39
Min																										-0.46	-0.04	0.14	0.32	-0.20	0.34	0.24	0.18	-0.42
OS																											0.01	-0.24	-0.65	0.49	-0.36	-0.31	-0.05	0.42
RRes																												-0.16	-0.23	0.03	-0.14	-0.09	-0.06	-0.04
RLoam																													-0.18	-0.29	0.36	0.11	0.03	-0.13
RPum																														-0.45	0.08	0.25	-0.14	-0.26
RSO																															0.15	0.17	-0.07	0.03

Appendix 10: Pearson's correlations between landscape and biotic variables from 173 sites in the Whanganui Catchment, New Zealand.

	CiP	RCiP	BG	IF	PF	PP	Scr	Tuss	Urb	RIF	RPP	RScr	RTuss	Rash	Rall	RMSS	Ash	All	OR	Pum	Loam	Rec	RRes	RLoam	RPum	RSO	Rnat	nat	Tree	Prin1	Prin2	Prin3
Isimp	0.03	-0.16	-0.04	-0.12	0.06	0.17	0.01	-0.13	0.13	-0.08	0.03	-0.08	-0.12	-0.22	-0.07	0.22	-0.17	-0.09	-0.02	-0.22	-0.05	0.05	0.09	-0.12	-0.18	0.02	-0.18	-0.20	-0.05	-0.19	-0.03	0.00
Simp	-0.11	-0.15	-0.02	-0.10	0.04	0.14	-0.01	-0.10	0.15	-0.05	0.04	-0.09	-0.12	-0.22	-0.04	0.25	-0.16	-0.08	-0.01	-0.19	-0.09	0.06	0.02	-0.15	-0.14	0.02	-0.16	-0.17	-0.04	-0.16	-0.04	-0.04
Imarg	0.26	0.07	0.17	0.29	-0.39	-0.22	0.09	0.26	-0.05	0.10	0.00	0.26	0.08	0.24	0.25	-0.20	0.30	0.25	0.02	0.41	-0.01	-0.03	-0.02	0.11	0.35	0.08	0.32	0.51	-0.10	0.38	0.11	0.20
MCI	-0.09	0.02	0.17	0.29	-0.24	-0.38	0.17	0.22	-0.18	0.29	-0.15	0.19	0.14	0.29	0.05	-0.16	0.27	0.10	0.00	0.35	-0.12	0.01	-0.13	0.17	0.32	0.05	0.45	0.53	0.03	0.42	-0.11	0.14
QMCI	-0.08	0.04	-0.02	0.26	-0.07	-0.30	0.13	0.06	-0.45	0.16	-0.16	0.21	0.11	0.20	-0.05	-0.07	-0.01	0.00	-0.06	0.14	-0.15	-0.13	-0.11	0.09	0.14	0.10	0.34	0.35	0.15	0.19	-0.25	0.06
EPT-Oxy	-0.02	-0.01	-0.02	0.29	-0.21	-0.29	0.30	0.02	-0.41	0.15	-0.16	0.31	0.11	0.21	-0.10	0.03	-0.05	-0.02	-0.08	0.10	-0.11	-0.14	-0.13	0.14	0.10	0.24	0.41	0.47	0.06	0.21	-0.31	0.22
DEC1	-0.16	-0.05	-0.43	-0.06	0.44	0.13	-0.05	-0.43	-0.43	-0.28	-0.05	-0.02	-0.12	-0.30	-0.29	0.33	-0.65	-0.28	-0.08	-0.53	-0.04	-0.34	-0.14	-0.19	-0.41	0.12	-0.31	-0.41	0.34	-0.55	-0.37	-0.20
DEC2	0.18	-0.11	-0.04	-0.11	0.02	0.34	-0.24	-0.10	0.33	-0.14	0.08	-0.22	-0.12	-0.07	-0.04	0.08	0.07	0.01	0.10	-0.13	0.24	0.04	-0.04	0.01	-0.09	-0.23	-0.33	-0.33	-0.07	-0.19	0.24	-0.01
NoSp	0.25	0.06	0.11	0.24	-0.28	-0.12	0.01	0.16	0.03	0.04	0.08	0.19	-0.03	0.16	0.22	-0.15	0.23	0.16	0.05	0.27	0.06	-0.05	0.00	0.06	0.28	0.06	0.15	0.33	-0.04	0.22	0.14	0.15
density	0.41	0.03	-0.04	0.21	-0.30	0.19	-0.04	-0.08	0.28	-0.04	0.30	-0.03	-0.13	0.08	0.17	-0.01	0.18	-0.04	0.15	-0.01	0.26	-0.04	0.06	0.00	0.17	0.00	-0.11	0.09	-0.08	-0.05	0.30	0.25
Aoteapsyche sp.	0.31	-0.07	0.10	0.16	-0.36	0.22	-0.05	-0.03	0.36	-0.05	0.28	-0.10	-0.03	0.10	0.11	-0.05	0.27	-0.17	0.21	-0.10	0.30	0.17	0.00	0.09	0.18	-0.07	-0.13	0.09	-0.18	0.00	0.34	0.28
Aphrophila neozelandica	0.18	0.02	0.18	-0.06	-0.25	-0.03	0.07	0.27	0.20	0.06	0.06	0.07	0.13	0.32	0.15	-0.23	0.50	0.08	0.03	0.37	0.17	0.16	-0.08	0.19	0.36	-0.17	0.16	0.20	-0.27	0.35	0.29	0.14
Archichauliodes diversus	0.28	0.10	-0.13	0.27	-0.24	0.12	0.02	-0.13	-0.13	-0.04	0.25	0.14	-0.14	0.08	0.13	-0.10	-0.07	-0.12	0.08	-0.11	0.28	-0.25	-0.03	0.10	0.07	0.21	0.01	0.15	0.01	-0.12	0.08	0.26
Austroclima sp.	0.10	0.21	-0.14	0.18	-0.06	0.11	-0.04	-0.18	-0.13	-0.01	0.14	0.05	-0.08	0.16	-0.11	0.01	-0.18	0.00	-0.12	-0.12	0.08	-0.12	-0.01	0.08	-0.05	0.20	-0.01	-0.01	0.09	-0.15	-0.04	0.16
Austrosimulium sp.	0.17	0.14	-0.01	-0.15	-0.09	0.25	-0.08	0.05	0.07	-0.11	0.24	-0.08	0.05	-0.06	0.17	-0.04	-0.02	0.16	-0.05	0.02	0.08	0.00	0.10	0.02	-0.01	-0.02	-0.14	-0.15	-0.21	-0.06	0.23	0.01
Coloburiscus humeralis	0.25	0.03	-0.06	0.34	-0.24	-0.03	0.09	-0.11	0.06	0.09	0.17	0.09	-0.15	0.25	0.12	-0.13	0.15	-0.05	0.05	0.14	0.22	-0.20	-0.03	0.09	0.22	0.06	0.08	0.26	0.06	0.04	0.11	0.24
Deleatidium sp.	0.18	-0.05	0.09	0.32	-0.42	-0.14	0.15	0.15	-0.11	0.21	-0.05	0.23	0.10	0.20	0.01	-0.01	0.10	0.01	0.02	0.11	0.01	0.06	0.04	0.18	0.14	0.18	0.39	0.49	-0.10	0.26	-0.08	0.36
Eriopterini sp.	0.23	0.11	0.10	-0.04	-0.20	0.20	-0.12	0.10	0.03	0.01	0.14	-0.04	0.04	0.07	0.07	-0.05	0.10	0.01	0.22	-0.04	0.25	0.13	0.04	0.04	0.10	-0.02	-0.01	-0.04	-0.20	0.02	0.24	0.15
Hydrobiosis sp.	0.23	0.02	0.20	0.14	-0.34	0.08	-0.05	0.12	0.39	0.09	0.24	0.00	-0.02	0.09	0.21	-0.23	0.25	0.00	0.05	0.14	0.11	0.16	0.21	0.05	0.16	0.10	0.07	0.18	-0.19	0.14	0.29	0.23
Maoridiamesa sp.	0.17	0.03	0.28	-0.05	-0.20	0.02	-0.07	0.24	0.24	0.24	0.05	-0.03	0.09	0.24	0.10	-0.21	0.36	0.04	0.11	0.23	0.07	0.30	0.06	0.17	0.20	-0.07	0.22	0.09	-0.22	0.29	0.25	0.16
Nesameletus sp.	0.07	-0.05	-0.01	0.53	-0.22	-0.27	0.05	-0.04	-0.17	0.17	-0.01	0.13	-0.04	0.05	0.08	-0.07	0.01	-0.09	0.07	0.01	0.05	-0.17	0.02	0.00	0.11	0.19	0.23	0.45	0.25	0.08	-0.15	0.19
Neurochorema confusum	0.08	0.05	0.35	-0.05	-0.18	-0.10	0.01	0.26	0.26	0.07	0.09	0.03	-0.02	0.06	0.17	-0.08	0.27	0.05	0.06	0.16	-0.03	0.24	0.10	-0.03	0.18	0.01	0.07	0.16	-0.19	0.23	0.19	0.07
Olinga feredayi	0.07	-0.03	-0.02	0.27	-0.14	-0.09	0.06	-0.07	-0.12	0.13	0.05	0.06	-0.08	0.19	0.12	-0.17	0.27	-0.17	0.18	0.09	0.17	-0.14	-0.08	-0.01	0.28	-0.06	0.12	0.22	0.10	0.09	0.06	0.12
Oxyethira albiceps	0.03	0.06	0.09	-0.08	-0.07	-0.03	0.03	0.14	0.33	-0.05	0.12	0.03	0.01	-0.03	0.25	-0.16	0.14	0.20	0.05	0.17	0.01	0.07	0.19	-0.02	0.06	0.01	-0.02	0.06	-0.12	0.10	0.20	-0.04
Potamopyrgus antipodarum	0.13	0.13	-0.38	-0.08	0.14	0.44	-0.04	-0.44	-0.10	-0.38	0.20	-0.06	-0.24	-0.24	-0.12	0.27	-0.40	-0.23	-0.09	-0.45	0.24	-0.27	0.09	-0.17	-0.37	0.08	-0.47	-0.43	0.05	-0.58	0.00	0.02
Psilochorema sp.	0.04	-0.04	0.15	0.14	-0.12	-0.17	0.03	0.11	-0.11	0.20	-0.13	0.13	0.04	0.09	-0.09	0.10	0.05	-0.13	0.01	0.02	-0.09	0.11	-0.03	0.12	0.05	0.06	0.28	0.23	0.02	0.17	-0.15	0.16
Pycnocentria sp.	0.06	0.17	-0.11	0.38	-0.11	-0.16	0.06	-0.12	-0.21	0.08	0.03	0.12	-0.08	0.03	-0.04	0.15	-0.12	-0.07	-0.07	-0.02	0.00	-0.23	-0.08	-0.04	0.01	0.32	0.12	0.28	0.22	-0.06	-0.19	0.18
Pycnocentrodes sp.	0.12	-0.05	-0.02	0.11	-0.15	-0.12	0.24	0.00	-0.07	0.03	0.05	0.12	0.00	0.25	0.15	-0.18	0.29	-0.05	0.09	0.23	0.15	-0.14	-0.02	0.03	0.29	-0.07	0.12	0.26	-0.04	0.17	0.10	0.10
Tanytarsus vespertinus	0.10	0.02	0.04	0.08	-0.14	0.13	-0.10	-0.01	0.36	-0.01	0.21	0.02	-0.12	0.04	0.14	-0.18	0.13	-0.03	0.08	0.05	0.17	0.08	0.05	0.05	0.07	0.08	-0.05	-0.01	-0.06	-0.01	0.22	0.10
Zelandobius sp.	0.07	-0.01	0.04	0.09	-0.11	-0.11	0.05	0.11	-0.03	0.02	-0.01	0.15	0.05	0.16	0.12	-0.21	0.25	0.16	-0.05	0.25	0.06	-0.01	-0.04	0.13	0.22	-0.14	0.15	0.19	-0.02	0.22	0.09	0.01
Zelandoperla sp.	0.05	-0.12	0.34	0.08	-0.23	-0.37	0.11	0.42	-0.10	0.33	-0.22	0.12	0.19	0.25	0.14	-0.26	0.38	0.19	0.06	0.47	-0.17	0.12	-0.03	0.16	0.36	-0.13	0.46	0.46	-0.13	0.55	0.01	0.06
Zephlebia sp.	-0.04	0.19	-0.29	0.06	0.03	0.11	0.18	-0.29	-0.19	-0.22	0.21	0.20	-0.18	0.01	-0.11	0.14	-0.34	-0.22	-0.14	-0.27	0.06	-0.22	-0.04	0.03	-0.18	0.29	-0.13	-0.05	0.08	-0.30	-0.16	0.11

Appendix 11: Correlations between the biotic indices and their canonical variables for the full landscape data set and biotic indices.

	V1	V2	V3	V4
Isimp	-0.48	0.33	0.00	-0.10
Simp	-0.42	0.15	-0.05	0.04
Imarg	0.85	-0.01	0.13	-0.49
MCI	0.87	-0.24	0.11	0.40
QMCI	0.74	0.00	-0.51	0.42
EPT-Oxy	0.76	0.37	-0.16	0.48

Appendix 12: Correlations between the landscape and their canonical variables for the full landscape data set and biotic indices.

	W1	W2	W3	W4
CiP	0.12	0.27	0.08	-0.59
RCiP	0.05	-0.18	-0.04	-0.08
BG	0.18	-0.24	0.33	-0.10
IF	0.43	0.11	-0.01	0.02
PF	-0.43	-0.27	-0.38	0.24
PP	-0.42	0.14	-0.05	-0.26
Scr	0.22	0.49	0.24	0.28
Tuss	0.28	-0.31	0.20	-0.20
Urb	-0.27	-0.27	0.62	-0.35
RIF	0.25	-0.21	0.17	0.26
RPP	-0.12	-0.02	0.05	-0.28
RScr	0.37	0.40	0.00	-0.03
RTuss	0.15	-0.09	0.04	0.11
Rash	0.35	-0.15	0.13	0.07
Rall	0.16	-0.24	0.10	-0.47
RMSS	-0.19	0.49	-0.06	0.19
Ash	0.29	-0.53	0.47	-0.22
All	0.20	-0.17	0.12	-0.35
OR	-0.02	-0.15	0.11	-0.07
Pum	0.46	-0.41	0.25	-0.26
Loam	-0.11	0.01	0.20	-0.14
Rec	-0.07	-0.23	0.27	-0.01
RRes	-0.11	-0.02	-0.05	-0.24
RLoam	0.18	-0.01	0.23	0.14
RPum	0.41	-0.35	0.21	-0.19
RSO	0.15	0.53	0.02	0.10
Rnat	0.55	0.07	0.16	0.25
Nat	0.73	0.18	0.30	0.05
Tree	-0.02	-0.14	-0.35	0.22

Appendix 13: Canonical correlation analysis of biological indices and catchment landscape variables for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

Canonical Variable	Canonical Correlation Co-efficient	Pr > F	Cumulative % explained of indices by indices canonical variants	Cumulative % explained of indices by landscape canonical variants
1	0.73	<.0001	0.51	0.27
2	0.58	<.0001	0.59	0.30
3	0.55	<.0001	0.67	0.32
4	0.46	0.008	0.74	0.34
5	0.27	0.687	0.89	0.35
6	0.16	0.912	1.00	0.35

Appendix 14: Canonical correlation analysis of biological indices and riparian landscape variables for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

Canonical Variable	Canonical Correlation Co-efficient	Pr > F	Cumulative % explained of indices by indices canonical variants	Cumulative % explained of indices by landscape canonical variants
1	0.70	<.0001	0.42	0.21
2	0.52	0.0002	0.51	0.23
3	0.36	0.162	0.70	0.26
4	0.29	0.492	0.85	0.27
5	0.23	0.708	0.94	0.27
6	0.15	0.814	1.00	0.28

Appendix 15: Correlations Between the community structure variables and their canonical variables for the full landscape data set and biotic indices.

	V1	V2	V3	V4	V5	V6	V7	V8
DEC1	-0.82	-0.19	0.23	-0.02	0.30	0.06	-0.02	0.02
DEC2	-0.35	0.55	-0.33	-0.17	-0.06	-0.09	-0.08	-0.14
NoSp	0.46	-0.04	0.39	-0.17	-0.15	-0.17	-0.11	0.07
density	0.15	0.42	0.31	-0.28	-0.13	0.07	-0.22	0.28
Aoteapsyche sp.	0.23	0.61	0.12	-0.16	0.03	0.26	-0.17	0.04
Aphrophila neozelandica	0.51	0.23	-0.02	-0.03	-0.07	-0.16	0.20	0.25
Archichauliodes diversus	0.20	0.06	0.66	-0.06	0.04	0.12	-0.22	0.32
Austroclima sp.	0.02	-0.12	0.40	0.31	0.04	-0.25	-0.21	0.22
Austrosimulium sp.	-0.09	0.12	-0.12	0.11	-0.24	-0.15	-0.25	0.35
Coloburiscus humeralis	0.33	0.07	0.53	-0.21	-0.12	-0.01	0.00	0.30
Deleatidium sp.	0.52	-0.15	0.33	-0.03	0.22	0.15	-0.42	0.08
Eriopterini sp.	0.16	0.20	0.01	0.10	0.22	0.05	-0.18	0.18
Hydrobiosis sp.	0.45	0.26	0.08	0.04	-0.33	0.31	-0.13	0.01
Maoridiamesa sp.	0.43	0.15	-0.17	0.11	0.02	0.11	0.06	0.26
Nesameletus sp.	0.36	-0.25	0.57	-0.37	0.14	0.32	-0.12	-0.02
Neurochorema confusum	0.41	0.13	0.03	0.06	-0.25	0.13	0.15	-0.10
Olinga feredayi	0.34	0.07	0.35	-0.38	0.43	0.02	0.31	0.12
Oxyethira albiceps	0.17	0.13	-0.12	0.07	-0.55	0.12	-0.03	-0.04
Potamopyrgus antipodarum	-0.63	0.30	0.24	0.11	0.10	-0.19	-0.15	-0.21
Psilochorema sp.	0.26	-0.19	0.18	-0.02	0.32	0.20	0.00	0.11
Pycnocentria sp.	0.18	-0.31	0.68	0.00	0.09	-0.15	-0.08	0.16
Pycnocentrodes sp.	0.33	0.05	0.28	-0.31	-0.03	-0.04	0.24	0.27
Tanytarsus vespertinus	0.22	0.28	0.11	0.18	-0.33	0.27	0.11	0.01
Zelandobius sp.	0.29	-0.01	0.01	-0.16	-0.01	-0.14	0.10	0.05
Zelandoperla sp.	0.60	-0.32	-0.12	-0.34	0.04	0.05	0.04	0.12
Zephlebia sp.	-0.09	-0.04	0.61	0.45	-0.02	-0.06	0.01	0.09

Appendix 16: Correlations between the landscape and their canonical variables for the full landscape data set and community structure variables.

	W1	W2	W3	W4	W5	W6	W7	W8
CiP	0.14	0.40	0.18	-0.19	0.01	-0.17	-0.50	0.30
RCiP	0.05	0.08	0.21	0.35	0.00	-0.30	-0.02	0.16
BG	0.49	-0.08	-0.39	0.00	0.01	0.12	0.00	-0.06
IF	0.20	-0.02	0.56	-0.39	0.17	0.11	-0.20	-0.20
PF	-0.57	-0.32	-0.10	0.04	-0.08	0.02	0.55	0.00
PP	-0.30	0.65	0.06	0.29	0.13	-0.15	-0.27	0.21
Scr	0.17	-0.14	0.07	0.04	-0.16	0.04	0.05	-0.07
Tuss	0.50	-0.25	-0.51	-0.01	-0.11	-0.04	-0.14	0.06
Urb	0.23	0.61	-0.17	0.03	-0.40	0.11	0.06	-0.09
RIF	0.38	-0.26	-0.20	-0.12	0.26	0.13	0.00	0.12
RPP	-0.02	0.56	0.32	0.15	-0.20	0.01	-0.07	0.29
RScr	0.15	-0.25	0.19	0.10	-0.10	-0.03	-0.04	-0.15
RTuss	0.20	-0.23	-0.40	0.05	0.08	0.10	-0.19	0.21
Rash	0.39	0.06	-0.06	0.03	0.11	-0.21	0.15	0.32
Rall	0.26	0.20	-0.01	-0.27	-0.33	-0.11	-0.02	0.11
RMSS	-0.32	-0.07	0.17	0.12	0.21	0.01	-0.25	-0.21
Ash	0.68	0.35	-0.33	-0.30	0.02	-0.24	0.27	0.15
All	0.21	-0.19	-0.34	-0.16	-0.35	-0.46	-0.25	0.05
OR	0.10	0.27	-0.07	-0.28	0.17	0.18	0.00	0.10
Pum	0.56	-0.22	-0.34	-0.28	-0.32	-0.42	0.18	0.10
Loam	-0.02	0.62	0.28	-0.02	0.19	-0.06	-0.02	0.22
Rec	0.37	0.12	-0.50	0.30	0.16	0.25	-0.08	-0.04
RRes	0.11	0.08	-0.05	0.08	-0.17	0.09	-0.28	-0.20
RLoam	0.23	0.00	-0.13	0.17	0.10	-0.01	-0.15	0.23
RPum	0.52	0.13	-0.10	-0.35	-0.06	-0.20	0.20	0.31
RSO	-0.03	-0.20	0.44	0.35	-0.07	0.22	-0.27	-0.13
Rnat	0.52	-0.50	-0.21	-0.01	0.19	0.13	-0.11	0.08
nat	0.66	-0.30	0.14	-0.31	-0.04	0.09	-0.25	-0.17
Tree	-0.33	-0.30	0.37	-0.28	0.07	0.11	0.31	-0.17

Appendix 17: Canonical correlation analysis of community structure and catchment landscape variables for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

Canonical Variable	Canonical Correlation Co-efficient	Pr > F	Cumulative % explained of the community by community canonical variants	Cumulative % explained of the community by landscape canonical variants
1	0.87	<.0001	0.12	0.09
2	0.82	<.0001	0.18	0.13
3	0.74	<.0001	0.29	0.19
4	0.70	<.0001	0.34	0.21
5	0.63	<.0001	0.38	0.23
6	0.60	<.0001	0.41	0.24
7	0.54	0.0037	0.44	0.25
8	0.50	0.056	0.48	0.26
16	0.00	.	0.66	0.28

Appendix 18: Canonical correlation analysis of community structure and riparian landscape variables for 173 sites sampled for benthic invertebrates between 1988 and 2000 in the Whanganui Catchment, New Zealand.

Canonical Variable	Canonical Correlation Co-efficient	Pr > F	Cumulative % explained of the community by community canonical variants	Cumulative % explained of the community by landscape canonical variants
1	0.84	<.0001	0.14	0.10
2	0.77	<.0001	0.21	0.14
3	0.64	<.0001	0.27	0.16
4	0.63	0.0006	0.29	0.17
5	0.53	0.0736	0.33	0.18
13	0.00	.	0.59	0.21