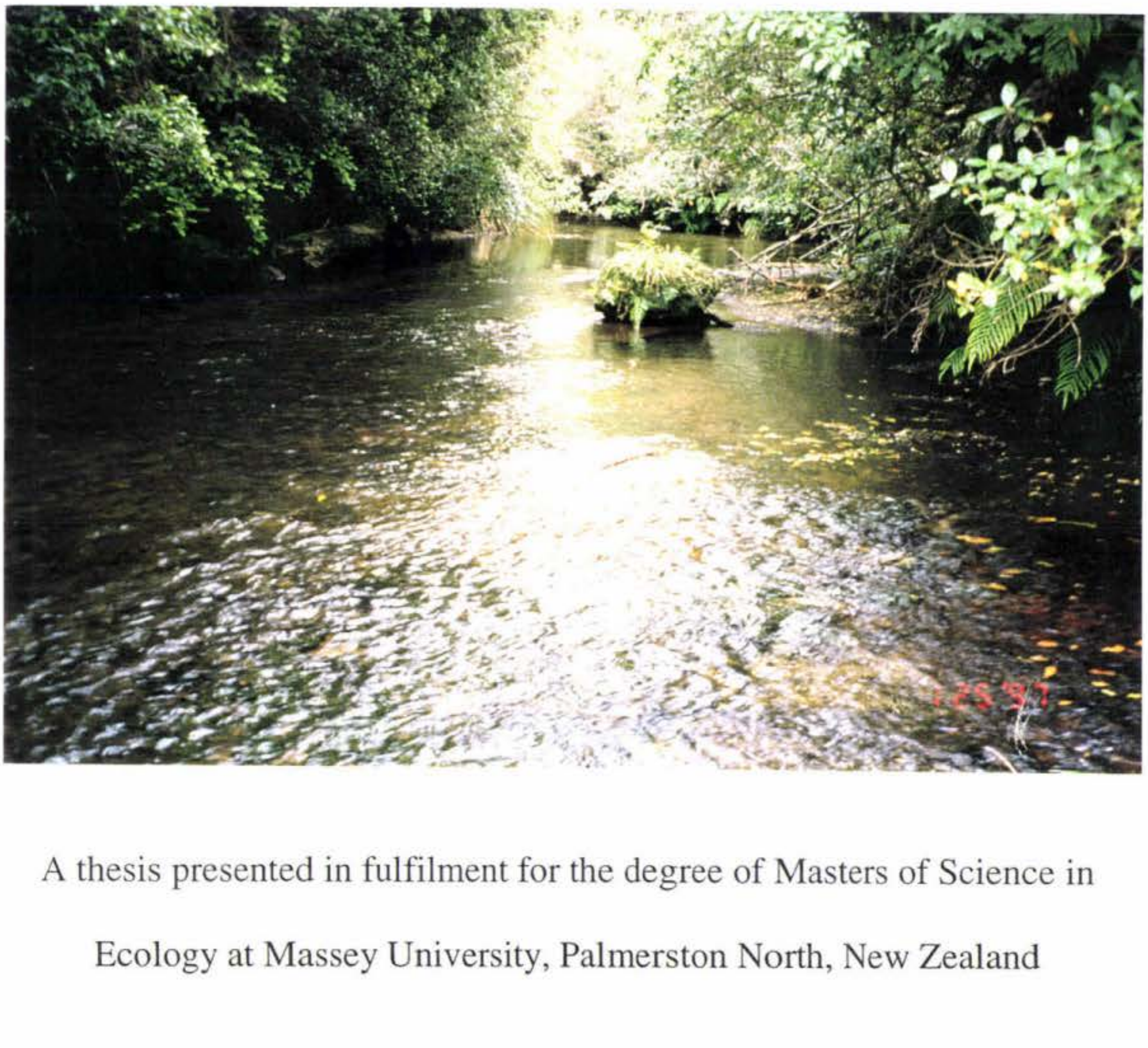
Leaf litter decomposition and stream macroinvertebrate communities of the Central Volcanic Plateau: the effects of landuse



A thesis presented in fulfilment for the degree of Masters of Science in

Ecology at Massey University, Palmerston North, New Zealand

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2000

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ABSTRACT

ABSTRACT

The effects of landuse on benthic macroinvertebrate community structure was assessed in 35 streams draining four different landuse activities (native (Beech and Broadleaf/Podocarp) forest, exotic (*Pinus radiata*) forest, hill country pasture and scrubland) around Lake Taupo, North Island, New Zealand, between January and March 1997. Ephemerotera (mainly *Deleatidium* sp.) were abundant in all landuse types. Diptera (Chironomidae), Coleoptera (Elmidae) and Trichoptera (*Pycnocentrodes aureola*) dominated invertebrate communities in open canopy pasture streams, where higher algal biomass existed. Native forest and exotic forest stream communities were considerably different. Native forest streams had higher water velocities, substrate stability and overhead cover, whereas exotic forest streams had low bed stability and high sand levels. In response to the higher sand levels in exotic streams, invertebrate communities had higher abundance of molluscs (*Potamopyrgus antipodarum*) and Coleoptera (Elmidae). Native stream communities were dominated by Ephemeroptera (*Deleatidium* sp. and *Colorburiscus humeralis*), Coleoptera (Elmidae) and Trichoptera (*Aoteapsyche colonica*). Landuse effects on invertebrates are different throughout New Zealand and will depend on local conditions, especially geology.

In December 1997 and January 1998, leaf litter decomposition and invertebrate colonisation were examined in 12 streams draining four different landuse activities around Lake Taupo. Mesh leaf tubes (0.2mm) and mesh leaf bags (3mm) containing dried Rangiora (*Brachyglottis repanda*) leaves were immersed into streams draining the four landuse types. Leaf tubes and bags were removed from streams after 14 days, 28 days and 42 days to measure the percentage of leaf weight lost from each leaf tube and bag and to assess the invertebrate colonisation of the leaf litter bags. Open canopy streams processed leaf litter in the tubes faster than closed canopy streams and it is likely that the increased temperatures and nutrient levels in open streams contributed to this phenomenon. Leaf bags were decomposed more rapidly in exotic streams where invertebrate densities in bags were highest among the landuse types. The low abundance of quality food available in exotic sites is likely to have contributed to the higher numbers of invertebrates feeding on the limited food resource. Invertebrate communities in all landuse types were distinctly different from each other, in contrast to benthic communities which were more similar. It appears that landuse does affect invertebrate communities in leaf bags, and this in turn influences leaf litter decomposition rates.

Keywords: abundance, algal biomass, benthic communities, exotic forest, hill country pasture, invertebrate colonisation, leaf litter decomposition, macroinvertebrate community structure, mesh bags, mesh tubes, native forest, percentage weight loss, scrubland.

CHAPTER ONE

General introduction

GENERAL INTRODUCTION

The landscape influences it's water bodies through multiple pathways and mechanisms, operating at different spatial scales (Allan & Johnson, 1997). A large-scale factor such as land use (Winterbourn, 1986) will have an important influence on the physicochemical characteristics of a stream (Richards *et al.*, 1997). Macroinvertebrate community structure is closely linked to these characteristics (Friberg *et al.*, 1997; Harding & Winterbourn, 1995; Quinn *et al.*, 1994; Quinn *et al.*, 1997) making them appropriate indicators for investigating environmental change (Stark, 1993).

Changes in land use practices such as those related to agricultural development are predicted to alter habitat characteristics, invertebrate community composition and water quality (Lenat, 1984; Corkum, 1990; Quinn & Hickey, 1990b). Changes in biotic structure are normally considered to be the likely result of changes to riparian vegetation giving rise to increased light levels and alterations to temperature and discharge patterns, and fertiliser and sediment runoff causing increases to instream nutrient levels and sedimentation (Cowie, 1985; Winterbourn, 1986; Prat & Ward, 1994; Quinn *et al.*, 1997).

Invertebrate community structure in response to land development generally changes from less tolerant taxa, such as, Ephemeroptera, Plecoptera and Trichoptera in forested streams to communities with lower taxonomic richness and a predominance of taxa sensitive to environmental extremes, such as Oligocheata and chironomids in response to agricultural development. Pastoral streams have generally been found to have lower taxonomic richness than streams draining less developed catchments (Harding & Winterbourn, 1995; Quinn *et al.*, 1997). In contrast, stable, mature forest vegetation provides shade, stabilises water temperature, and reduces erosion and sediment flow (Rounick & Winterbourn, 1982).

Previous studies (Allen, 1959; Quinn & Hickey, 1990; Reed *et al.*, 1994; Richards & Host, 1994; Townsend *et al.*, 1997; Winterbourn, 1986) illustrate that changes to landuse can effect benthic community structure. However, while it is clear that structure is affected, it is less clear how, or if, community function is affected. Although the definition of community function is unclear, it includes community process and attributes such as nutrient cycling and food processing (Minshall, 1988; Resh *et al.*, 1988; Richards, Johnson & Host, 1995).

Leaf processing, one measure of community function, is influenced by a number of factors including; chemistry; temperature and velocity, leaf species, microbial activity and invertebrate feeding (Superkropp, Klug & Cummins, 1975; Davis & Winterbourn, 1977; Meyer, 1980, Irons *et al.*, 1994). Land development usually causes a number of changes to streams that may influence leaf processing including increased nutrient inputs, flow variability, bank erosion and turbidity along with modifications to the temperature regime, bed structure and organic inputs (Wilcock, 1986; Quinn *et al*, 1992).

Catchment disturbances such as agriculture and plantation forestry can greatly influence the timing, quantity and quality of allocthonous inputs to streams (Sweeney, 1993; Tuchman & King, 1993; Delong & Brusven, 1994) and therefore have significant effects on stream communities (Newbold *et al.*, 1980; Webster & Waide, 1982).

Although the processing of terrestrial leaf material in streams has been extensively studied in the Northern Hemisphere, especially North America (see Webster & Benfield, 1986) there has been relatively few studies in New Zealand (Collier & Winterbourn, 1986; Davis & Winterbourn, 1977; Young *et al.*, 1994). Most New Zealand studies have compared the breakdown of leaf litter within a single landuse. However, none have compared the breakdown of leaf litter in streams from different landuse types. The aim of this study was to establish whether different landuse types effect invertebrate community structure and how this change in structure affects one aspect of community function, namely, food processing.

REFERENCES

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

Allan, K.R. 1959. Effect of land development on stream bottom faunas. *Proceedings of the New Zealand Ecological Society* **7:** 20-21.

Collier, K.J. and M.J. Winterbourn 1986. Processing of willow leaves in two suburban streams in Christchurch, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **20**:575-582.

Corkum, L.D. 1990. Intrabiome distributional patterns of lotic macroinvertebrate assemblages. *Canadian Journal of Fisheries and Aquatic Sciences* **47:** 2147- 2157.

Cowie, B. 1985. An analysis of changes in the invertebrate community along a southern New Zealand montane stream. *Hydrobiologia* **120:** 35-46.

Davis, S.F. and Winterbourn, M.J. 1977. Breakdown and colonization of Nothofagus leaves in a New Zealand stream. *Oikos* **28**:250-255.

DeLong, M.D and M.A Brusven 1994. Allochtohnous input of organic material from different riparian habitats of an agriculturally impacted stream. *Environmental Management* **18**: 59-71.

Friberg, N., M.J. Winterbourn, K. A Shearer. and S. E. Larsen 1997. Benthic communities of forest streams in the South Island, New Zealand: effects of forest type and location. *Archiv für Hydrobiologie* **138(3):** 289-306.

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.

Irons, J., M. Oswood, J. Stout and C. Pringle 1994. Latitudinal patterns in leaf litter breakdown: is temperature really important? *Freshwater Biology* **32**: 401-411.

Lenat, D.R. 1984. Agriculture and stream water quality: a biological evaluation of erosion control practices. *Environmental Management* **8:** 333-344.

Minshall, F. W. 1984. Aquatic insect-substratum relationships. In: Resh, V. H.; Rosenberg, D. M. ed. The ecology of aquatic insects, New York, Praeger, pp. 358-400.

Meyer, J.L. 1980. Dynamics of phosphorus and organic matter during leaf decomposition in a forest stream. *Oikos* **34**: 44-53.

Newbold, J.D., D.C. Erman and K.B. Roby 1980. Effects of logging on macroinvertebrates in streams with and without buffer strips. *Canadian Journal of Fisheries and aquatic Sciences* **37**: 1076-1085.

Prat, N. and J.V. Ward 1994. The tamed river. In: Margalefs, R., editor. Limnology Now: a Paradigm of Planetary Problems. Amsterdam: Elsevier Science. p 219-236.

Quinn, J.M. and C.W. Hickey 1990b. Magnitude of effects of substrate particle size, recent flooding and catchment development on benthic invertebrates in 88 New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research* **24:** 411-427.

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:** 579-597.

Quinn, J.M., A.B. Cooper, R.J. Davies-Colley, J.C.Rutherford and R.B. Williamson 1994. Land-use effects on New Zealand hill country streams and implications for riparian management. In: Link, G.L. and Naiman, R.J., editors. Proceedings of the International Workshop on the Ecology and Management of Aquatic-Terrestrial Ecotones. Seattle: University of Washington.

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

Reed, J.L., I.C. Campbell and P.C.E. Bailey 1994. The relationship between invertebrate assemblages and available food at forest and pasture sites in three south-eastern Australian streams. *Freshwater Biology* **32:** 641-650.

Resh, V.H., A.V. Brown, A.P. Covich, M.E. Gurtz, H.W. Li, G.W. Minshall, S.R. Reice, A.L. Sheldon, J.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. and G. Host 1994. Examining land-use influences on stream habitats and macroinvertebrates: a GIS approach. *Water Resources Bulletin* **30:** 729-738.

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.

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.

Suberkropp, K., M.J. Klug and K.W. Cummins 1975. Community processing of leaf litter in a woodland stream. *Verhandlungen, Internationalen Vereinigung fur theroretische und angewandte Limnologie* **19**: 1653 -1658.

Sweeny, B.W. 1993. Effects of streamside vegetation on macroinvertebrate communities of white clay creek in eastern North America. *Proceedings of the Academy of Natural Sciences Philadelphia* **144**: 381-340.

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

Tuchman, N.C and R.H King 1993. Changes in mechanisms of summer detritus processing between wooded and agricultural sites in Michigan headwater stream. *Hydrobiologia* **268**: 155-127

Webster, J.R and E.F. Benfield 1986. Vascular plant break-down in freshwater ecosystems. *Annual Review of ecology and Systematics* **17**: 567-594.

Webster, J.R and J.B. Waide 1992. Effects of forest clearcutting on leaf breakdown in a Southern Appalachian stream. *Freshwater Biology* **12**: 331-344

Wilcock, R.J. 1986. Agricultural run-off: a source of water pollution in New Zealand? *New Zealand Agricultural Science* **20**: 98-103.

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

Young, R.G., A.D. Huryn and C.R. Townsend 1994. Effects of agricultural development on processing of tussock leaf litter in high country New Zealand streams. *Freshwater Biology* **32**: 413-427.

CHAPTER TWO

Stream macroinvertebrate communities of the Central Volcanic Plateau: the effects of landuse

ABSTRACT

Benthic macroinvertebrate communities in streams draining four different landuse activities (native (Beech and broadleaf/podocarp) forest, exotic (*Pinus Radiata*) forest, hill country pasture and scrubland) were sampled in 35 first to third order streams around Lake Taupo, North Island, New Zealand, between January and March 1997. Ephemeroptera (mainly *Deleatidium* sp.) were abundant in all landuse types, but Diptera (Chironomidae), Coleoptera (Elmidae) and Trichoptera (*Pycnocentrodes aureola*) otherwise dominated invertebrate communities in pastoral sites. Forested streams of native and exotic were found to exhibit differences in invertebrate community structure. Native forest streams had higher water velocities, substrate stability and overhead cover and were dominated by Ephemeroptera (*Deleatidium* sp. and *Colorburiscus humeralis*), Coleoptera (Elmidae) and Trichoptera (*Aoteapsyche colonica*). Exotic streams had low bed stability and high levels of sand. Invertebrate communities at exotic sites had lower diversity and low numbers of Plecoptera and Ephemeroptera, but had higher numbers of molluscs (*Potamopyrgus antipodarum*) and Coleoptera (Elmidae). The fine nature of volcanic soils in this region are thought to have contributed to the increased sediment found at exotic sites. Increased sand levels in exotic streams may have caused the reduction of invertebrate abundance at these sites.

Keywords: abundance, benthic macroinvertebrate communities, diversity, exotic forest, hill country pasture, landuse, native forest, particulate organic matter, periphyton biomass, scrubland.

INTRODUCTION

The character of a stream or river reflects the integration of physical and biological processes occurring within their catchment (Johnson & Gage, 1997). Catchment landuse has been identified as a major large-scale factor affecting invertebrate community composition in New Zealand streams (e.g., Quinn & Hickey 1990a, 1990b; Scott *et al.*, 1994; Harding & Winterbourn, 1995). Whilst land use contributes to the determination of the physical nature of the stream channel and the substrata of it's bed, riparian vegetation determines the flows of energy and matter that provide the food resources for the stream inhabitants.

Anthropogenic disturbance resulting in alterations to the riparian vegetation through land use activities such as exotic forestry and pastoral farming are predicted to alter habitat characteristics, invertebrate composition and water quality (Lenat, 1984; Corkum, 1990; Quinn & Hickey, 1990a; Fahey & Rowe, 1992; Prat & Ward, 1994). Invertebrate communities are generally responsive to changes in their environment (Stark, 1985). Increased light, temperature, altered discharge patterns, increased nutrient inputs and sedimentation are all documented to have adverse effects on benthic faunas (Cowie, 1985; Winterbourn, 1986; Smith *et al.*, 1993; Prat & Ward, 1994; Quinn *et al.,* 1994).

Changes in invertebrate community structure in response to land development has been well documented in both New Zealand (e.g., Allan, 1959; Winterbourn, 1986; Quinn & Hickey, 1990b; Smith *et al.*, 1993; Quinn *et al.*, 1997; Townsend *et al.*, 1997) and overseas (Dance & Hynes, 1980; Reed *et al.*, 1994; Richards & Host, 1994; Ventura & Harper, 1996). These studies generally illustrate a shift from less tolerant taxa, such as, Ephemeroptera, Plecoptera and Trichoptera in forested streams to communities with lower taxonomic richness and a predominance of taxa sensitive to environmental extremes, such as, Oligocheata and chironomids in response to agricultural development.

Previous New Zealand studies (Harding & Winterbourn, 1995; Quinn *et al.,* 1994, Quinn *et al.,* 1997; Quinn & Hickey, 1990a,b) have not considered the

Central Volcanic Plateau and have focused on comparisons between native and pasture, native and exotic or exotic and pasture landuses. This study investigates differences in physico-chemical conditions and benthic invertebrate community structure in streams draining four different land use types: native forests (Broadleaf/podocarp and Beech), exotic plantation forests (*Pinus radiata*), pasture (sheep and beef) and scrubland around the perimeter of lake Taupo.

STUDY SITES

All study sites were first to third order streams situated around the perimeter of Lake Taupo, North Island, New Zealand (176° 05'E, 38° 42'S). The geology is predominantly faulted and tilted sequences of ignimbrite, lake sediment and lava underlain by basement greywacke and argillite of meszozoic-paleozoic origin (Pullar & Birrell, 1975). In contrast, surficial geology is poorly consolidated pumiceous ignimbrite erupted during the last 20,000 years from the central volcanic plateau (Lake Taupo) (Pullar & Birrell, 1975). Study sites ranged in altitude between 320m and 750m a.s.l. (Fig. 2.1.).

Sites were placed into four landuse categories (defined as the vegetation covering >80% of the catchment); beech and broadleaf/podocarp forests, exotic plantation forest, scrubland and hill country pasture. Native forest streams (Nf 1-Nf 10) are from the Kaimanawa (beech) and Pureora (podocarp/broadleaf) forests. Exotic *Pinus radiata* plantation forests Pf 1-Pf 10) are approximately 20 years old and had riparian vegetation consisting mainly of scrub and blackberry. Scrubland (Sc 1-Sc 5) had riparian growth that was often well developed and consisted of various flaxes, gorse, Kanuka and willows. Agricultural activity in hill country pasture catchments (Pa 1-Pa 10) consisted of low to moderate intensity sheep and beef grazing. Riparian vegetation (when apparent) in the pasture catchments usually consisted of ungrazed exotic pasture species with stock access to the stream channel predominant.

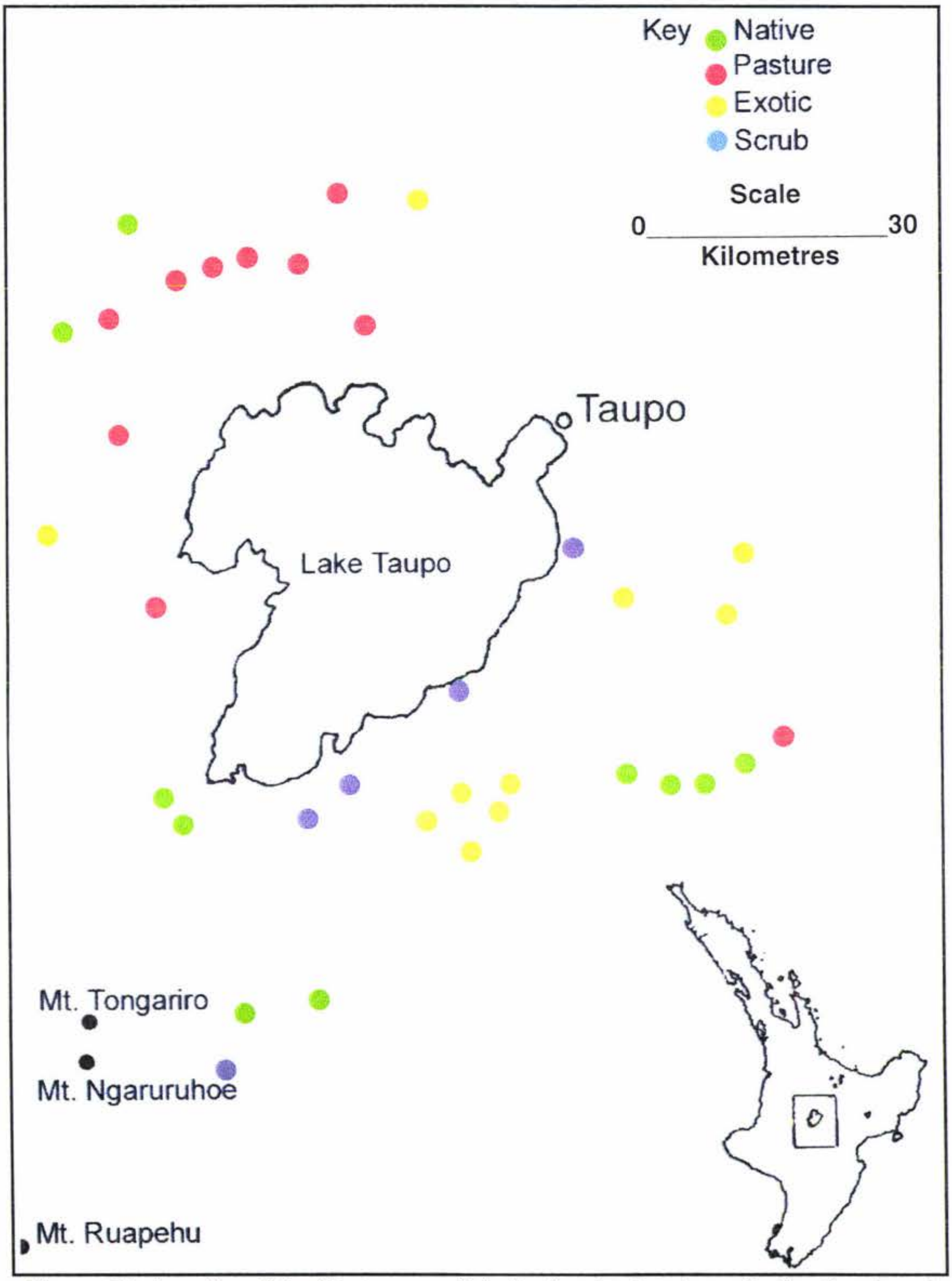

Figure 2.1. Location of the 35 streams within the four landuse types (native (beech and broadleaf/podocarp) forests, hill country pasture, exotic *Pinus Radiata* forest, and scrubland) which were sampled between January and March 1997 around Lake Taupo, North Island, New Zealand.

Plate 2.1. An exotic forest site (left) and a scrubland site (right) from a survey of 35 streams draining native forests, hill country pasture, exotic forest and scrubland catchments, around Lake Taupo, between January and March 1997.

Plate 2.2. A pasture site (top) and a native forest site (bottom) from a survey of 35 streams draining native forests, hill country pasture, exotic forest and scrubland catchments, around Lake Taupo, between January and March 1997.

MATERIALS AND METHODS

Sampling protocol

Macroinvertebrates were collected in four replicate 250 μm mesh, 0.11m^2 Surber samples between January and March 1997. They were preserved in 70% alcohol, sieved through a 250μm Endecott sieve and animals identified and enumerated with a 40× microscope using available keys (Chapman and Lewis, 1976; Cowley, 1978; McFarlane, 1951; Towns and Peters, 1996; Winterbourn, 1973; Winterbourn and Gregson, 1989). If taxa could not be identified to species they were assigned to apparent morphospecies. The remainder of the sample was separated into course (>0.1mm) and medium (>250μm-1mm) particulate organic matter (POM) using 1mm and 250μm Endecott sieves. This material was then dried at 80 °C for five days, weighed and then ashed for two hours at 600°C. Samples were then re-weighed to obtain an ash free dry weight (AFDW). Mass of organic matter was determined as the quantity of mass lost after ashing.

Concurrent with each invertebrate collection site, a range of physico-chemical parameters were measured (Table 2.1.). Conductivity, temperature and dissolved oxygen were measured using an Orio 122 conductivity and temperature meter and a YSI model 58 meter, respectively. Five water velocity readings, taken with a velocity head rod, were measured midstream at each sample point, along with 4 width and 5 depth measurements along the stream reach. Substrate composition: sand (<0.2cm diameter) gravel (0.2-6cm), small cobbles (6-12cm), large cobbles (13-26cm), boulders (>26cm) and bedrock, the percentage overhead cover and type of riparian vegetation (native forest, exotic woodland, scrub, crop/pasture and other) were visually assessed. 500ml water samples were taken back to the laboratory on ice and frozen. Samples were thawed before pH measurement using an Orion 250A pH meter. Suspended solids in this 500ml sample were measured by filtering through a preweighed Whatman GF/C filter (pore size=0.7μm) and dried overnight at 105 °C, before weighing again and subtracting the filter weight.

Algal biomass was assessed by extracting photosynthetic pigments (chlorophyll *a* and phaeophytin) in 90% acetone, for 24 hours at 5 °C, from five small cobbles (mean circumference = 10 cm) collected on each sampling occasion. Wavelength absorption was measured on a Jenway 6105 UV/Visible Spectrophotometer at 410, 430, 665 and 720nm. Total pigment concentration was calculated using the method of Moss (1967 a, b). Stone surface area was calculated using height, width and length measurements following Graham *et al.,* (1988) and used to express periphyton biomass per unit stone surface area.

Statistical analysis

Data were log $_{10}$ (x+1) transformed prior to analysis where appropriate to remove heterogeneity of variance. Differences in physicochemical and biological measures among land use types were analysed with two-way analysis of variance (ANOVA) using the SAS statistical package (SAS, 1989), and Duncan multiple range means tests (P=0.05). Landuse and stream order were treated as fixed effects. Correlations between the physicochemical variables were correlated with five indices of community structure using SAS (1989). These indices were the Berger-Parker dominance index (Berger & Parker, 1970), Margelefs index (Clifford & Stevenson, 1975), EPT (Lenat, 1988), MCI (Stark, 1985) and QMCI (Stark, 1993). The Berger-Parker dominance index is given by: $D = N_{max}/N$ where N = total number of individuals and N_{max} = number of individuals in the most abundant species. Indices that provided an indication of diversity were taxa richness, and Margalefs index is given by: D = (S-1)/1n/ N where S = species number and N = total number of individuals. An increase in Margalefs index indicates an increase in diversity, whereas an increase in the Berger-Parker index indicates a decrease in diversity (Death & Winterbourn, 1995). EPT scores consisted of the sum of Ephemeroptera, Plecoptera and Trichoptera taxa present divided by the total number of taxa.

The MCI weights taxa according to their tolerance to organic enrichment. Taxa that are characteristic of higher water quality score more highly than those commonly found in polluted conditions (Stark, 1993). The MCI is given by:

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

Where the site score is the sum of individual taxon scores (Stark, 1993).

The QMCI (Stark, 1993) is given by:

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

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

To examine overall trends in community structure, Detrended Correspondence Analysis (DECORANA) was performed using the PC-ORD statistical package (McCune & Mefford, 1995). Environmental variables were correlated with the DECORANA axes using Pearson's linear correlation.

RESULTS

Physicochemical characteristics

Streams with native forest catchments had the highest mean current velocities (ranging from 0.6 to 1.2m/s^{-1}), the most stable streambeds (Fig. 2.2.), highest percentage cover and occurred at higher altitudes (Table 2.1.). Conductivity, temperature, mean depth (ranging from 9 to 25cm) and percentage of sand (ranging from 5 to 10%) were all lowest in the native forest streams (Fig. 2.3.). Exotic forest streams had the narrowest (between 1 and 4.6m) and deepest channels (ranging from 12 to 42cm), highest percentage of sand (5 to 70%), least stable substrate and lowest current velocity (0.2 to 0.6m/s^{-1}). Scrub streams had the widest stream channels (mean ranged from 2.6 to 7.1m), highest water temperatures and conductivity and occurred at the lowest altitudes. Pasture streams had the lowest percentage cover. All streams had pH values in the circumneutral range (6.5 to 8.8) and suspended solid concentrations ranging from 2.9 to 803.3mg/l.

Periphyton and Particulate organic matter

Chlorophyll *a* ($F_{3,34}=4.39$, $P=0.01$) and total pigment ($F_{3,34}=4.92$, $P=0.01$) were highest at pasture sites, ranging from 0.33 to $2.54ug/\text{cm}^2$ and lowest at sites within exotic forests (0.07 to $0.78ug/\text{cm}^2$) (Fig. 2.4.).

Table 2.1. Physical and chemical characteristics, F-values,Degrees of freedom and P-values testing the null hypothesis that physico-chemical parameters were similar between landuse types recorded from 35 streams on the Central Volcanic Plateau, between January and March, 1997, which drain native forest, exotic forest, hill country pasture or scrub catchments.

Sites	Map (Ref)	Stream	Mean width	Mean depth	Mean velocity	Conductivity	D.O	Temperature	pH	Cover	Stability	Sand	Suspended	Altitude
	(1:250 000)	order	(m)	(cm)	(m/s-1)	(uS/cm-2)	(mg/L)	(0C)		%	(Total)	%	Solids (mg/L)	(masl)
Native														
Omoho	T19 461485	2	3.1 (2.5-4.0)	20.75 (13-24)	0.83	110.7	10.4	13.6	8.6	50	107	5	18.46	570
Waihi	T19 472463	3	3.3 (2.5-5.0)	14.25 (10-20)	0.57	31.1	11.2	12.3	8.8	90	81	5	7.14	480
Kokaho	T17 895452	3	10.0 (8.0-12.0)	19.5 (8-32)	1.19	34.9	10.2	11.6	7.3	70	50	5	2.86	490
Waimonoa	T17 920465	2	7.2 (6.0-9.0)	10.75 (8-24)	0.99	36.7	10.3	11.5	7.2	50	60	5	14.56	590
Waiharuru	U19 445830	1	2.2 (2.0-2.5)	15.25 (9-24)	0.77	22.9	9.8	8.9	6.7	80	82	5	11.76	790
Te Arero	U19 456981	2	2.8 (2.0-4.0)	14.25 (10-21)	0.93	31.7	9.8	9.4	6.7	100	76	5	26.09	780
Pirua	U19 465860	2	1.7 (1.5-2.5)	9 (6-14)	0.78	40.4	9.7	8.9	8.1	100	94	5	21.62	720
Manukatuata	U19 462910	1	1.5 (1.0-2.0)	9.5 (8-16)	0.68	32.8	9.8	9	7.4	100	99	10	6.45	820
Mangatawai	T19 221753	3	6.5 (6.0-7.0)	25.25 (19-31)	0.72	70.1	10.1	5.6	7.7	80	57	5	65.63	800
Waipa	T19 231754	3	5.2 (4.0-7.0)	22.5 (15-32)	0.65	73.6	7.8	4.7	6.5	100	55	5	21.21	710
Pasture														
Orongopioi	T18 632459	1	.88 (.5-1.0)	26.75 (20-35)	0.22	41.7	9.7	14.4	8.7	0	110	60	256.41	530
Otupoto	T18 777456	1	3.1 (2.0-4.0)	11.75 (8-14)	0.59	28.6	10.9	13.2	6.9	5	81	5	3.17	540
Waimeharua	T17 854431	2	3.7 (3.0-4.5)	23.5 (19-31)	0.94	28.6	9.9	11.6	6.9	20	79	10	23.19	420
Kokaho	T17 882495	3	6.1 (5.5-7.0)	19 (12-26)	1	50.5	11.1	12.9	8	10	52	5	2.86	460
Terengaogaio	T17 901534	3	4.1 (4.0-4.5)	22.75 (16-28)	0.91	60.6	10.2	14.1	8.6	50	89	5	803.33	470
Swampy	T17909555	3	2.0 (1.7-2.5)	26.5 (22-33)	0.77	77.3	10.3	13.9	7.8	30	101	10	144.18	490
Waikino	T17 955566	3	2.5 (1.5-4.0)	20 (18-22)	0.57	81.5	9.8	12.8	6.9	0	78	10	175.76	360
Whakarukia	T17 945579	2	1.6 (1.0-2.0)	32.25 (28-37)	0.03	79.1	8.1	16.3	7	0	73	10	39.44	370
Potungutungu	T17 940583	2	2.6 (2.0-3.0)	27.75 (24-32)	0.94	71.6	11.4	12.1	8	0	89	15	37.68	370
Taharua	U19 496945	2	3.1 (2.0-4.0)	24 (13-31)	0.19	63.1	8.8	10.3	8	0	113	5	2.86	650
Exotic														
Whanganui trib	T18 678402	1	1.1 (1.0-1.5)	12.5 (5-20)	0.16	57.9	9.9	12.3	7.8	100	90	70	41.18	560
Mokauteure	T17 966674	3	3.1 (2.5-3.5)	37.5 (24-51)	0.52	80.4	9.5	15.1	8.3	60	112	15	8.69	470
Waitetoko	T19 483694	1	1.0 (1.0-1.5)	20.25 (12-30)	0.47	98.7	8.2	11.2	7.8	60	92	20	63.64	570
Hingapo	T19 456685	1	1.2 (1.5-2.0)	46.5 (28-61)	0.38	91.7	9.5	10.4	8.6	80	68	50	36.11	590
Waitapu	U19 465786	2	2.1 (1.7-2.5)	36 (32-41)	0.27	95.8	8.5	10.5	8.1	90	86	60	61.11	540
Whataroa	T19 476692	2	1.5 (1.0-2.0)	18 (15-21)	0.5	82.5	8.9	10.5	7.9	90	84	35	29.41	540
Mangakahakaha	U18 571853	2	1.5 (1.0-2.0)	21.25 (16-28)	0.38	76.1	8.3	10.3	7.1	90	91	40	234.28	570
Mangamutu	U18 621803	3	1.3 (1.0-1.5)	19.5 (12-34)	0.22	86.3	8.8	11.1	7.9	60	80	65	85.29	380
Mangakowhitiwhiti	T19 455653	3	4.6 (4.0-6.0)	42.75 (35-48)	0.57	52.6	9.4	10.8	8.2	20	78	10	16.9	550
Tangihamutu	U18 623875	2	3.7 (3.5-4.0)	22.75 (12-31)	0.61	44	9.4	11.1	7.9	20	79	5	64.06	690
Scrub														
Oturere	T19 220756	3	6.0 (5.0-7.0)	18.25 (12-25)	0.72	113.3	12.4	9.6	7.5	50	65	10	171.01	870
Waiotaka	T19 432562	3	5.5 (4.0-7.0)	21.25 (16-27)	0.57	69.2	9.4	18.2	6.8	20	84	10	41.15	370
Waimarino	T19 461593	3	7.1 (6.5-8.0)	17 (12-22)	0.52	68.4	10.3	18.3	7.1	20	85	5	37.14	370
Waipehi	T18 541683	2	2.6 (2.0-3.0)	22.25 (9-24)	0.84	76.6	11.3	12.8	8.7	60	70	5	183.33	370
Waioratene	U18 651786	1	5.7 (5.0-6.0)	27.5 (21-34)	0.61	67.7	8.3	15.2	8.2	30	88	20	97.32	320
F-value			3.63	4.01	6.15	3.3	2.49	6.11	1.3	18.81	3.6	6.23	1.58	6.61
d,f			3,34	3,34	3,34	3,34	3,34	3,34	3,34	3,34	3,34	3,34	3,34	3,34
P-value			0.02	0.01	0.00	0.03	0.08	0.00	0.29	0.00	0.02	0.03	0.22	0.00

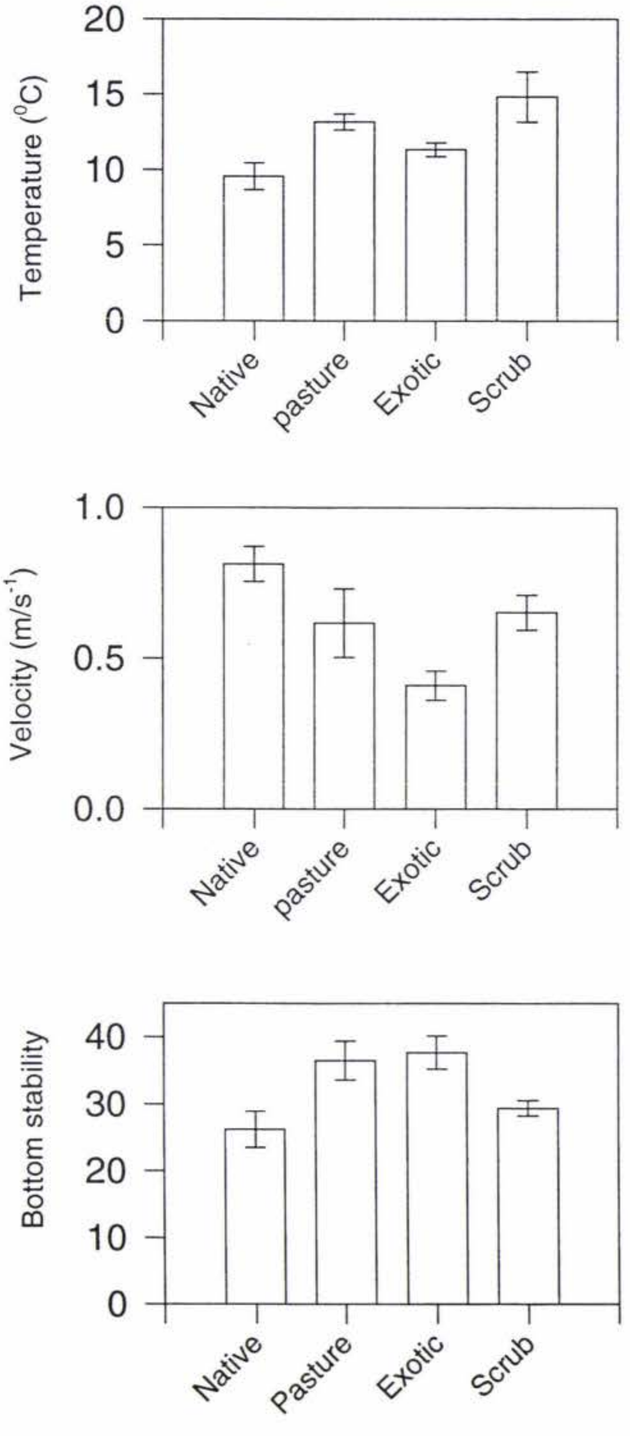

Figure 2.2. (Mean ± S.E.) (top) temperature (middle) velocity and (bottom) bottom stability component (Pfankuch stability index) measured in 35 streams around Lake Taupo between January and March 1997, which drain native forest, hill country pasture, exotic forest and scrubland catchments.

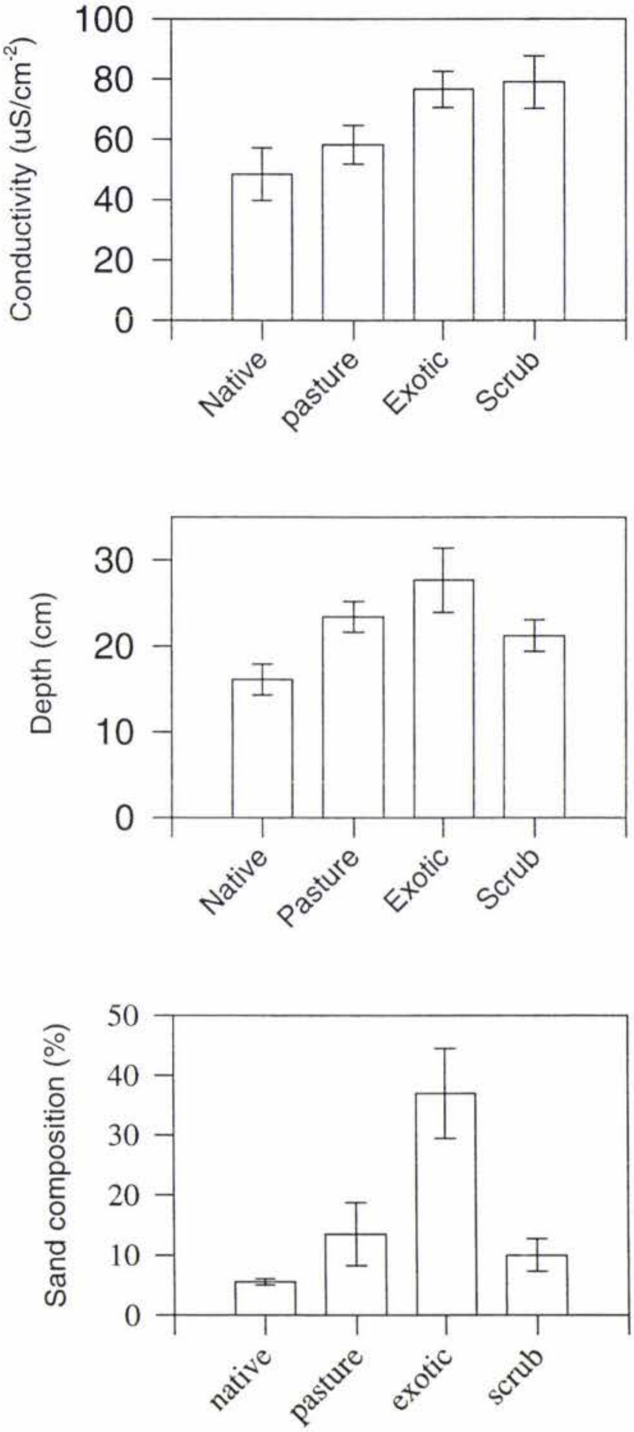

Figure 2.3. (Mean ± S.E.) (top) conductivity (middle) depth and (bottom) sand percentage measured in 35 streams around Lake Taupo between January and March 1997, which drain native forest, hill country pasture, exotic forest and scrubland catchments.

Mean TPOM (Total particulate organic matter) ranged between 0.45 and 9.30g /AFDW 0.1m^2 for Waihi (Nf4) and Hingapo (Ef4) streams respectively (Fig. 2.4). Native, scrub and pasture streams all had significantly lower TPOM than exotic forest streams, with lowest levels of TPOM in scrub streams *($F_{3,27}=4.14$, $P=0.02$).*

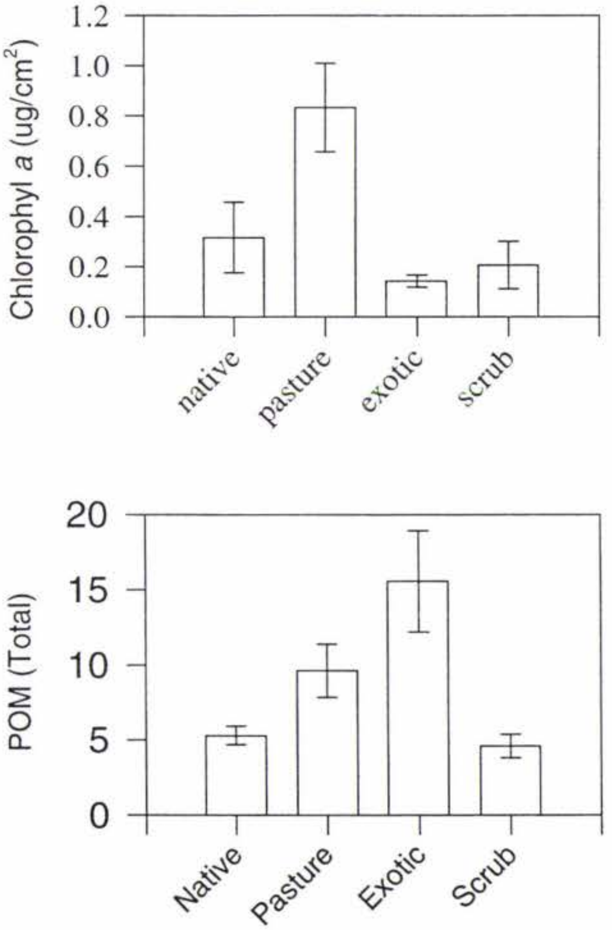

Figure 2.4. (Mean ± S.E.) (top) chlorophyll *a* and (bottom) particulate organic matter (total) measured in 35 streams around Lake Taupo between January and March 1997, which drain native forest, hill country pasture, exotic forest and scrubland catchments.

Community Structure

A total of 62 macroinvertebrate taxa were collected from the 35 study sites (Appendix 1). *Deleatidium* sp. were most abundant in native and pasture streams whereas scrub and exotic streams were dominated by Elmidae and *Zephlebia dentata* respectively (Table 2.2.). The total number of taxa collected from any site ranged between 32 in Otupoto stream (Pa2) and 5 in Mokauteure stream (Ef2). Overall species richness was greatest in native streams (averaging 23 taxa) and lowest in exotic streams (averaging 15 taxa) ($F_{3,34}=2.97$, $P=0.04$) (Fig. 2.5.). Relative abundance (Fig. 2.6.) of Plecoptera was highest in native streams ($F_{3,34}=3.97$, $P=0.02$) whereas scrub streams contained a significantly higher abundance of Coleoptera ($F_{3,34}=4.64$, $P=0.01$). Collectors were highest in pasture and scrub streams ($F_{3,34}=3.82$, $P=0.02$).

Table 2.2. The five most common macroinvertebrate taxa collected in each of native forest,
exotic forest, hill country pasture and scrub streams between January and March 1997.

Native	Pasture
Deleatidium spp.	*Deleatidium* spp.
Colorburiscus humeralis	Elmidae
Chironomidae	Chironomidae
Elmidae	*Pycnocentrodes aureola*
Aoteapsyche colonica	Oligocheata
Exotic	**Scrub**
Zephlebia dentata	Elmidae
Elmidae	*Deleatidium* spp.
Deleatidium spp.	Chironomidae
Potamopyrgus antipodarum	*Colorburiscus humeralis*
Zelollessica spp	*Eriopterini*

Margalefs index was highest in native streams and lowest in exotic streams with pasture and scrub streams intermediate *($F_{3,34}$=9.68,P<0.001)*. The total number of invertebrates collected (4 replicates combined) ranged between 15 and 1103. Scrub and pasture streams had the highest numbers of individuals with native and exotic streams the lowest *($F_{3,34}$=2.68, P=0.04)* (Fig.2.5.).

MCI scores (Fig. 2.7.) were significantly higher in native streams (mean 127) than those in pasture (mean 110), exotic (mean 108) and scrub (mean 105) *($F_{3,34}$=3.31, P=0.02)*. The QMCI (Fig. 2.7.) indicated a similar trend, although scrub streams scores were higher than pasture and exotic sites the difference was not significant *($F_{3,34}$=3.86,P=0.06)*. The percentage of Ephemeroptera, Plecoptera and Trichoptera (EPT%) (Fig.2.7.) was highest in native streams (ranging from 41 to 74%) and lowest in exotic streams (ranging from 29 to 65%). Pasture and scrub were intermediate (ranging from 22 to75% and 46 to 63%) respectively.

Ordination

Distinct communities between landuse type are not apparent in a plot of DECORANA scores (Fig. 2.8.). However, axis one generally grades sites from higher current velocity to lower current velocity, whereas axis two grades sites from forested sites (low on axis two) to open canopy sites (high on axis two). The only exceptions were the native Waihi stream (Nf2) and the exotic Mangamutu stream (Ef8) which were located high on axis two. Axis one accounted for 15.8%, and axis two 9.5% of the variation in ordination scores. Increasing axis one scores were indicative of sites with higher pollution tolerant taxa. *Physa* and Oligochaeta were both correlated with high scores on axis one whereas *Deleatidium* spp. and *Archichauliodes diversus* were common among sites that scored low on axis one. Axis one correlated with a number of site characteristics (Table 2.3.1.). Communities to the right of axis 1 characteristic of sites with narrow channels, high to medium levels of sand substrate, low dissolved oxygen and low current velocities.

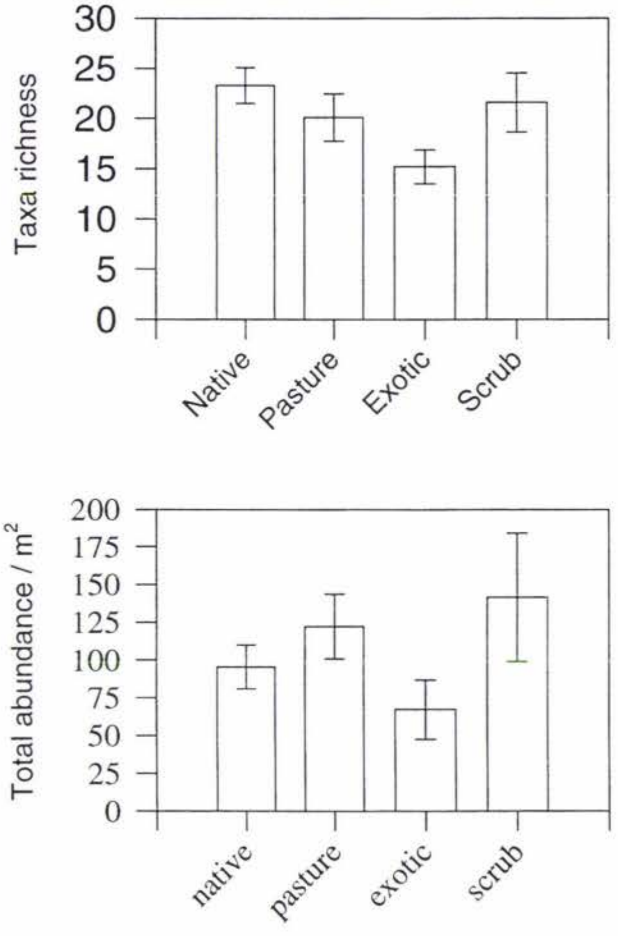

Figure 2.5. (Mean ± S.E.) (top) taxa richness and (bottom) total abundance measured in 35 streams around Lake Taupo between January and March 1997, which drain native forest, hill country pasture, exotic forest and scrubland catchments.

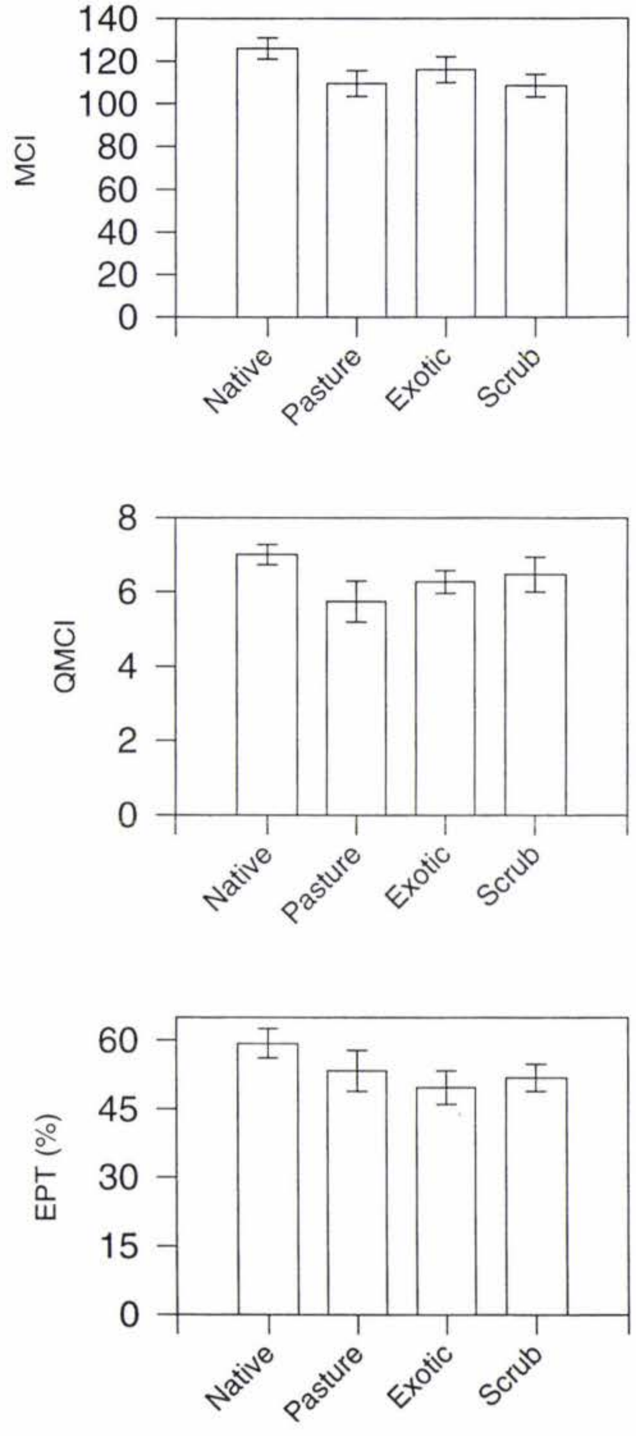

Figure 2.7. (Mean ± S.E.) (top) MCI (middle) QMCI and (bottom) EPT percentage measured in 35 streams around Lake Taupo between January and March 1997, which drain native forest, hill country pasture, exotic forest and scrubland catchments

The MCI, QMCI and EPT were all negatively correlated with axis one. Increasing axis two scores were indicative of sites with higher periphyton biomass. *Pycnocentrodes aureola* and Chironomid spp were strongly associated with these high axis two scores whereas *Stenoperla prasina*, *Austroperla cyrene* and *Colorburiscus humeralis* commonly occurred at sites that scored low on axis two. MCI, QMCI and altitude were all negatively correlated to axis two (Table 2.3.2.).

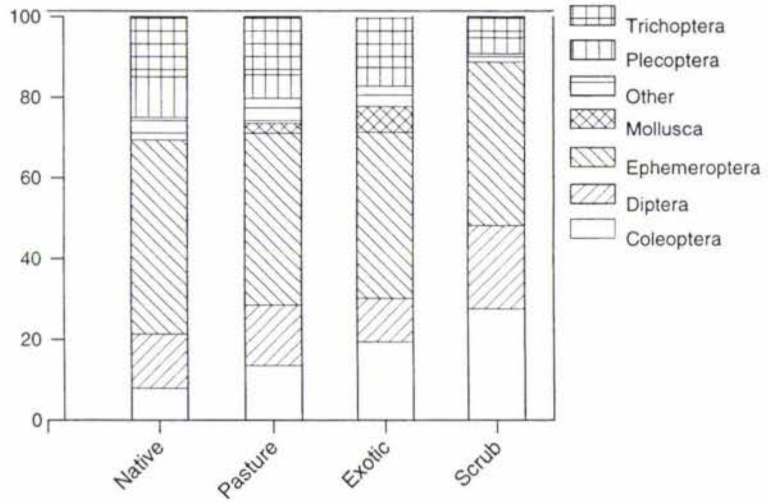

Figure 2.6. Mean relative abundance of individuals from seven taxonomic groups collected from 35 streams around Lake Taupo between January and March 1997, which drain native forest, hill country pasture, exotic forest and scrubland catchments.

Table 2.3.1. Taxa correlated (* = $P<0.05$) with Detrended Correspondence Analysis axes for macroinvertebrate communities collected from 35 streams in native forest, hill country pasture, exotic forest and scrub, sampled between January and March 1997.

Taxon	AXIS 1	AXIS 2	AXIS 3
Amphipoda	0.39 *	0.34 *	0.43 *
Aphrophila neozelandica	-0.27	0.52 *	-0.23
Archichauliodes diversus	-0.44 *	-0.26	-0.16
Austroperla cyrene	-0.14	-0.52 *	-0.08
Certopogonidae	-0.03	0.42 *	-0.26
Chironomid sp. 1	-0.18	0.45 *	-0.26
Chironomid sp. 2	-0.16	0.45 *	-0.37 *
Chironomid sp. 3	-0.32 *	0.54 *	-0.14
Chironomid sp. 4	-0.03	0.53 *	-0.29
Chironomid sp. 5	0.29	0.35 *	0.32 *
Colorburiscus humeralis	-0.30	-0.53 *	-0.03
Deleatidium spp.	-0.68 *	-0.26	0.01
Elmidae	-0.52 *	-0.79 *	-0.09
Empididae	-0.04	0.36 *	-0.15
Eriopterini	-0.12	-0.66 *	0.06
Hydrobiosis parumbripennis	0.01	0.12	-0.56 *
Hydrometridae	-0.02	0.33 *	-0.09
Hydrophilidae	0.24	0.12	0.57 *
Neocurupira hudsoni	-0.13	0.38 *	-0.12
Neurochorema confusum	-0.34 *	0.16	-0.19
Oeconesus maori	0.32 *	-0.10	0.41 *
Oligocheate	0.75 *	-0.18	-0.46 *
Olinga feredayi	-0.40 *	0.21	-0.18
Paralimnophila skusei	0.38 *	0.06	0.41 *
Physa	0.64 *	0.00	-0.37 *
Potamopyrgus antipodarum	0.09	0.41 *	0.26

Table 2.3.2. Variables correlated (* = $P<0.05$) with Detrended Corresponence Analysis axes for macroinvertebrate communities collected from 35 streams in native forest, hill country pasture, exotic forest and scrub, sampled between January and March 1997.

Parameter	AXIS 1	AXIS 2	AXIS 3
Dissolved oxygen	-0.48 *	0.11	-0.07
Velocity	-0.53 *	0.01	-0.19
Width	-0.44 *	0.20	-0.13
Altitude	-0.04	-0.35 *	0.00
MCI	-0.38 *	-0.47 *	0.38 *
QMCI	-0.54 *	-0.43 *	0.42 *
EPT%	-0.65 *	0.22	0.23

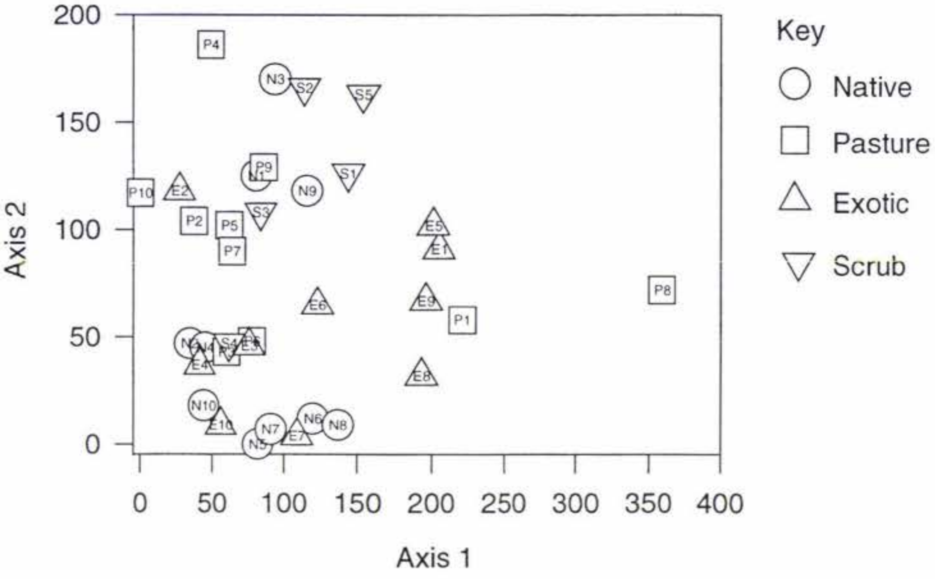

Figure 2.8. Axis 1 of a Detrended Correspondence Analysis (DECORANA) as a function of axis 2 for macroinvertebrate communities collected from 35 streams around Lake Taupo between January and March 1997, which drain native forest, hill country pasture, exotic forest and scrubland catchments.

DISCUSSION

Elevated light and temperature resulting from the removal of riparian vegetation have been known to cause increases in periphyton biomass in streams (Biggs, 1990; Holopainen, 1992; Quinn, 1994). The finding of higher algal biomass in open canopy streams than closed canopy streams in this study is consistent with this (Quinn & Hickey, 1990b; Quinn *et al.*, 1997). However, in comparison to scrub streams which also had open canopies, agriculturally developed catchments had 3-fold higher algal epilithon. Overhead cover and riparian vegetation had both been significantly reduced in pasture and scrub catchments, yet only pasture streams had highly elevated periphyton levels. This may be explained by increased nutrient levels from fertiliser and soil runoff at pasture sites (Minshall, 1984; Quinn *et al.*, 1992, 1994; Quinn & Hickey, 1990b).

Increased periphyton biomass in open streams may be responsible for the higher macroinvertebrate abundance in the pasture and scrub streams (Biggs, 1990; Jowett & Richardson, 1990; Quinn & Hickey, 1990b; Quinn *et al.*, 1994; Friberg, 1997). Decreased invertebrate abundance in exotic sites are likely to be a result of reduced primary production (Quinn *et al.*, 1992), a reduction in food quality (Stumn & Morgan, 1981; Gregory, 1983; Sweeney & Vannote, 1986) low substrate stability and high percentages of silt and sand (Quinn & Hickey, 1990b).

Species richness and diversity declined as overall stream stability decreased (Death & Winterbourn, 1995). Streambeds in native forest streams were the most stable, whereas exotic sites had the least stable substrates because they were dominated by sand. Previous studies (Collier, 1989; Graynoth, 1979; Harding & Winterbourn, 1995) indicate that benthic invertebrate communities comparable to those in native forest are re-established following planting with exotic conifers. However, this did not occur at my Taupo sites where exotic and native stream faunas were considerably different. Some physical habitat differences between exotic and native streams examined during this study are evident. The general lack of an understory in exotic catchments and the fine nature of volcanic soils in this study area may have resulted in higher sediment inputs in exotic streams compared to streams draining undisturbed catchments with lush understory growth.

The number of taxa belonging to the more sensitive ephemeropteran, plecopteran, and trichopteran insect orders were lower in exotic streams while molluscs and coleopterans were more abundant (Waters, 1995). Decreases in the total number of individuals and taxa in exotic streams may be a result of suffocation of the stream bed by high silt and sand levels, filling the interstitial spaces and reducing oxygen availability to subsurface organisms (Ryder, 1991; Schalchli, 1992; Wagner & La Perriere, 1985; Waters, 1995). Taxa abundant at exotic sites were similar to those found to be affected by increased sedimentation in experiments conducted by Ryder (1989). *Deleatidium* sp. were reduced in in experiments conducted by Ryder (1989). *Deleatidium* sp. were reduced in abundance in exotic sites, whereas *Potamopyrgus antipodarum* and Oligocheata were abundant. These shifts towards communities dominated by taxa that are less sensitive to environmental extremes at the exotic sites was also reflected in the MCI and QMCI values, which were lower at the exotic sites. This would indicate that exotic forestry on the Central Volcanic Plateau is more detrimental to water quality than low to moderate intensity sheep and beef farming in the area.

In the present study, exotic stream invertebrate community composition was different from the other landuse types. This finding contrasts with the results of others (Friberg *et al.*, 1997; Harding & Winterbourn, 1995; Quinn & Hickey, 1990; Quinn *et al.*, 1994; Quinn *et al.*, 1997) which found exotic and native forest communities to be similar and contrast with pasture communities. However, in this study exotic streams had low bed stability and high levels of sand, it is likely that this may have caused the difference in both exotic stream communities and native stream communities. Pasture streams appear to be less effected by landuse as a result of less intensive farming at the studied sites than in other studies. In conclusion, the effects of landuse change will not be consistent throughout New Zealand, but will be affected by different land conditions, especially geology.

REFERENCES

Allan, K.R. 1959. Effect of land development on stream bottom faunas. *Proceedings of the New Zealand Ecological Society* **7:** 20-21.

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

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

Clifford, H.T. and W. Stephenson 1975. An Introduction to Numerical Classification. New York: Academic press.

Collier, K. J., M. J. Winterbourn and K. J. Jackson 1989. Impacts of wetland afforestation on the distribution of benthic invertebrates in acid streams of Westland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **23**: 479-490.

Corkum, L.D. 1990. Intrabiome distributional patterns of lotic macroinvertebrate assemblages. *Canadian Journal of Fisheries and Aquatic Sciences* **47:** 2147-2157.

Cowie, B. 1985. An analysis of changes in the invertebrate community along a southern New Zealand montane stream. *Hydrobiologia* **120:** 35-46.

Cowley, D.R. 1978. Studies on the larvae of New Zealand Trichoptera. *New Zealand Journal of Zoology* **5:** 639-750.

Cummins, K.W., G. W. Minshall, J. R. Sedall, C. E. Cushing and R. C. Petersen 1984. Stream Ecosystem Theory. *Verhandlungen der Internationalen Vereinigung für theoretisch und angewandte Limnologie* **22:** 1818-1827.

Dance, K.W. and H.B.N. Hynes 1980. Some effects of agricultural land-use on stream insect communities. *Environmental Pollution (Series A)* **22:** 19-28.

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

Fahey, B.D. and L.K. Rowe 1992. Land use impacts. In: Mosley M.P., editor. Waters of New Zealand. Wellington: New Zealand Hydrological Society Inc. p 265-284.

Friberg, N. and M.J. Winterbourn 1997. Effects of native and exotic forest on stream biota in New Zealand: a colonisation study. *Marine and Freshwater Research* **48:** 267-275.

Friberg, N., M.J. Winterbourn, K. A Shearer. and S. E. Larsen 1997. Benthic communities of forest streams in the South Island, New Zealand: effects of forest type and location. *Archiv für Hydrobiologie* **138(3):** 289-306.

Graham, A.A., D. J. McCaughan and F. S. McKee 1988. Measurement of surface area of stones. *Hydrobiologia* **157:** 85-87.

Graynoth, E. 1979. Effects of logging on stream environments and faunas in Nelson. *New Zealand Journal of Marine and Freshwater Research* **13**: 79-109

Gregory, S.V. 1983. Plant-herbivore interactions in stream ecosystems. In: Barnes J. R. and Minshall, G. W., editors. Stream ecology, application and testing of general ecological theory. New York: Phlenum Press. p 157-190.

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.

Holopainen, A.L. and P. Huttunen 1992. Effects of forest clear-cutting and soil disturbance on the biology of small forest brooks. *Hydrobiologia* **243/244**: 457-464

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

Lenat, D.R. 1984. Agriculture and stream water quality: a biological evaluation of erosion control practices. *Environmental Management* **8:** 333-344.

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:** 222-233.

McCune, B. and Mefford, M.J. 1995. PC-ORD Multivariate Analysis of Ecological Data, Version 2.0. Oregon: MjM Software Design.

McFarlane, A.G. 1951. Caddisfly larvae (Trichoptera) of the family Rhyacophilidae. *Records of the Canterbury Museum* **5:** 267-289.

Minshall, F. W. 1984. Aquatic insect-substratum relationships. In: Resh, V. H.; Rosenberg, D. M. ed. The ecology of aquatic insects, New York, Praeger, pp. 358-400.

Moss, B. 1967a. A spectrophotometric method for the estimation of percentage degradation of chlorophyll's to pheo-pigments in extracts of algae. *Limnology and Oceanography* **12:** 335-340.

Moss, B. 1967b. A note on the estimation of chlorophyll *a* in freshwater algal communities. *Limnology and Oceanography* **12:** 340-342.

Pfankuch, D. J. 1975. Stream reach inventory and channel stability evaluation. U.S. Department of Agriculture Forest Service/Northern Region.

Prat, N. and J.V. Ward 1994. The tamed river. In: Margalefs, R., editor. Limnology Now: a Paradigm of Planetary Problems. Amsterdam: Elsevier Science. p 219-236.

Quinn, J.M. and C.W. Hickey 1990a. Characterisation and classification of benthic invertebrate communities in 88 New Zealand rivers in relation to environmental factors. *New Zealand Journal of Marine and Freshwater Research* **24:** 387-409.

Quinn, J.M. and C.W. Hickey 1990b. Magnitude of effects of substrate particle size, recent flooding and catchment development on benthic invertebrates in 88 New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research* **24:** 411-427.

Quinn, J.M., A.B. Cooper, R.J. Davies-Colley, J.C.Rutherford and R.B. Williamson 1994. Land-use effects on New Zealand hill country streams and implications for riparian management. In: Link, G.L. and Naiman, R.J., editors. Proceedings of the International Workshop on the Ecology and Management of Aquatic-Terrestrial Ecotones. Seattle: University of Washington.

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:** 579-597.

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

Reed, J.L., I.C. Campbell and P.C.E. Bailey1994. The relationship between invertebrate assemblages and available food at forest and pasture sites in three south-eastern Australian streams. *Freshwater Biology* **32:** 641-650.

Richards, C. and G. Host 1994. Examining land-use influences on stream habitats and macroinvertebrates: a GIS approach. *Water Resources Bulletin* **30:** 729-738.

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. Host 1996. Landscape scale influences on stream habitats and biota. *Canadian Journal of Fisheries and Aquatic Sciences* **53:** 295-311.

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

Ryder, G. I. 1989. Experimental Studies on the Effects of Fine Sediments on Lotic Invertebrates. Unpublished Ph.D. thesis, University of Otago, New Zealand.

SAS 1989. SAS User's Guide: Statistics. Version 6.1 SAS Institute Incorporated, Cary, North Carolina, USA.

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

Smith, M.J., A.B. Cooper and J.M. Quinn 1993. Land-water interactions research at Whatawhata - the start. Hamilton, New Zealand: National Institute of Water and Atmospheric Research, Ecosystems. Publication 4.

Stark, J.D. 1985. A macroinvertebrate community index of water quality for stony streams. *Water and Soil Miscellaneous Publication* **87:** 53p.

Stark, J.D. 1993. Performance of the Macroinvertebrate Community Index: effects of sampling method, sample replication, water depth, current velocity, and substratum on index values. *New Zealand Journal of Marine and Freshwater Research* **21:** 463-478.

Stumn, W. and Morgan, J.J. 1981. Aquatic Chemistry. New York: J. Wiley and Sons.

Sweeney, B.W. and R.L. Vanote 1986. Growth and production of a stream stonefly: influences of diet and temperature. *Ecology* **67:** 1396-1410.

Towns, D.R. and W.L. Peters 1996. Leptophlebiidae (Insecta: Ephemeroptera*).* *Fauna of New Zealand* **36:** 37-50.

Townsend, C.R. 1989. The patch dynamics concept of stream community ecology. *Journal of the North American Benthological Society* **8:** 36-50.

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

Ventura, M. and D. Harper 1996. The impacts of acid precipitation mediated by geology and forestry upon upland stream invertebrate communities. *Archive für_Hydrobiologia* **138(2):** 161-173.

Wagener, S.M. and J.D. LaPerriere 1985. Effects of placer mining on invertebrate communities of interior Alaska streams. *Freshwater Biology* **4**: 208-214.

Waters, T.F. 1995. Sediment in Streams: Sources, Biological Effects, and Control. American Fisheries Society Monograph 7.

Williamson, R.B., R.K. Smith and J.M. Quinn 1992. Effects of riparian grazing and channelisation on streams in Southland, New Zealand. 1. Channel form and stability. *New Zealand Journal of Marine and Freshwater Research* **26:** 241-258.

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.

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

Winterbourn, M.J. and Gregson, K.L.W. 1989. Guide to the Aquatic Insects of New Zealand. Bulletin of the Entomological Society of New Zealand 9, 96 p.

CHAPTER THREE

The effects of landuse on invertebrate and microbial breakdown of leaf litter

ABSTRACT

Leaf litter decomposition in leaf bags and leaf tubes and invertebrate colonisation of leaf litter bags were investigated in 12 streams of differing landuse (native, pasture, exotic and scrub) around lake Taupo, New Zealand. Mesh tubes (0.2mm) and mesh bags (3mm) containing dried Rangiora (*Brachyglottis repanda*) leaves were immersed into streams draining native beech and broadleaf/podocarp forests, hill country pasture, exotic *Pinus radiata* and scrubland. Leaf litter bags and tubes were removed from streams after 14, 28 and 42 days to measure the percentage of weight lost from each leaf bag and leaf tube and to assess invertebrate colonisation of the leaf bags. Decomposition of leaf litter was more rapid in tubes removed from streams with open canopies than closed canopies and appeared to be influenced by increased temperatures and nutrient inputs at these sites. Decomposition rates of leaf litter were faster in exotic leaf bags, pasture and native leaf bags were intermediate and scrub stream leaf bags were slowest. Chironomidae, Oligocheata and *Potamopyrgus antipodarum* were the most abundant leaf bag colonists. Leaf bags in exotic and native streams supported the highest invertebrate densities, but shredders were most abundant in leaf bags from native streams. All landuse types showed distinct differences in macroinvertebrate community composition in contrast to benthic communities which were quite similar. It appears then that landuse does influence benthic invertebrate communities in leaf bags which in turn influence the decomposition rate of leaf litter within leaf bags.

Keywords: abundance, colonisation, decomposition, exotic forest, hill country pasture, landuse, leaf litter bags, leaf litter tubes, macroinvertebrate community composition, native forest, scrubland, weight loss.

INTRODUCTION

Terrestrial vegetation that enters the stream channel provides an important source of detrital food and habitat for many stream invertebrates (Kaushik & Hynes, 1971; Peterson & Cummins, 1974; Anderson & Sedell, 1979; Rounick & Winterbourn, 1983a).

Processing of leaf litter is influenced by a number of factors including: water chemistry; temperature and current velocity, leaf species, microbial activity and invertebrate feeding (Suberkropp, Klug & Cummins, 1975; Davis & Winterbourn, 1977; Meyer, 1980, Irons *et al.*, 1994). Typically, land development causes a number of changes to streams that may influence leaf processing including increased nutrient inputs, flow variability, bank erosion and turbidity along with modifications to the temperature regime, bed structure and organic inputs (Wilcock, 1986; Quinn *et al.*, 1992). Catchment disturbances such as agriculture and plantation forestry can greatly influence the timing, quantity and quality of allocthonous inputs to streams (Sweeney, 1993; Tuchman & King, 1993; Delong & Brusven, 1994) and therefore significantly effect stream communities (Newbold *et al.*, 1980; Webster & Waide, 1982). These effects include shifts in invertebrate functional feeding group distributions because of increases in autochthonous primary production and decreases in allochthonous leaf detritus (Gurtz & Wallace, 1984; Wallace & Gurtz, 1986).

Leaf processing is a complex interaction of physical, chemical, microbial and animal processes and therefore measurements of breakdown rates may be useful for assessing the effects of anthropogenic disturbance, such as land development, on stream systems (Webster & Benfield, 1986) as it is affected by a wide range of variables that themselves can be influenced by disturbance to an ecosystem (Young *et al.*, 1994). Although the influence of many such changes on physical characteristics of streams are well documented (Harding & Winterbourn, 1995; Quinn *et al.*, 1994; Quinn *et al.*, 1997) effects of land development on animal communities and biotic processes are less well understood (Whiles & Wallace, 1997).

Previous studies of leaf litter processing have examined the dramatic changes to catchments such as logging (Webster & Waide, 1982; Meyer & Johnson, 1983; Griffith & Perry, 1991), channelisation (Gelroth & Marzolf, 1978), highway construction (Stout & Coburn, 1989) or differences between agricultural (Young *et al.*, 1994) and forested stream reaches (Bird & Kaushik, 1992). My approach is to investigate leaf litter decomposition of several sites from four different landuse activities.

The aims of the present study are to examine the effect of land management on processing of leaf litter in 0.2mm mesh tubes and 3mm mesh bags and to investigate invertebrate colonisation of leaf litter bags. Invertebrate and microbial decomposition of leaf litter in the different landuse types will be compared. Thus, invertebrate communities inhabiting leaf bags and decomposition of Rangirora (*Brachyglottis repanda*) litter in leaf bags and leaf tubes were examined in 12 streams draining native forest, exotic forestry, scrubland and pastoral farming land use types.

STUDY SITES

The 12 study sites were second and third order streams which were situated around Lake Taupo, North Island, New Zealand (176° 05'E, 38° 42'S). Study sites ranged in altitude between 320m and 750m a.s.l. (Fig. 3.1.).

Landuse categories were defined as sites wherthe vegetation characteristic of that landuse covers >80% of the catchment. They included native beech and broadleaf/podocarp forests, exotic plantation forest, scrubland, and hill country pasture. Native beech (Kaimanawa forest) and broadleaf/podocarp (Pureora forest) (Nf 1-Nf 3) had extensive forest vegetation down to the stream channel. Exotic *Pinus radiata* plantation forests (Pf 1-Pf 3) originate back approximately 20 years and had riparian vegetation consisting mainly of scrub, ferns and blackberry. Scrubland (Sc 1-Sc 3) had riparian growth that was often well developed and consisted of various flaxes, gorse, Kanuka and willows. Agricultural activity in pasture catchments (Pa 1-Pa 3) consisted of moderate to low intensity sheep/beef grazing. Riparian vegetation at these pasture streams usually consisted of exotic pasture species with stock access to the stream channel predominant.

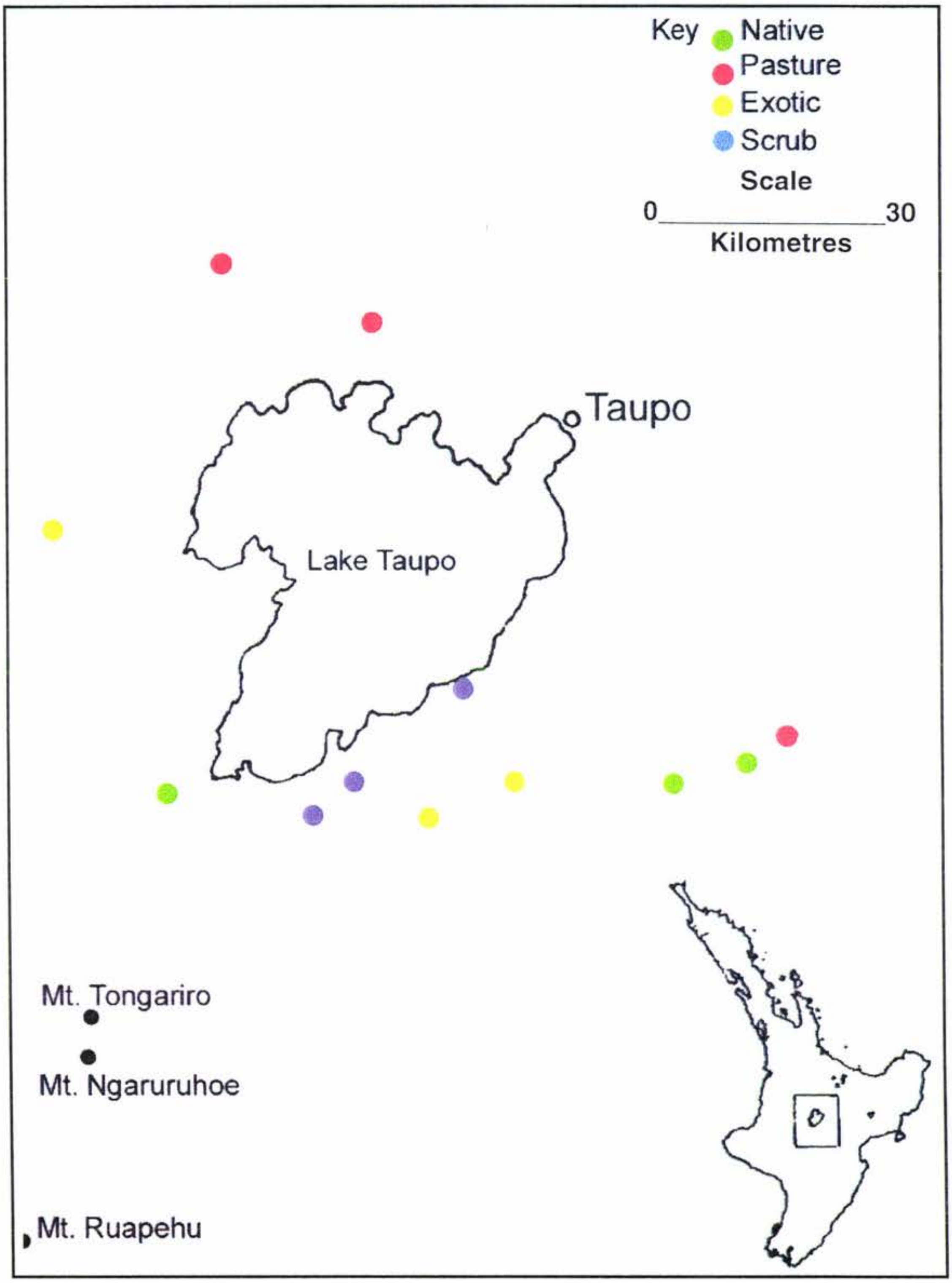

Figure 3.1. Location of the 12 streams within four landuse types (native (beech and broadleaf/podocarp) forests, hill country pasture, exotic *Pinus radiata* forest and scrubland) which were studied between December 1997 and January 1998, around Lake Taupo, North Island, New Zealand.

Plate 3.1. A native forest (left) and a pasture (right) study site where leaf bag and leaf tube decomposition was assessed between December 1997 and January 1998.

MATERIALS AND METHODS

Fresh leaves of Rangiora were placed into pre-weighed bags (heat-sealed 12x15 cm bags constructed from 3mm plastic mesh) before drying at 60°C to constant weight. After drying, the bags were re-weighed and leaf weight determined to the nearest milligram. These 3mm mesh bags allowed invertebrate and microbial access to the leaves. To exclude invertebrates and evaluate microbial and non-invertebrate weight losses, leaf tubes were constructed from 9cm long, 40mm diameter P.V.C tubing sealed at each end with 0.2mm mesh and fastened with electrical cable ties.

On 1, December 1997, 12 bags and 12 tubes were placed randomly in riffles at each site. The leaf bags were tethered with 9.1kg monofilament fishing line to 25cm metal pegs that were driven into the streambed. Tubes were attached with the same dimensioned pegs, which were threaded through the tubes and into the streambed. The leaf bags and tubes were positioned against rocks to simulate natural accumulations.

Four leaf bags and four leaf tubes were randomly recovered from each stream after two weeks (14 days) (14, December), four weeks (28 days) (28, December) and six weeks (42 days) (11. January) by collection into a kick net (250μm mesh). Samples were preserved in 5% formalin for later laboratory analysis.

A range of physicochemical parameters were measured at each site visit. Temperature was measured with an Orion 122 temperature meter. Four water velocity readings were taken midstream with a velocity head rod at each site, along with four width and four depth measurements. The overhead cover was visually assessed. pH was measured with an Orion 250A pH meter. Suspended solids in a 500ml water sample were measured by filtering through a preweighed Whatman GF/C filter (pore size=0.7μm) and dried overnight at 105°C, before subtracting the filter weight and then re-weighing.

Invertebrates in each leaf bag were sorted and enumerated by eye in a white tray. Taxa were then identified under a 40x binocular microscope to species (except Oligochaete and Chironomidae) using Winterbourn and Gregson (1989). Rinsed leaves from leaf bags and leaf tubes were re-dried at 60^0C until constant weight and then re-weighed to the nearest milligram.

Data were log 10 (x+1) transformed prior to analysis where appropriate to remove heterogeneity of variance. Differences among physicochemical measures, landuse types and decomposition rates were analysed with two-way analysis of variance (ANOVA) using the SAS statistical package (SAS, 1989), and Duncan multiple range means tests with an alpha level of (P=0.05). To examine overall trends in community structure, Detrended Correspondence Analysis (DECORANA) was performed using the PC-ORD statistical package (McCune & Mefford, 1995).

RESULTS

Stream characteristics

All twelve streams were physically similar. Water temperatures in these narrow to medium width (mean range 1.5 - 4m) streams ranged from 7.4 to 18.4°C and had slow to medium flow rates (0.4 - 0.8m/s^{-1}). Mean depth ranged from 12 to 31cm (Table 3.1.). pH values were circumneutral and ranged from 6.8 to 8.8. Overhead cover was significantly higher in forested catchments than scrub and pasture sites ($F_{3,11}=8.79$, $P=0.01$). Substrate composition in streams of native catchments was highly diverse and had high stability, whereas at exotic and pasture streams substrate consisted largely of sand and therefore were less stable.

Weight loss (Leaf litter tubes)

After 14 days, scrub leaf tubes had decomposed 7% faster than native leaf tubes ($F_{3,35}=7.41$, $P<0.001$) (Fig. 3.2.). After 28 days the open canopy streams of pasture and scrub had lost more leaf litter weight than the closed canopy streams of native and exotic leaf tubes ($F_{3,35}=5.99$, $P=0.002$). The leaf tubes in the open canopy streams after 42 days had decomposed 7-9% faster than leaf tubes from closed canopy streams ($F_{3,35}=9.04$, $P<0.001$).

Table 3.1. Physicochemical characteristics recorded from twelve streams draining native (beech and broadleaf/podocarp) forests, hill country pasture, exotic (*Pinus radiata)* forest and scrubland catchments between December 1997 and January 1998.

Sites	Map (Ref) (1:250 000)	Mean width (m)	Mean depth (cm)	Velocity (ms-1)	Temperature (oC)	pH	Cover (%)	Suspended solids (mg/L)	Altitude
Native									
Te Arero	U19 456981	2.8 (2.0-4.0)	18.25 (14-24)	0.81	9.9 (7.4-12.2)	8.1	100	29.19	780
Pirua	U19 465860	2.5 (1.7-3.0)	19.25 (14-25)	0.61	10.4 (9.1-13.4)	7.4	100	34.56	720
Waihi	T19 472463	2.7 (2.5-3.2)	20 (17-27)	0.57	13.3 (11.4-16.3)	8.8	90	18.23	480
Pasture									
Taharua	U19 496945	3.1 (2.0-4.0)	24 (13-31)	0.44	12.1 (10.4-16.1)	8	0	2.96	650
Waikino	T17 955566	2.5 (1.5-4.0)	20 (18-22)	0.61	14.4 (12.4-18.4)	6.9	0	62.35	360
Potungutungu	T17 940583	2.6 (2.0-3.0)	21.75 (18-26)	0.83	12.8 (9.8-14.2)	8	0	37.68	370
Exotic									
Mokauteure	T17 966674	3.1 (2.5-3.5)	22.5 (18-26)	0.54	13.1 (11.7-15.4)	8.3	60	8.69	470
Waitapu	U19 465786	2.2 (1.7-2.5)	20.5 (16-23)	0.42	10.5 (8.1-12.9)	8.1	90	41.39	540
Tangihamutu	U18 623875	3.8 (3.5-4.0)	22.75 (12-31)	0.68	11.1 (9.4-14.8)	7.9	20	64.06	690
Scrub									
Waiotaka	T19 432562	3.6 (2.8-3.8)	21.25 (16-27)	0.53	14.7 (12.6-17.4)	6.8	20	41.15	370
Waimarino	T19 461593	2.6 (2.3-3.0)	22.25 (12-24)	0.84	12.8 (11.4-14.1)	8.7	60	37.14	370
Waipehi	T18 541683	3.7 (2.8-4.0)	17 (12-22)	0.52	13.4 (12.9-14.0)	7.1	20	28.45	370

Weight loss (Leaf litter bags)

Weight of leaf litter bags declined in an approximately linear manner (Fig. 3.2.) and after 14 days leaf bags from exotic and native catchments had decomposed 6-11% faster than leaf bags from scrub and pasture catchments ($F_{2,35}=9.99$, $P<0.001$) (Table 3.2.). After 28 days the leaf bags from exotic and native streams had 6–14% higher weight loss than leaf bags from scrub streams. Pasture leaf bags had the lowest decomposition rate after 28 days ($F_{2,35}=13.77$, $P<0.001$). Exotic leaf bags lost 18% more leaf weight than scrub after 42 days. Pasture and native stream leaf litter losses were intermediate after 42 days (53% and 52% respectively).

Scrub and native leaf bag weight losses were linear with time ($F_{2,26}=68.31$, $P<0.001$) and ($F_{2,26}=44.47$, $P<0.001$) respectively. However, exotic and pasture leaf weight losses increased significantly between days 28 and 42 ($F_{2,26}=30.37$, $P<0.001$) and ($F_{2,26}=34.72$, $P<0.001$) respectively (Table 3.3.).

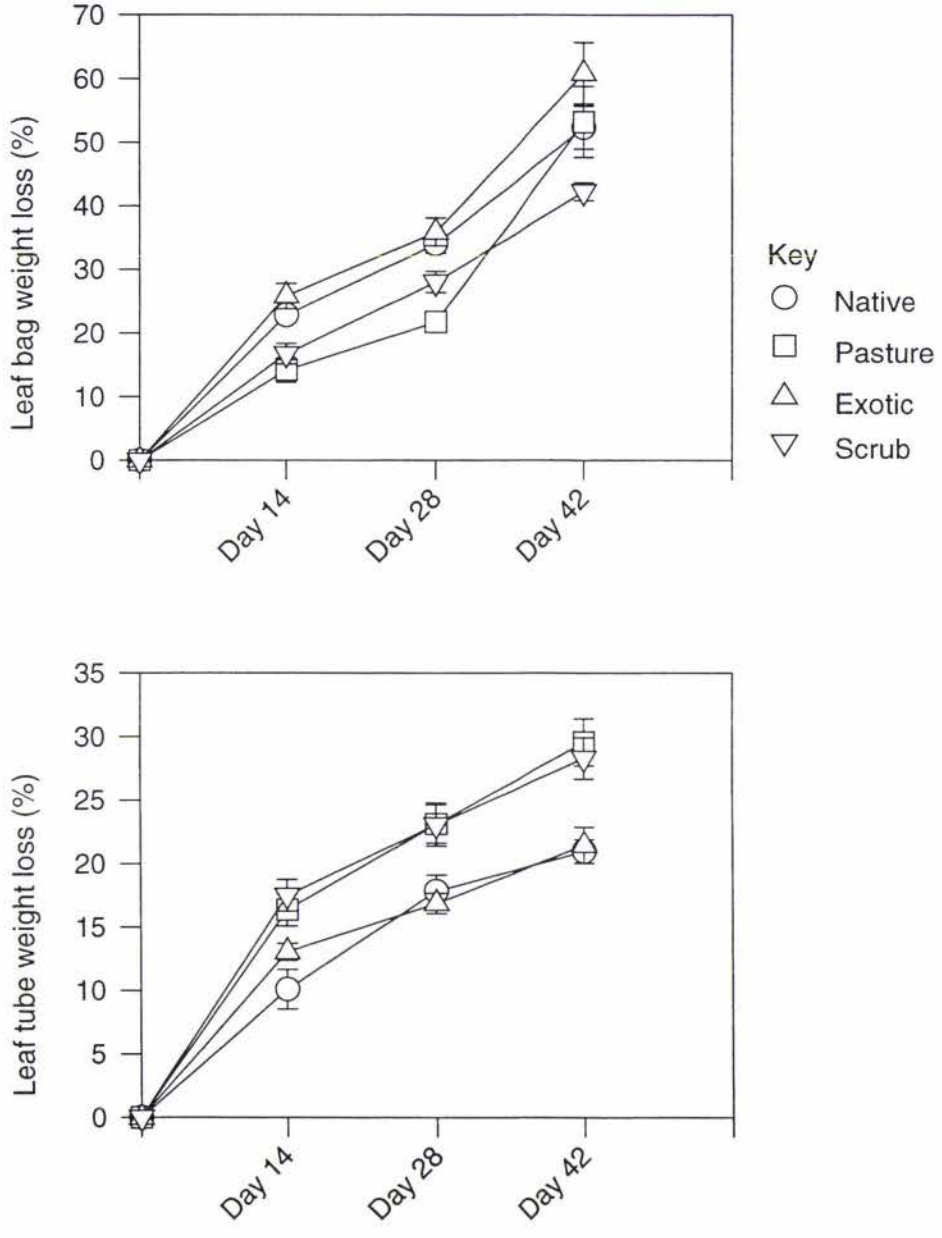

Figure 3.2. (Mean ±S.E.) (top) leaf bag (3mm mesh) percentage weight loss and (bottom) leaf tube (0.2mm mesh) percentage weight loss measured in 12 streams around Lake Taupo between December 1997 and January 1998, which drain native forest, hill country pasture, exotic forest or scrub catchments.

Table 3.2. Values showing the leaf bag characteristics and the percentage of leaf weight lost in Days 14, 28 and 42 between December 1997 and January 1998.

		Day 14				Day 28				Day 42			
Stream	Rep	Original dry leaf weight (g)	Dry leaf weight remaining (g)	Difference (g)	Weight lost (%)	Original dry leaf weight (g)	Dry leaf weight remaining (g)	Difference (g)	Weight lost (%)	Original dry leaf weight (g)	Dry leaf weight remaining (g)	Difference (g)	Weight lost (%)
Native													
Te Arero	1	9.70	7.84	1.86	19.18	10.22	7.47	2.75	26.91	12.41	7.57	4.84	39.00
Te Arero	2	10.27	7.92	2.35	22.88	12.10	7.56	4.54	37.52	13.17	6.62	6.55	49.73
Te Arero	3	12.34	9.77	2.57	20.83	11.63	7.40	4.23	36.37	8.85	4.30	4.55	51.41
Pirua	1	10.19	7.54	2.65	26.01	11.34	7.01	4.33	38.18	10.78	3.50	7.28	67.53
Pirua	2	13.26	10.81	2.45	18.48	10.52	7.01	3.51	33.37	8.71	2.66	6.05	69.46
Pirua	3	13.96	11.05	2.91	20.85	13.66	9.91	3.75	27.45	10.32	5.17	5.15	49.90
Waihi	1	11.37	8.71	2.66	23.39	10.72	7.16	3.56	33.21	13.13	6.12	7.01	53.39
Waihi	2	8.36	6.26	2.10	25.12	10.61	7.06	3.55	33.46	12.93	6.87	6.06	46.87
Waihi	3	11.94	8.50	3.44	28.81	11.97	7.22	4.75	39.68	12.54	7.07	5.47	43.62
Pasture													
Taharua	1	11.37	10.56	0.81	7.12	9.08	7.16	1.92	21.15	9.80	4.18	5.62	57.35
Taharua	2	7.92	7.54	0.38	4.80	10.48	8.72	1.76	16.79	11.36	5.47	5.89	51.85
Taharua	3	13.46	10.79	2.67	19.84	9.36	8.32	1.84	19.66	9.23	4.03	5.20	56.34
Waikino	1	13.52	11.42	2.10	15.53	8.20	5.92	2.28	27.80	8.38	1.48	6.90	82.34
Waikino	2	9.14	7.19	1.95	21.33	10.38	8.09	2.29	22.06	10.56	2.74	7.82	74.05
Waikino	3	12.60	10.55	2.05	16.27	9.42	6.89	2.53	26.86	11.14	6.88	4.26	38.24
Potungutungu	1	10.51	8.95	1.56	14.84	15.73	13.62	2.59	16.47	9.41	6.63	2.78	29.54
Potungutungu	2	13.87	12.48	1.39	10.02	11.08	9.01	2.07	18.68	8.00	4.57	3.43	42.88
Potungutungu	3	11.53	9.46	2.07	17.95	7.27	5.36	1.91	26.27	9.04	4.87	4.17	46.13
Exotic													
Mokauteure	1	10.14	7.38	2.76	27.22	12.09	6.90	5.19	42.93	11.04	6.16	4.88	44.20
Mokauteure	2	8.48	5.82	2.66	31.37	11.39	6.66	4.73	41.53	8.08	2.27	5.81	71.91
Mokauteure	3	8.67	6.57	2.10	24.22	11.36	7.19	4.17	36.71	12.35	5.53	6.82	55.22
Waitapu	1	10.80	9.02	1.78	16.48	10.01	6.58	3.43	34.27	9.83	3.92	5.91	60.12
Waitapu	2	8.49	6.25	2.24	26.38	8.49	6.25	2.24	26.38	8.95	5.64	3.31	36.98
Waitapu	3	10.15	7.30	2.85	28.08	8.00	4.93	3.07	38.38	9.86	3.02	6.84	69.37
Tangihamutu	1	8.24	5.26	2.98	36.17	8.33	4.72	3.61	43.34	9.74	2.70	7.04	72.28
Tangihamutu	2	9.02	6.92	2.10	23.28	11.33	8.43	2.90	25.60	8.76	3.89	4.87	55.59
Tangihamutu	3	10.40	8.44	1.96	18.85	11.32	7.51	3.81	33.66	8.40	1.51	6.89	82.02
Scrub													
Waiotaka	1	10.69	8.13	2.56	23.95	8.40	6.15	2.25	26.79	12.81	6.76	6.05	47.23
Waiotaka	2	11.90	9.97	1.93	16.22	8.64	6.56	2.08	24.07	10.72	6.02	4.70	43.84
Waiotaka	3	13.37	11.57	1.80	13.46	10.73	7.68	3.05	28.42	12.16	6.93	5.23	43.01
Waimarino	1	13.58	11.74	1.84	13.55	9.82	6.41	3.41	34.73	10.73	6.87	3.86	35.97
Waimarino	2	11.61	9.54	2.07	17.83	9.97	7.03	2.94	29.49	10.08	6.21	3.87	38.39
Waimarino	3	12.02	9.01	3.01	25.04	9.82	6.48	3.34	34.01	10.93	6.29	4.64	42.45
Waipehi	1	7.95	7.05	0.90	11.32	13.18	10.54	2.64	20.03	13.28	7.55	5.73	43.15
Waipehi	2	8.34	7.23	1.11	13.31	7.62	5.24	2.38	31.23	9.86	5.10	4.76	48.28
Waipehi	3	9.84	8.28	1.56	15.85	10.83	8.27	2.56	23.64	12.45	7.73	4.72	37.91

Colonisation of leaf bags by invertebrates

Thirty-two taxa of invertebrates were found in leaf litter bags (ranging from 7 in Potunutungu (P3) stream to 21 in Pirua (N2) and Waimarina (S2) streams) and increased linearly with time ($F_{2,23}=7.40$, $P=0.003$) (Fig. 3.3.) Of these, eight could be categorised as obligate or facultative shredders; *Austroperla cyrene, Oeconesus maori, Ptilodactylidae, Zelandobius* sp., *Zellolessica* sp., *Olinga jeanae, Hudsonema amabilis, Pycnocentria* sp. as indicated in previous studies (Winterbourn, Crowe & Rounick, 1984; Chadderton, 1988; Quinn & Hickey, 1990; Parkyn & Winterbourn, 1997).

Table 3.3. F-values, degrees of freedom and *P*-values (* = P<0.001) for two-way ANOVA's performed on the percentage of leaf bag weight lost in landuse type and in Days 14, 28 and 42 (Time) between December 1997 and January 1998.

Variables	F-value	d.f	P-value
Percentage lost v. time (Day 14)	9.99	3,35	0.0001
Percentage lost v. time (Day 28)	13.77	3,35	0.0001
Percentage lost v. time (Day 42)	3.43	3,35	0.02
Percentage lost v. native	44.47	2,26	0.0001
Percentage lost v. pasture	34.72	2,26	0.0001
Percentage lost v. exotic	30.37	2,26	0.0001
Percentage lost v. scrub	68.31	2,26	0.0001

Native and scrub leaf bags had significantly higher numbers of species on all dates, exotic less and pasture leaf bags the least *($F_{3,35}$=12.57, $P<0.001$)*. Chironomidae was the most abundant taxon in all bags on all sampling occasions followed by Oligochaeta and *Potamopyrgus antipodarum*. The total number of invertebrates (Fig. 3.3.) in each landuse increased over time, with the period between days 28 and 42 having significantly higher numbers than the first and second periods *($F_{3,35}$=8.46, $P<0.001$)*. After 42 days the highest numbers of invertebrates were found in exotic leaf bags, intermediate density in native and pasture and the lowest in scrub leaf bags, whereas shredders were significantly more abundant in native leaf bags while scrub and exotic leaf bags were intermediate, pasture leaf bags were lowest *($F_{3,35}$=18.30, $P<0.001$)*.

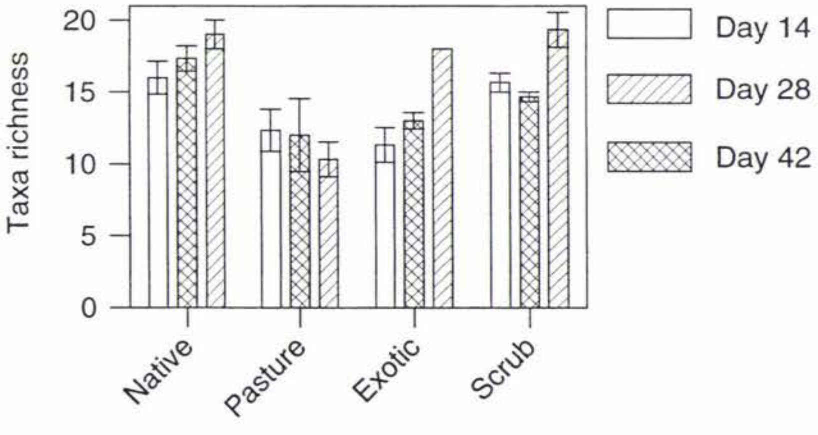

Figure 3.3. (Mean ±S.E.) taxa richness per leaf bag (3mm mesh) of macroinvertebrates collected from 12 streams around Lake Taupo between December 1997 and January 1998, which drain native forest, hill country pasture, exotic forest or scrub catchments.

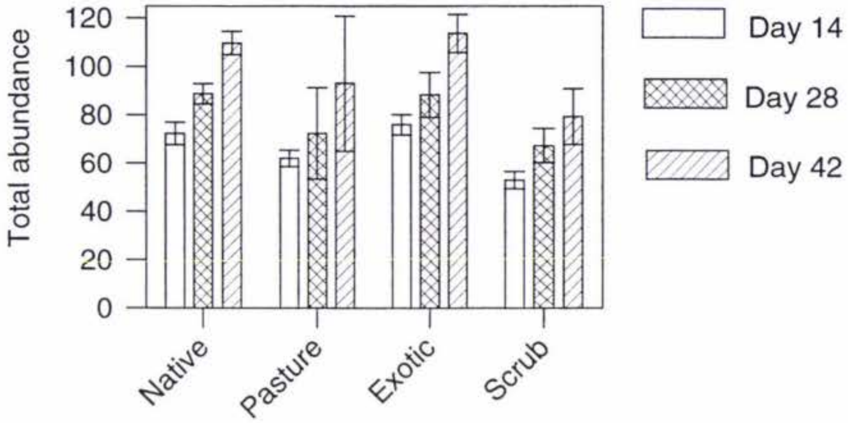

Figure 3.3. (Mean ±S.E.) total abundance per leaf bag (3mm mesh), of macroinvertebrates collected from 12 streams around Lake Taupo between December 1997 and January 1998, which drain native forest, hill country pasture, exotic forest or scrub catchments.

Relative abundance of invertebrate functional feeding groups differed between landuse type and time (Fig. 3.4.). Collectors were more abundant in native and exotic leaf bags than in pasture and scrub leaf bags *($F_{3,35}$=4.65, P=0.01)*. Exotic leaf bags had significantly more browsers than any other landuse type *($F_{3,35}$=12.59, P<0.001)* (Table 3.4.). Except for a few small oligocheates and chironomids, no animals were found in the 0.2mm mesh tubes.

Detrended Correspondence Analysis of invertebrate community composition of leaf bags shows a clear separation of leaf bag communities according to landuse on Axis 1 (Fig.3.5.). The DECORANA for the leaf bag invertebrates shows that leaf bag communities within each site, regardless of the time they were in each stream, were separated according to landuse type.

Table 3.4. F-values, degress of freedom and P-values (* = P<0.001) for two-way ANOVA's performed on invertebrate abundance and functional feeding groups from leaf litter bags between December 1997 and January 1998.

Variables	F-value	d,f	P-value
Shredders			
v. landuse	18.3	3,34	0.001*
v. time	13.78	2,34	0.001*
Collectors			
v. landuse	4.65	3,34	0.01
v. time	9.23	2,34	0.001*
Browsers			
v. landuse	12.59	3,34	0.001*
v. time	3.62	2,34	0.04
Grazers			
v. landuse	0.98	3,34	0.4
v. time	0.7	2,34	0.5
Predators			
v. landuse	3.48	3,34	0.03
v. time	3.44	2,34	0.04
Filter feeders			
v. landuse	7.78	3,34	0.001*
v. time	0.06	2,34	0.9

Positive correlations with axis 1 were found for taxa associated with exotic leaf bags and negative correlations for taxa associated with native leaf bags, pasture and scrub communities were intermediate on axis 1. Native leaf bags had higher abundance of *Aphrophila neozelandica, Austroperla cyrene,* Chironomid sp. 2, Chironomid sp. 4, *Colorburiscus humeralis, Deleatidium* sp., *Oeconesus maori* and oligocheata. Exotic leaf bags had more *Acroperla spiniger,* Chironomid sp. 5, elmidae, *Potamopyrgus antipodarum, Psilochorema* sp., *Stenoperla prasina* and *Zephlebia dentata.*

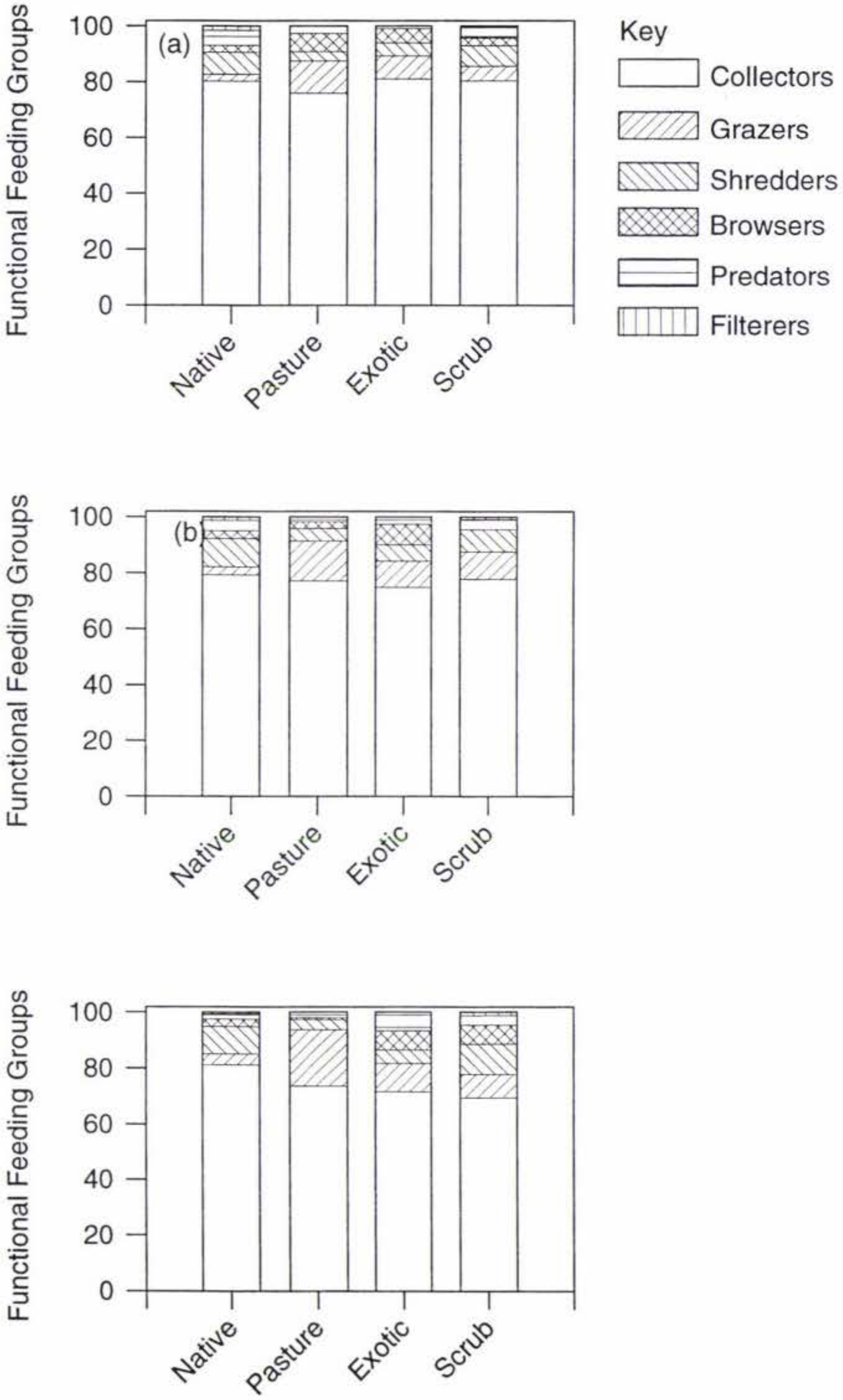

Figure 3.4. Mean relative abundance of individuals from six functional feeding groups at (top) Day 14, (middle) Day 28, (bottom) Day 42, from streams around Lake Taupo between December 1997 and January 1998, which drain native forest, hill country pasture, exotic forest or scrub catchments.

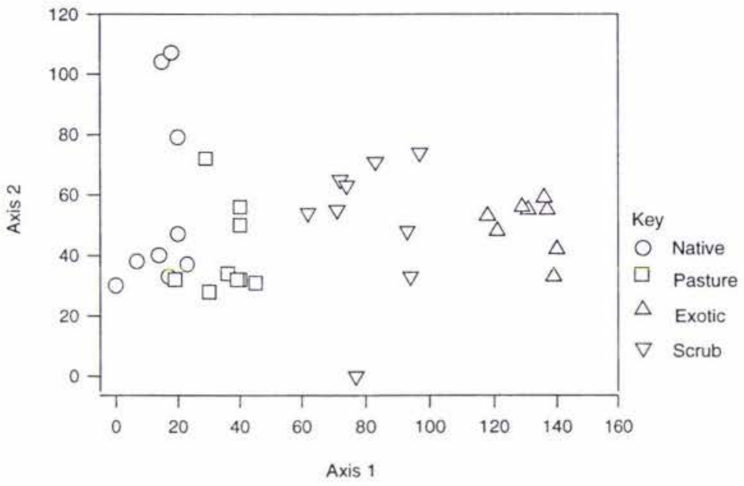

Figure 3.5 Axis 1 of a Detrended Correspondence Analysis (DECORANA) as a function of axis 2 for macroinvertebrate communities collected from 12 streams around Lake Taupo between December 1997 and January 1998, which drain native forest, hill country pasture, exotic forest or scrub catchments.

DISCUSSION

When comparing processing of leaf material in different streams, a number of factors govern breakdown rates such as water temperature, water velocity, nutrients, microbial and invertebrate activity and the quantity and quality of detritus available (Bird and Kaushik, 1992, Davis and Winterbourn, 1977; Riece, 1974; Webster and Benfield, 1986).

Leaf weight losses in tubes were the result of microbial breakdown and leaching losses only. Leaf tubes in streams with open canopies had much faster rates of leaf processing than leaf tubes from streams in forested catchments. Previous studies (Campbell *et al.*, 1992; Reice 1974; Short & Ward, 1980; Webster & Benfield, 1986; Young *et al.*, 1994) have shown that temperature is increased in open canopy streams, which in turn, enhances leaf processing. Increased levels of nutrient inputs (which were not measured in this study) have also been shown to increase leaf litter decomposition and to be higher in agriculturally developed streams (Elwood *et al.*, 1981; Kaushik & Hynes, 1971; Meyer & Johnson, 1983; Sedell, Triska & Triska, 1975; Triska, Sedell & Buckley, 1975). Therefore it is likely that a combination of increased temperature and higher nutrient levels in open streams in this study may have lead to the increased weight loss in the 0.2mm mesh leaf tubes from open canopy streams.

Leaf litter bags decomposed faster in streams which drained exotic forest catchments. Leaf bags from pastoral and native catchments experienced similar weight losses while scrub leaf bag weight losses were lowest. Several workers e.g. Cummins *et al.*, (1973), Anderson and Grafius (1975) have demonstrated the important role played by invertebrates in the break down of leaves in streams. Total invertebrates in this study were higher in exotic leaf bags, native and pasture leaf bags were intermediate and scrub leaf bags were lowest. This pattern was similar to the observed weight loss between landuse types. The link between total invertebrate numbers and the percentage of weight lost in leaf bags, may indicate that faster rates of processing are associated with increased abundance and biomass of invertebrates (Anderson & Sedell, 1977; Hart & Howmiller, 1975; Petersen & Cummins, 1974; Wallace *et al.*, 1982). Increased leaf litter loss in exotic and pasture leaf bags between days 28 and 42, coincided with increased invertebrate abundance in exotic (mean increase 28%) and pasture (mean increase 29%) leaf bags during the same period.

The low abundance of quality leaf detritus within exotic streams may have resulted in the faster processing rates of leaf bags within these streams because of the possible convergence of invertebrates onto a more limited food supply (Bird and Kaushick 1992; Webster and Waide 1982). Pine litter has previously been shown to be nutritionally inferior to other litter types (Taylor *et al.,* 1989, Klemmedson, 1992; Friberg & Jacobsen, 1994) and biological communities and processes associated with allochthonous inputs may be reduced in streams draining exotic forest (Whiles & Wallace, 1997). The leaf litter bags put into streams represented a higher quality food resource.

Ordination of invertebrate community composition of leaf bag invertebrates shows a clear separation according to landuse. However, this was not the case for invertebrate communities from benthic samples (Chapter two) which were not separated by landuse type. Leaf bags in exotic streams had higher abundance of *Acroperla spiniger*, Chironomid sp.5, elmidae, *Potamopyrgus antipodarum* and *Zephlebia dentata*, whereas native stream leaf bags had higher numbers of *Aphrophila neozelandica*, *Austroperla cyrene*, Chironomid sp.2, Chironomid sp.4, *Colorburiscus humeralis* and *Deleatidium* sp. Pasture and scrub leaf bag invertebrates were intermediate. It would seem logical that leaf bags would have a fauna more characteristic of the time the bags were in each stream than landuse type, however this did not occur. It appears then that invertebrates common to each particular landuse type, colonised the leaf bags initially and thereafter community composition in the leaf bags remained relatively similar during the experiment. It is likely that the low abundance of good quality food in the exotic streams in this study increased the numbers of invertebrates colonising exotic leaf bags, whereas food resources within native forests were similar to that of the leaf bags.

In conclusion, higher temperatures and increased nutrient levels that occur in open canopy streams are important factors influencing microbial breakdown of leaf litter. Food resource quality and invertebrate feeding influenced decomposition of leaf litter in 3mm mesh bags. Low abundance of quality food within exotic streams is likely to have caused higher numbers of invertebrates to use the supply of a more limited food item, in turn increasing invertebrate abundance in exotic leaf bags and therefore elevating leaf litter decomposition in exotic sites. Community composition was not effected time but was determined by landuse type.

REFERENCES

Anderson, N.H and J.R. Sedell 1979. Detritus processing by macroinverbrates in stream ecosytems. *Annual Review of Entomology* **24**: 351-377.

Anderson, N.H. and E. Grafius 1975. Utilization and processing of allochtonous material by stream Trichoptera. *Verhandlungen der Internationalen Vereinigung fur theoretisch und angewandte Limnologie* **19**: 3083-3088.

Bird, G.A. and N.K. Kaushik 1992. Invertebrate colonization and processing maple leaf litter in a forested and an agricultural reach of a stream. *Hydrobiologia* **234**: 64-77.

Campbell, I.C., , K.R. James, B.T. Hart and A. Devereaux 1992. Allochthonous coarse particulate organic material in forest and pasture reaches of two south-eastern Australian streams. *Freshwater Biology* **27**: 353-365.

Cummins, K.W., R.C. Peterson, F.O. Howard, J.C. Wuycheck, and V.I. Holt 1973. The utilization of leaf litter by stream detritivores. *Ecology* **54**: 335-345.

Davis, S.F and M.J. Winterbourn 1977. Breakdown and colonisation of Nothofagus leaves in a New Zealand stream. *Oikos* **28**: 250-255.

DeLong, M.D and M.A Brusven 1994. Allochtohnous input of organic material from different riparian habitats of an agriculturally impacted stream. *Environmental Management* **18**: 59-71.

Elwood, J.W., J.D Newbold, A.F Trimble, and R.W Stark 1981. The limiting role of phosphorus in a woodland stream ecosystem: Effects of P enrichment on leaf decomposition and primary producers. *Ecology* **62**; 146-158.

Friberg, N. and D. Jacobsen 1994. Feeding plasticity of two detritivore-shredders. *Freshwater Biology* **32**: 133-142.

Gelroth, J.V. and G.R. Marzolf 1978. Primary production and leaf litter decomposition in natural and channelized portions of a Kansas stream. *American Midland Naturalist* **99**: 238-243.

Griffith, M.B and S.A. Perry 1991. Leaf pack processing in two Applachian mountain streams draining catchements with different management histories. *Hydrobiologia* **220**: 247-254.

Gurtz, M.E. and J.B. Wallace 1984. Substrate-mediated response of stream inverterbrates to disturbance. *Ecology* **65**: 1556-1569.

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.

Hart, S.D and R.P. Howmiller 1975. Studies on the decomposition of allochthonous detritus in two Southern California streams. *Verhandlungen der Internationalen Vereinigung fur theoretisch und angewandte Limnologie* **19**: 1665-1674.

Irons, J., M. Oswood, J. Stout and C. Pringle 1994. Latitudinal patterns in leaf litter breakdown: is temperature really important*? Freshwater Biology* **32**: 401-411.

Jowett, I.G. and J. Richardson 1990. Microhabitat preferences of benthic invertebrates in a New Zealand river and the development of instream flow habitat models for *Deleatidium* sp. *New Zealand Journal of Marine and FreshwaterResearch* **24:** 11-22.

Kauskik, N.K. and H.B.N Hynes 1971. The fate of the dead leaves that fall intostreams. *Archiv fur Hydrobiologie* **68**: 465-515.

Klemmendson, J.O. 1992. Decomposition and nutrient release from mixtures of Gambel Oak and Ponderosa Pine leaf litter. *Forest Ecology Management* **47**: 349-361.

McCune, B. and M.J. Mefford 1995. PC-ORD Multivariate Analysis of Ecological Data, Version 2.0. Oregon: MjM Software Design.

Meyer, J.L and C. Johnson 1983. The influence of elevated nitrate concentration on the rate of leaf decomposition in a stream. *Freshwater Biology* **13**: 177-183.

Meyer, J.L. 1980. Dynamics of phosphorus and organic matter during leaf decomposition in a forest stream. *Oikos* **34**: 44-53.

Newbold, J.D., D.C. Erman and K.B. Roby 1980. Effects of logging on macroinvertebrates in streams with and without buffer strips. *Canadian Journal of Fisheries and aquatic Sciences* **37**: 1076-1085.

Petersen, R.C and K.W. Cummins 1974. Leaf processing in a woodland stream. *Freshwater Biology* **4**: 343-368.

Pfankuch, D. J. 1975. Stream reach inventory and channel stability evaluation. U.S. Department of Agriculture Forest Service/Northern Region.

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:** 579-597.

Quinn, J.M., A.B. Cooper, R.J. Davies-Colley, J.C. Rutherford and R.B. Williamson 1994. Land-use effects on New Zealand hill country streams and implications for riparian management. In: Link, G.L. and Naiman, R.J., editors. Proceedings of the International Workshop on the Ecology and Management of Aquatic-Terrestrial Ecotones. Seattle: University of Washington.

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

Reice, S.R. 1974. Environmental patchiness and the break-down of leaf litter in a woodland stream. *Ecology* **55**: 1271-1282.

Rounick, J.S and M.J. Winterbourn 1983. Leaf processing in two contrasting beech forest streams: effects of physical and biotic factors on litter breakdown. *Archive fur Hydrobiologie* **96**: 448-474.

SAS 1989. SAS User's Guide: Statistics. Version 6.1 SAS Institute Incorporated, Cary, North Carolina, USA.

Sedell, J.R., F.J Triska, and N.S Triska 1975. The processing of conifer and hardwood leaves in two coniferous forest streams. 1. Weight loss and associated invertebrates. *Verhandlungen der Internationalen Vereinigung fur theoretisch und angewante* **19**: 1617-1627.

Short, R.A. and J.V. Ward 1980. Leaf litter processing in a regulated Rocky Mountain stream. *Canadian Journal of Fisheries and Aquatic Sciences* **37**: 123-127.

Stout, B.M. and C.B. Coburn 1989. Impact of highway construction on leaf pack processing in aquatic habitats of eastern Tennessee. *Hydrobiologia* **178**: 233-242.

Suberkropp, K., M.J. Klug and K.W. Cummins 1975. Community processing of leaf litter in a woodland stream. *Verhandlungen, Internationalen Vereinigung fur theroretische und angewandte Limnologie* **19**: 1653-1658.

Sweeny, B.W. 1993. Effects of streamside vegetation on macroinvertebrate communities of white clay creek in eastern North America. *Proceedings of the Academy of Natural Sciences, Philadelphia* **144**: 381-340.

Taylor, B.R, D. Parkinson and R.L Vannote 1986. Growth and production of a stream stonefly: influence of diet and temperature. *Ecology* **67**: 1396-1410.

Triska, F.J., J.R. Sedell and B. Buckley 1975. The processing of conifer and hardwood leaves in two coniferous forest streams. 2. Biochemical and nutrient changes. *Verhandlungen der Internationalen Vereinigung fur theoretisch und angewandte Limnologie* **19**: 1628-1640.

Tuchman, N.C and R.H King 1993. Changes in mechanisms of summer detritus processing between wooded and agricultural sites in Michigan headwater stream. *Hydrobiologia* **268**: 155-127

Wallace, J.B., D.H Ross and J.L. Meyer 1982. Seston and dissolved organic carbon dynamics in a southern Appalachian stream. *Ecology* **63**: 824-838.

Webster, J.R and E.F. Benfield 1986. Vascular plant break-down in freshwater ecosystems. *Annual Review of ecology and Systematics* **17**: 567-594.

Webster, J.R and J.B. Waide 1992. Effects of forest clearcutting on leaf breakdown in a southern Appalachian stream. *Freshwater Biology* **12**: 331-344

Whiles, M.R. and J.B. Wallace 1997. Leaf litter decomposition and macroinvertebrate communities in headwater streams draining pine and hardwood catchments. *Hydrobiologia* **353**: 107-119.

Wilcock, R.J. 1986. Agricultural run-off: a source of water pollution in New Zealand? *New Zealand Agricultural Science* **20**: 98-103.

Winterbourn, M.J. and K.L.W. Gregson 1989. Guide to the Aquatic Insects of New Zealand. *Bulletin of the Entomological Society of New Zealand* 9, 96p.

Young, R.G., A.D. Huryn and C.R. Townsend 1994. Effects of agricultural development on processing of tussock leaf litter in high country New Zealand streams. *Freshwater Biology* **32**: 413-427.

CHAPTER FOUR

General discussion

GENERAL DISCUSSION

The benthic macroinvertebrate communities in the thirty-five streams sampled around Lake Taupo did not form distinctly different groups in regards to catchment landuse type. Previous studies (Collier, 1989; Friberg *et al.*, 1997; Graynoth, 1979; Harding & Winterbourn, 1995; Quinn *et al.,* 1997) have found native and exotic stream community composition to be similar. However, although there were no distinct groups, invertebrate community composition between native and exotic streams was different in this study. Native streams had highly stable stream beds and the lowest temperatures. *Plecoptera* were abundant in native streams, which also had the highest species richness and EPT%. Exotic streams contrast with native sites and possessed the lowest stability, species richness and EPT%.

Increased sand in exotic streams is likely to have resulted in the reduced total invertebrate abundance and taxa richness found at these sites (Ryder, 1991; Waters, 1995). The fine nature of soils in this volcanic region and reduced understory at exotic sites may have contributed to the increased levels of sand in the substrate at these streams. Invertebrate fauna characteristic of increased sedimentation was found at exotic sites, where Elmidae, *Deleatidium* sp and *Potamopyrgus antipodarum* dominated communities. Native sites had abundant *Deleatidium* sp., *Colorburiscus humeralis* and *Aoteapsyche colonica* (Waters, 1995).

A number of factors govern processing of leaf material such as water temperature, current velocity, nutrient levels, microbial and invertebrate activity and the quality and quantity of detritus available (Bird & Kaushik, 1992; Davis & Winterbourn, 1977). Leaf litter tubes (0.2mm) were decomposed faster in open canopy streams than closed canopy streams. Increased temperature and nutrient levels in open canopy streams may have contributed to the elevated weight losses experienced at those sites (Campbell *et al.*, 1992; Elwood *et al.*, 1981; Meyer & Johnson, 1983; ; Sedell, Triska & Triska, 1975; Short & Ward, 1980; Young *et al.,* 1994). Leaf litter bags were decomposed more rapidly in exotic sites where it is thought that the low abundance of quality food available at these sites influenced invertebrates to utilise the higher quality limited food resource. Exotic leaf bags had the highest number of invertebrates in comparison to other landuse types. It is likely that the increased invertebrate abundance at exotic streams contributed to the increased decomposition of leaf bags (Anderson & Sedell, 1977; Hart & Howmiller, 1975; Wallace *et al*, 1982).

Previous studies have noted increased shredder abundance influences the rate on leaf litter decomposition (Whiles & Wallace, 1997; Benfield, Jones & Patterson, 1977). In this study, native leaf bags possessed higher numbers of shredders but this was not reflected in the decomposition rate, although native stream leaf bag weight losses were intermediate (Bird & Kaushik, 1992).

Landuse clearly influenced the macroinvertebrate community composition of leaf litter bags in contrast to the finding that benthic invertebrate communities (Chapter 2) were not separated by landuse type. Landuse effects on invertebrate communities are different throughout New Zealand and will depend on local conditions, especially geology.

REFERENCES

Anderson, N.H and J.R. Sedell 1979. Detritus processing by macroinverbrates in stream ecosytems. *Annual Review of Entomology* **24**: 351-377

Benfield, E.F., D.S. Jones and M.F. Patterson 1977. Leaf pack processing in a pastureland stream. *Oilos* **29**: 99-103.

Bird, G.A. and N.K. Kaushik 1992. Invertebrate colonization and processing of maple leaf litter in a forested and an agricultural reach of a stream. *Hydrobiologia* **234**: 64-77.

Campbell, I.C., , K.R. James, B.T. Hart and A. Devereaux 1992. Allochthonous coarse particulate organic material in forest and pasture reaches of two south-eastern Australian streams. *Freshwater Biology* **27**: 353-365.

Collier, K. J., M. J. Winterbourn and K. J. Jackson 1989. Impacts of wetland afforestation on the distribution of benthic invertebrates in acid streams of Westland, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **23**: 479-490.

Davis, S.F. and Winterbourn, M.J. 1977. Breakdown and colonization of Nothofagus leaves in a New Zealand stream. *Oikos* **28**:250-255.

Elwood, J.W., J.D Newbold, A.F Trimble, and R.W Stark 1981. The limiting role of phosphorus in a woodland stream ecosystem: Effects of P enrichment on leaf decomposition and primary producers. *Ecology* **62**; 146-158.

Friberg, N., M.J. Winterbourn, K. A Shearer. and S. E. Larsen 1997. Benthic communities of forest streams in the South Island, New Zealand: effects of forest type and location. *Archiv für Hydrobiologie* **138(3):** 289-306.

Graynoth, E. 1979. Effects of logging on stream environments and faunas in Nelson. *New Zealand Journal of Marine and Freshwater Research* **13**: 79-109

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.

Hart, S.D and R.P. Howmiller 1975. Studies on the decomposition of allochthonous detritus in two Southern California streams. *Verhandlungen der Internationalen Vereinigung fur theoretisch und angewandte Limnologie* **19**: 1665-1674.

Meyer, J.L and C. Johnson 1983. The influence of elevated nitrate concentration on the rate of leaf decomposition in a stream. *Freshwater Biology* **13**: 177-183.

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:** 579-597.

Ryder, G. I. 1989. Experimental Studies on the Effects of Fine Sediments on Lotic Invertebrates. Unpublished Ph.D. thesis, University of Otago, New Zealand.

Sedell, J.R., F.J Triska, and N.S Triska 1975. The processing of conifer and hardwood leaves in two coniferous forest streams. 1. Weight loss and associated invertebrates. *Verhandlungen der Internationalen Vereinigung fur theoretisch und angewante* **19**: 1617-1627.

Short, R.A. and J.V. Ward 1980. Leaf litter processing in a regulated Rocky Mountain stream. *Canadian Journal of Fisheries and Aquatic Sciences* **37**: 123-127.

Wallace, J.B., D.H Ross and J.L. Meyer 1982. Seston and dissolved organic carbon dynamics in a southern Appalachian stream. *Ecology* **63**: 824-838.

Waters, T.F. 1995. Sediment in Streams: Sources, Biological Effects, and Control. American Fisheries Society Monograph 7.

Young, R.G., A.D. Huryn and C.R. Townsend 1994. Effects of agricultural development on processing of tussock leaf litter in high country New Zealand streams. *Freshwater Biology* **32**: 413-427.

ACKNOWLEDGEMENTS

ACKNOWLEDGEMENTS

I would firstly like to thank my chief supervisor Dr. Russell Death for his valued input to the setting up, study design, statistical analysis and critiquing of this project. Thanks also to my co-supervisor Dr. Ian Henderson, for his guidance, statistical analysis and general good advice.

I also received a great amount of advice, help and knowledge from my fellow stream ecology students at Massey University for whom I extend my appreciation. Reece Fowler and Jonny Horrox for assistance in all aspects of my thesis from identification, statistical analysis and script writing and editing, as well providing me with all the knowledge of computers I now possess. Jason Gibson, Kimberly Dunning and Graeme Franklin for assistance in identification and general advice.

Thanks also to the Ecology department, Massey University, for making this project possible, especially Hayden Hewitt, Kathy Lake and Jens Jorgensen for help with the creation of the leaf litter bags and tubes.

I am grateful to Kate Stewart for her constant motivation, assistance in the field and general assistance in the formatting of my thesis.

Thanks to 'Harry', who in disregard of his condition(s) carried me to and from my study sites, many miles up roads that have never been conquered before by two-wheel drive vehicles, and provided me with a dry place to stay when I was lost.

To my family and friends who have been a constant source of motivation and have given me the support and encouragement I frequently needed. Thank you.

APPENDICES

APPENDIX 1.

Mean invertebrate abundance from four replicate Surber samples taken from 35 streams draining native beech and broadleaf/podocarp forests (N1-N10), hill country pasture (P1-P10), exotic Pinus radiata forest (E1-E10) and scrubland (S1-S5) catchments around Lake Taupo during January and March 1997.

TAXON	N1	N2	N3	N4	N5	N6	N7	N8	N9	N10
Acroperla spiniger	0.3	0.3	3.0	0.3	0.0	0.0	0.0	1.5	0.0	0.0
Amphpoda A	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aoteapsyche colonica	1.3	13.0	0.3	0.0	4.0	16.3	0.8	4.3	0.3	0.0
Aphrophila neozelandica	0.5	0.3	8.5	0.8	0.0	0.0	0.0	0.0	3.5	0.3
Archichauliodes diversus	1.3	5.3	3.8	15.5	4.0	8.5	0.3	0.0	0.3	0.0
Austrosimulium austrolense	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.3	0.0
Austroclima sp.	0.0	4.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Austroperla cyrene	0.0	0.0	0.0	1.0	3.3	8.0	0.0	1.8	1.0	0.5
Beraeoptera roria	0.0	0.3	1.8	2.0	1.8	1.3	0.0	0.0	0.0	0.5
Ceratopogonidae	0.3	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.1	0.0	0.0	5.8	0.0	0.0	0.0	0.0	0.0	0.3	0.8
Chironomid sp.2	1.8	0.0	9.5	0.5	0.0	0.8	0.0	0.0	0.0	0.0
Chironomid sp.3	5.3	0.3	11.0	3.3	2.0	0.5	0.0	0.0	1.5	1.3
Chironomid sp.4	1.3	0.0	5.3	0.5	0.0	0.5	0.0	0.0	0.0	0.0
Chironomid sp.5	0.8	0.0	1.3	0.3	0.8	0.5	0.5	0.0	2.5	2.8
Chironomid sp.6	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
Chironomid sp.7	1.8	0.3	0.0	1.3	0.0	2.0	0.3	0.0	0.0	0.0
Chironomid sp.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.10	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.11	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	1.0	5.0
Chironomid sp.12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.13	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Colorburiscus humeralis	0.0	6.0	0.0	28.3	9.5	62.3	3.5	14.3	4.8	0.3
Confluens sp.	0.0	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Deleatidium sp.	12.8	38.0	4.8	56.8	44.3	0.0	25.8	22.0	13.0	35.0
Elmidae	0.0	4.0	11.8	26.0	7.5	5.5	0.3	1.3	4.8	11.3
Empididae	0.3	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eriopterini sp.	0.3	0.0	0.0	1.3	13.5	15.3	8.0	2.8	0.3	3.0
Helicopsyche albescens	0.0	3.3	0.0	1.0	0.0	0.0	0.0	0.3	0.0	0.0
Hudsonema amabilis	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrobiosis parumbripennis	1.8	2.0	1.5	2.0	1.5	1.5	0.0	0.0	1.0	1.8
Hydora	0.3	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydraenidae	0.0	0.3	0.0	2.5	0.3	0.0	0.0	0.0	0.0	0.0
Hydrobiosella sp.	0.0	0.0	0.3	0.0	0.8	1.0	0.0	0.0	0.0	0.0
Hydrometridae	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Megaleptaperla diminuta	0.5	5.8	0.5	5.3	7.0	4.3	0.3	1.0	0.8	0.5
Neocurupira hudsoni	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nesameletus sp.	0.0	0.0	0.3	0.3	4.3	0.8	0.0	0.3	0.0	24.3
Neurochorema confusum	2.5	0.0	1.8	2.3	0.8	0.0	0.3	0.0	0.0	0.5
Oeconesus maori	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.3	0.0	0.0
Oligocheata	0.0	0.0	0.5	0.0	0.0	5.3	0.0	13.3	0.0	0.0
Olinga feredayi	0.3	0.8	2.3	5.0	0.0	0.0	0.0	0.0	0.0	0.0
Paranephrons planifrons	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Paralimnophila skusei	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Physa acuta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Potamopyrgus antipodarum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Psilochorema sp.	0.3	0.0	0.5	0.3	0.0	1.0	0.0	0.8	0.0	3.5
Ptilodactylidae	0.0	1.3	0.8	1.0	2.8	1.3	0.3	1.0	0.3	0.0
Pycnocetrella sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Pycnocentria sp.	0.0	1.0	7.5	0.5	0.0	0.0	0.3	0.0	0.0	0.0
Pycnocentrodes sp.	0.5	0.0	1.0	0.5	0.3	0.0	0.3	0.5	0.0	0.0
Rallidens mcfarlanei	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Scirtidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stenoperla prasina	0.0	0.0	0.0	2.0	6.0	5.8	1.5	3.3	0.0	0.5
Stratiomyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tanyderidae	0.0	0.0	0.0	0.5	1.8	0.3	0.3	0.3	0.0	0.0
Zelolessica sp.	0.0	0.0	15.0	0.5	0.8	1.8	0.0	0.3	0.0	0.3
Zelandobius confusus	0.5	0.0	2.5	5.3	1.8	1.8	1.0	6.0	0.0	0.0
Zelandoperla sp.	9.0	0.0	0.0	0.0	0.3	2.3	0.0	0.3	0.8	5.3
Zephlebia dentata	0.0	0.0	18.5	1.5	0.3	4.3	3.0	4.3	0.0	0.0

TAXON	P1	P2	P3	P4	P5	P6	P7	P8	P9	P10
Acroperla spiniger	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amphpoda A	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0
Aoteapsyche colonica	1.8	0.8	0.0	0.3	0.8	1.0	0.8	0.0	0.0	0.0
Aphrophila neozelandica	0.0	2.3	2.3	21.0	0.5	0.0	2.3	0.0	1.3	0.0
Archichauliodes diversus	0.0	14.3	5.8	0.8	4.5	2.0	1.5	0.0	0.0	0.0
Austrosimulium austrolense	0.0	0.0	0.0	0.5	0.0	4.0	0.0	0.3	0.0	0.0
Austroclima sp.	0.0	4.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Austroperla cyrene	0.0	0.3	2.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Beraeoptera roria	0.0	1.8	0.0	2.8	0.0	0.0	0.0	0.0	0.0	0.0
Ceratopogonidae	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.1	0.0	16.3	1.0	3.5	3.5	0.3	0.0	0.5	14.0	0.0
Chironomid sp.2	0.0	9.8	0.3	5.8	0.0	0.0	0.8	0.8	0.0	0.0
Chironomid sp.3	0.8	4.5	0.5	8.3	2.8	2.8	18.5	0.0	15.5	0.0
Chironomid sp.4	0.0	2.3	0.0	12.3	0.3	0.5	0.0	0.0	2.8	0.0
Chironomid sp.5	0.0	0.0	0.0	0.0	0.8	0.0	0.3	0.5	0.0	0.0
Chironomid sp.6	0.0	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.7	0.0	0.5	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.8	0.0	0.3	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
Chironomid sp.9	0.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.10	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.11	2.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.12	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.13	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Colorburiscus humeralis	0.0	4.0	9.0	0.0	5.3	13.8	5.0	0.0	0.0	0.0
Confluens sp.	0.0	0.0	0.0	8.8	0.0	0.0	0.0	0.0	0.0	0.0
Deleatidium sp.	0.0	56.5	39.8	3.0	37.8	70.0	31.5	0.0	30.3	190.8
Elmidae	0.0	46.0	49.8	17.3	9.5	7.5	20.0	0.0	3.8	2.0
Empididae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eriopterini sp.	0.5	1.5	4.0	0.0	0.0	3.0	0.5	0.0	0.8	0.0
Helicopsyche albescens	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Hudsonema amabilis	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
Hydrobiosis parumbripennis	0.5	2.3	1.3	1.3	0.5	1.3	0.0	2.3	3.0	0.0
Hydora	0.0	6.5	0.3	0.5	0.3	0.0	0.0	0.0	0.0	0.0
Hydraenidae	0.0	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrobiosella sp.	0.3	0.0	0.0	0.0	0.3	0.3	0.5	0.0	0.8	1.0
Hydrometridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Megaleptaperla diminuta	0.3	2.5	2.3	0.8	0.5	0.0	4.3	0.0	1.5	0.0
Neocurupira hudsoni	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Nesameletus sp.	0.0	0.3	0.3	1.5	0.0	0.5	0.0	0.0	0.0	0.3
Neurochorema confusum	0.3	0.5	2.3	1.5	0.5	0.3	1.0	0.0	0.0	0.3
Oeconesus maori	0.8	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
Oligocheata	3.0	0.0	0.0	0.0	0.0	0.0	0.0	32.3	0.5	0.0
Olinga feredayi	0.0	2.5	0.5	2.3	0.3	0.0	0.0	0.0	0.0	1.0
Paranephrons planifrons	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Paralimnophila skusei	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Physa acuta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	17.0	0.0	0.0
Potamopyrgus antipodarum	0.0	0.0	0.0	0.0	0.0	0.0	13.3	0.0	0.8	0.0
Psilochorema sp.	0.3	0.3	0.0	0.3	0.3	0.0	0.0	0.8	0.0	0.0
Ptilodactylidae	1.0	5.0	1.8	0.0	0.3	0.0	0.0	0.0	0.0	0.0
Pycnocetrella sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pycnocentria sp.	0.0	1.0	0.3	17.0	0.5	0.3	0.3	0.0	0.0	12.3
Pycnocentrodes sp.	0.0	7.8	0.0	35.0	3.8	0.0	0.0	0.0	0.3	5.3
Rallidens mcfarlanei	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scirtidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stenoperla prasina	0.0	0.0	2.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Stratiomyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Tanyderidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Zelolessica sp.	0.0	0.0	0.0	42.5	1.5	0.0	0.0	0.0	0.0	0.0
Zelandobius confusus	0.0	4.8	9.5	7.0	1.5	1.3	5.0	0.0	12.5	1.8
Zelandoperla sp.	0.0	0.3	1.0	0.8	0.0	0.0	3.5	0.0	0.5	0.0
Zephlebia dentata	5.0	0.0	2.8	7.0	4.0	0.5	0.0	0.0	0.0	0.3

TAXON	E1	E2	E3	E4	E5	E6	E7	E8	E9	E10
Acroperla spiniger	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0
Amphpoda A	0.5	0.0	0.3	0.0	0.3	0.0	1.0	1.3	0.0	0.0
Aoteapsyche colonica	0.0	0.0	11.3	0.5	0.0	0.0	0.3	5.0	0.5	0.0
Aphrophila neozelandica	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
Archichauliodes diversus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	11.8	10.5
Austrosimulium austrolense	0.0	0.0	2.5	0.3	0.0	0.0	0.0	0.5	0.0	0.0
Austroclima sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Austroperla cyrene	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	3.3	1.3
Beraeoptera roria	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.8	1.0
Ceratopogonidae	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.3	0.0
Chironomid sp.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.3	0.0	1.3	0.5	1.8	0.5	0.0	0.3	4.5	0.3	0.8
Chironomid sp.4	0.0	0.0	5.5	0.0	0.0	0.0	0.5	0.0	0.0	0.0
Chironomid sp.5	1.3	0.0	0.3	0.8	0.0	0.0	10.0	7.8	0.0	0.0
Chironomid sp.6	0.0	0.0	0.0	0.0	1.3	0.0	0.0	0.0	0.0	0.3
Chironomid sp.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.10	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.11	0.0	0.0	0.3	0.0	0.0	0.3	0.0	0.3	1.3	0.0
Chironomid sp.12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomid sp.13	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.3
Colorburiscus humeralis	1.3	0.0	0.0	0.0	0.0	0.3	0.0	0.0	8.3	5.3
Confluens sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Deleatidium sp.	0.0	0.3	0.0	8.8	6.0	0.8	0.3	0.0	48.3	25.8
Elmidae	0.0	1.5	0.8	3.0	3.5	0.8	0.0	0.8	58.5	42.8
Empididae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eriopterini sp.	0.0	0.0	0.0	2.0	6.5	0.5	4.5	0.0	5.5	6.8
Helicopsyche albescens	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hudsonema amabilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrobiosis parumbripennis	0.0	0.0	1.8	0.5	0.3	0.0	0.0	0.0	1.0	0.0
Hydora	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydraenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0
Hydrobiosella sp.	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.3	0.0
Hydrometridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Megaleptaperla diminuta	0.5	0.0	0.0	1.5	2.0	0.0	5.0	0.3	3.8	2.5
Neocurupira hudsoni	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nesameletus sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0	3.8
Neurochorema confusum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0
Oeconesus maori	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oligocheata	0.0	0.0	2.5	1.5	1.0	2.8	0.8	0.0	0.0	0.0
Olinga feredayi	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	1.0	0.3
Paranephrons planifrons	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Paralimnophila skusei	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Physa acuta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Potamopyrgus antipodarum	0.5	0.0	1.3	0.0	0.0	0.0	0.0	41.8	0.0	0.0
Psilochorema sp.	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0
Ptilodactylidae	0.3	0.0	11.8	0.3	0.8	0.8	1.8	0.0	1.8	0.5
Pycnocetrella sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pycnocentria sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.5	0.0	0.3
Pycnocentrodes sp.	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0
Rallidens mcfarlanei	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scirtidae	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stenoperla prasina	0.0	0.0	0.0	0.0	0.3	0.3	1.5	0.0	1.0	0.0
Stratiomyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tanyderidae	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	2.0	0.3
Zelolessica sp.	0.3	0.0	0.0	0.8	0.0	0.0	2.0	40.8	0.0	0.8
Zelandobius confusus	0.3	0.5	0.0	0.3	0.3	0.0	0.0	0.5	0.0	1.0
Zelandoperla sp.	0.0	0.0	1.8	1.3	0.0	0.0	0.0	0.0	0.3	0.5
Zephlebia dentata	11.0	0.0	80.8	1.0	0.5	2.3	21.8	44.3	1.3	0.8

TAXON	S1	S2	S3	S4	S5
Acroperla spiniger	0.0	0.0	0.0	0.0	0.0
Amphpoda A	0.0	0.0	0.0	0.0	0.0
Aoteapsyche colonica	2.5	9.5	1.5	0.0	3.5
Aphrophila neozelandica	4.8	0.5	0.0	0.0	0.8
Archichauliodes diversus	0.0	4.5	0.5	0.0	0.3
Austrosimulium austrolense	0.0	0.0	0.3	0.0	0.8
Austroclima sp.	0.0	0.0	0.0	0.0	0.0
Austroperla cyrene	1.0	1.0	0.3	0.0	0.0
Beraeoptera roria	0.5	0.5	0.0	0.0	0.0
Ceratopogonidae	0.0	0.0	0.0	0.0	0.0
Chironomid sp.1	11.0	0.0	0.3	0.0	0.8
Chironomid sp.2	0.8	2.5	0.8	0.0	2.0
Chironomid sp.3	13.0	2.3	5.8	0.5	6.5
Chironomid sp.4	0.0	0.0	0.8	0.0	29.3
Chironomid sp.5	1.5	0.0	0.0	0.0	3.3
Chironomid sp.6	0.5	0.5	0.0	0.0	0.3
Chironomid sp.7	3.8	4.0	4.3	0.0	0.0
Chironomid sp.8	0.0	0.0	0.0	0.0	0.0
Chironomid sp.9	0.0	0.0	0.0	0.0	0.0
Chironomid sp.10	0.0	0.0	0.0	0.0	0.8
Chironomid sp.11	0.8	0.5	0.3	0.3	2.0
Chironomid sp.12	0.0	0.0	0.0	0.0	0.0
Chironomid sp.13	0.0	0.0	0.0	0.0	0.0
Colorburiscus humeralis	1.3	22.5	15.0	0.3	6.3
Confluens sp.	0.0	0.0	0.0	0.0	0.0
Deleatidium sp.	15.3	54.3	73.0	14.5	61.3
Elmidae	18.8	43.3	30.8	0.5	128.5
Empididae	0.0	0.0	0.0	0.0	0.0
Eriopterini sp.	0.8	15.8	7.0	0.3	6.3
Helicopsyche albescens	0.0	0.0	0.0	0.0	0.0
Hudsonema amabilis	0.0	0.0	0.0	0.0	0.0
Hydrobiosis parumbripennis	1.3	0.8	1.0	0.0	4.3
Hydora	0.0	0.0	0.0	0.0	0.0
Hydraenidae	0.0	0.0	0.0	0.0	0.0
Hydrobiosella sp.	0.0	0.0	0.3	1.3	0.0
Hydrometridae	0.0	0.0	0.0	0.0	0.0
Hydrophilidae	0.0	0.0	0.0	0.0	0.0
Megaleptaperla diminuta	2.8	0.0	0.3	0.3	0.5
Neocurupira hudsoni	0.0	0.0	0.0	0.0	0.0
Nesameletus sp.	0.0	0.0	0.0	0.0	0.0
Neurochorema confusum	3.8	1.5	0.0	0.0	0.8
Oeconesus maori	0.0	0.0	0.0	0.0	0.0
Oligocheata	0.0	0.0	1.3	0.0	1.5
Olinga feredayi	0.0	0.5	1.3	0.0	0.3
Paranephrons planifrons	0.0	0.0	0.0	0.0	0.0
Paralimnophila skusei	0.0	0.0	0.0	0.0	0.0
Physa acuta	0.0	0.0	0.0	0.0	0.0
Potamopyrgus antipodarum	0.3	0.0	0.0	0.0	0.3
Psilochorema sp.	0.0	0.3	0.0	0.0	0.0
Ptilodactylidae	0.0	0.0	0.0	0.0	0.0
Pycnocetrella sp.	0.0	0.0	0.0	0.0	0.0
Pycnocentria sp.	0.0	0.5	0.0	0.0	0.5
Pycnocentrodes sp.	0.3	0.5	0.0	0.0	0.0
Rallidens mcfarlanei	0.0	0.0	0.0	0.0	0.0
Scirtidae	0.0	0.0	0.0	0.0	14.8
Stenoperla prasina	0.3	0.3	0.3	0.0	0.0
Stratiomyidae	0.0	0.0	0.0	0.0	0.0
Tanyderidae	0.3	0.0	0.0	0.0	0.0
Zelolessica sp.	1.8	0.3	0.0	0.0	0.5
Zelandobius confusus	0.0	6.5	0.5	0.3	0.3
Zelandoperla sp.	8.0	0.3	0.5	0.0	0.0
Zephlebia dentata	0.5	0.0	0.0	0.0	0.0

APPENDIX 2

Mean invertebrate abundance from leaf litter bags in 12 streams draining native beech /podocarp forests (NA1-NA3), hill country pasture (PA1-PA3),exotic Pinus radiata forest and broadleaf (EA1-EA3) or scrubland (SA1-SA3) catchments around Lake Taupo 1997 and January 1998 between December .

TAXON	NA1	NA2	NA3	NB1	NB2	NB3	NC1	NC2	NC3
Aoteapsyche colonica	0.00	0.00	0.33	0.00	0.67	0.67	0.00	0.67	0.33
Aphrophilla neozelandia	1.00	1.00	1.67	1.00	1.33	1.00	0.67	0.33	0.67
Archichauliodes diversus	0.33	0.33	0.00	0.67	0.67	0.33	0.67	0.33	0.00
Austroperla cyrene	0.33	0.08	2.00	1.00	0.00	3.33	0.67	1.00	1.00
Chironomid sp.1	14.33	14.08	13.67	23.33	20.00	17.33	28.00	38.33	24.67
Chironomid sp.2	6.33	14.08	14.00	12.67	10.00	10.33	10.67	9.67	9.67
Chironomid sp.3	5.67	6.17	6.33	5.33	3.33	4.33	1.33	4.33	7.00
Chironomid sp.4	4.67	3.92	9.33	3.33	7.00	7.33	2.33	6.67	0.00
Chironomid sp.11	11.33	13.83	9.33	15.67	21.33	17.00	23.00	21.67	22.67
Colorburiscus humeralis	0.67	0.17	2.00	0.67	0.00	1.00	0.00	0.33	0.33
Deleatidium sp.	2.00	0.50	3.33	2.33	0.00	5.00	3.00	0.00	5.67
Eriopterini sp.	2.33	0.58	1.67	3.67	0.00	3.00	4.00	0.00	2.33
Hydrobiosis parumbripennis	1.00	0.50	0.00	1.00	0.67	0.00	0.33	0.67	0.00
Hydrophilidae	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Megaleptaperla diminuta	1.67	2.17	2.00	1.00	1.00	1.00	0.33	0.33	0.67
Oeconesus maori	0.33	0.08	0.00	0.00	1.00	0.33	0.67	0.67	0.33
Oligocheata	7.33	6.83	6.67	6.67	8.00	11.33	13.33	14.67	23.00
Olinga feredayi	0.00	1.25	0.00	0.00	2.67	1.33	1.00	2.67	1.67
Potamopyrgus antipodarum	2.00	0.50	3.33	2.33	0.00	5.33	2.67	1.33	8.67
Ptilodactylidae	0.67	2.42	1.67	0.67	3.67	2.00	2.00	2.67	2.00
Stenoperla prasina	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.67
Zelandobius confusus	2.33	1.08	2.67	3.67	1.00	4.67	5.00	2.33	5.00
Zellolessica sp.	0.00	1.25	0.00	0.00	1.33	0.00	0.00	3.67	0.00

TAXON	PA1	PA2	PA3	PB1	PB2	PB3	PC1	PC2	PC3
Aphrophilla neozelandica	0.33	0.08	1.00	0.67	0.00	0.00	0.00	0.00	0.00
Austroperla cyrene	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.67
Ceratopognidae	1.00	0.25	0.00	0.67	0.33	0.00	0.33	0.00	0.00
Chironomid sp.1	13.33	11.08	10.33	12.00	16.67	11.33	19.67	22.00	13.67
Chironomid sp.2	7.00	5.00	9.00	6.00	11.00	6.67	16.67	14.67	2.67
Chironomid sp.3	7.00	4.75	7.00	4.67	9.67	0.00	10.00	10.67	6.67
Chironomid sp.4	3.33	5.33	8.00	6.33	2.33	11.67	3.00	5.00	7.33
Chironomid sp.11	13.00	9.75	14.00	14.33	23.33	14.67	9.00	10.00	19.00
Deleatidium sp.	1.67	2.42	3.67	2.33	2.00	0.00	0.00	2.00	0.00
Elmidae	0.67	0.92	2.33	0.33	0.67	0.00	0.00	0.00	0.00
Eriopterini sp.	1.00	0.25	0.00	1.33	0.00	0.00	2.67	0.00	0.00
Hydrobiosis parumbripennis	0.67	0.67	0.00	0.33	0.67	0.00	1.00	0.00	0.00
Megaleptaperla diminuta	0.00	0.75	0.00	0.00	0.67	0.00	2.67	1.00	0.00
Oligocheata	2.33	7.08	4.00	2.00	11.67	2.33	6.00	24.33	3.00
Potamopyrgus antipodarum	1.33	14.58	0.00	2.67	27.33	0.00	0.00	56.33	0.00
Pycnocentrodes sp.	1.00	0.25	0.00	0.33	0.00	0.67	0.00	0.00	0.00
Zelandobius confusus	2.00	1.50	3.00	2.33	3.33	4.00	3.33	2.33	4.00

TAXON	EA1	EA2	EA3	EB1	EB2	EB3	EC1	EC2	EC3
Acroperla spiniger	0.67	0.17	0.00	1.33	0.00	0.00	1.00	0.33	1.33
Archichauliodes diversus	0.00	0.00	0.00	0.00	0.00	0.00	0.33	1.00	2.67
Chironomid sp.1	18.00	13.75	9.33	18.00	23.33	16.33	23.00	16.00	30.33
Chironomid sp.3	7.67	13.92	12.00	11.67	21.33	10.33	9.00	12.00	13.00
Chironomid sp.4	0.00	1.75	0.00	1.33	1.67	3.00	3.33	2.33	3.33
Chironomid sp.5	18.67	11.42	25.67	18.00	11.00	9.67	13.67	21.67	22.00
Chironomid sp.11	15.00	18.00	12.33	18.33	9.67	9.67	18.00	20.33	14.67
Elmidae	3.67	5.17	1.33	5.00	7.00	2.67	6.00	6.00	4.67
Eriopterini sp.	1.00	0.25	0.00	1.67	0.33	2.67	2.00	3.00	5.33
Hydrobiosis parumbripennis	0.00	0.25	0.00	0.00	0.33	0.00	1.00	0.67	0.33
Megaleptaperla diminuta	0.33	0.33	0.00	0.67	0.67	0.00	1.67	1.00	5.67
Oligocheata	3.00	3.25	1.00	5.00	4.00	1.00	5.67	1.67	4.33
Potamopyrgus antipodarum	10.00	4.75	5.67	13.67	4.33	7.00	22.67	4.00	7.33
Psilochorema sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.67	0.33	0.67
Ptilodactylidae	4.33	1.08	0.00	5.33	0.00	0.00	3.00	2.33	4.00
Stenoperla prasina	0.33	0.33	0.00	1.33	2.00	1.33	2.00	2.00	2.00
Zelandobius confusus	1.67	2.17	2.33	2.33	3.00	4.67	2.33	2.67	2.00
Zephlebia dentata	0.00	0.00	0.67	0.00	0.00	3.33	0.67	1.33	2.00

TAXON	SA1	SA2	SA3	SB1	SB2	SB3	SC1	SC2	SC3
Aoteapsyche colonica	0.33	0.08	0.33	0.33	0.67	0.33	1.00	0.33	1.00
Archichauliodes diversus	0.33	0.08	0.00	1.00	0.67	1.00	0.67	0.67	0.67
Austroperla cyrene	0.67	0.17	0.00	0.67	0.00	0.33	0.67	0.33	0.33
Chironomid sp.1	12.67	6.67	10.00	12.00	9.33	6.33	18.33	9.00	8.33
Chironomid sp.2	11.00	8.50	16.67	12.67	11.67	12.67	11.67	5.33	14.67
Chironomid sp.4	3.67	7.67	5.00	1.00	1.00	1.33	8.33	7.00	5.00
Chironomid sp.5	4.67	1.42	1.00	16.67	19.67	2.00	3.67	7.67	14.33
Chironomid sp.11	8.00	21.00	4.67	10.00	20.00	9.67	24.00	7.67	6.67
Ceratopogonidae	2.33	1.58	1.00	0.33	1.00	2.00	1.00	1.00	2.67
Elmidae	0.67	1.92	1.33	0.00	0.00	0.00	6.67	5.00	4.67
Eriopterini sp.	1.00	1.75	1.00	1.00	4.33	6.00	3.33	1.33	1.67
Hudsonema amabilis	0.33	0.33	0.00	0.67	1.33	0.00	1.00	0.67	0.33
Oligochaeta	0	0	0.00	0.00	0	0.00	3.67	1.67	2.67
Olinga feredayi	1.33	1.58	2.33	2.67	1.00	2.67	2.67	3.00	0.00
Physa acuta	0.00	0.00	0.00	0.00	0.00	5.67	0.00	1.33	1.00
Potamapyrgus antipodarum	1.67	2.42	2.33	3.67	6.00	4.67	7.00	3.67	5.33
Pycnocentria sp.	0.33	0.08	0.67	0.00	0.00	1.00	1.33	0.33	0.00
Pycnocentrodes sp.	0.67	0.42	0.67	0.00	0.00	0.00	1.00	0.33	0.00
Stenoperla prasina	0.00	0.50	0.33	0.00	1.33	0.00	0.33	1.00	0.33
Zelandobius confusus	1.67	0.92	1.00	1.00	2.33	1.33	3.67	4.67	5.33
Zellolessica sp.	0.00	0.50	0.00	0.00	1.00	0.00	1.00	0.33	0.00