Effects of exotic forestry on stream macroinvertebrates: the influence of scale in North Island, New Zealand streams

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ABSTRACT

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The effects of exotic forestry, particularly harvesting, on stream macroinvertebrate community structure was assessed in four regions, Lismore, Tawarau, Pirongia and Te Wera, of central North Island, New Zealand. A survey of 42 streams draining exotic forest (of different ages), native forest or pasture catchments was conducted in January and February 1997. Stream macroinvertebrate communities in 12 of these streams were further monitored every two months for 11 months, to assess changes in macroinvertebrate communities following harvesting and road construction.

Recent forest harvesting (within six years) was associated with an increase in the abundance of pollution tolerant taxa (e.g., Oligochaeta, Crustacea and Mollusca) and a decrease in the abundance of Ephemeroptera and Plecoptera, indicating a shift in community structure. Diversity was not affected by harvesting however. Communities in streams draining mature exotic forest were similar to those in streams draining native forest, both being dominated by Ephemeroptera. Similarly, streams in harvested and pasture catchments were comparable. However, invertebrate response to forestry differed in the four regions, with Lismore and Te Wera forests showing distinct differences in community composition between streams in mature exotic forest and those in harvested exotic forest, while Pirongia and Tawarau showed little or no difference. Large scale factors such as geology were important in determining the response of communities to forest harvesting through their influence on substrate characteristics and susceptibility to erosion, and masked clear differences between land use types when compared between regions. Within regions however, communities differed more between land use types.

Pre and post harvest monitoring revealed that changes in community composition were immediate from commencement of harvesting. Road construction had little effect on community composition but did lead to increased abundance of invertebrates. Physiochemical characteristics associated with differences in macroinvertebrate community structure between streams in harvested and mature exotic forest included sedimentation, stream stability and removal of riparian vegetation which altered light and water temperature regimes, and invertebrate food sources.

The effect of sedimentation was tested experimentally for 21 days in three streams draining mature exotic forest catchments and two draining pasture catchments. Total abundance of macroinvertebrates decreased significantly with light and heavy sedimentation, and community composition showed a change from Ephemeroptera to Coleoptera dominated. Taxa most affected by sedimentation were Ephemeroptera and Plecoptera. Stream communities in exotic forest showed different responses to sedimentation than those in pasture, indicating that if pollution or sediment tolerant taxa are already present, sedimentation may not change the community significantly.

Keywords: community composition, exotic forestry, harvesting, land use, macroinvertebrates, riparian vegetation, sedimentation, spatial scale, temporal scale, water quality.

CHAPTER ONE

General introduction

GENERAL INTRODUCTION

The pattern of ecological response to disturbance provides a means of predicting and measuring the effectiveness of stream management regimes (Statzner et al., 1997). Exotic forestry is a major industry in New Zealand and current resource management legislation demands that any adverse effects of forestry or other land use practices are minimised. Forest harvesting is potentially the most damaging forest management practice with respect to stream disturbance and invertebrate community structure (Campbell and Doeg, 1989; O'Loughlin, 1995), changing the stream environment by influencing light and temperature regimes, nutrient concentrations, riparian vegetation input, substrate characteristics and water chemistry (Campbell and Doeg, 1989). Many overseas studies have found long term community changes such as higher abundance and lower diversity of macroinvertebrates in streams with harvested catchments (e.g., Growns and Davis, 1991; Ormerod et al., 1993; Fore and Karr, 1996; Brown et al., 1997). In contrast, such invertebrate responses to forest harvesting in New Zealand appear to be short term (Graynoth, 1979; Winterbourn and Rounick, 1985). However, in New Zealand, studies on the effects of forest harvesting on stream communities are less numerous and almost solely confined to the South Island (e.g., Graynoth, 1979; Winterbourn and Rounick, 1985; Winterbourn, 1986; Collier et al., 1989) despite the fact most exotic forests are concentrated in the North Island (Bloomfield and Watson, 1988).

Scale has been identified as one of the most important underlying factors determining patterns and processes in ecology (Levin, 1992; Pickett and Cadenasso, 1995). Recent attention in fresh water ecology has focused on the influence of scale on invertebrate responses to disturbance (Biggs et al., 1990; Biggs and Gerbeaux, 1993; Scarsbrook and Townsend, 1993; Fisher, 1994; Biggs, 1995; Friberg and Winterbourn, 1997; Friberg et al., 1997; Townsend et al., 1997). The traditional approach in ecology has been to study communities on small spatial and temporal scales at sites with similar characteristics i.e., climate, topography and geology (Levin, 1992). However, large scale influences on stream communities, such as geology and climate, may also be important in determining macroinvertebrate communities, limiting the applicability of studies to other regions. If regional differences do occur they may influence the response of communities to forestry and other land use types between regions (Biggs and Gerbeaux, 1993; Biggs, 1995; Friberg et al., 1997; Harding et al., 1997; Richards et al., 1997; Townsend et al., 1997). Although the effects of land use on stream communities has been studied in New Zealand at several spatial scales (e.g., Quinn and Hickey, 1990; Friberg et al., 1997; Quinn et al., 1997), none have considered how responses of invertebrates to harvesting or forest age differ with spatial scale.

Environmental factors that influence invertebrate communities also vary seasonally e.g., changes in light, temperature and rainfall; and with time since disturbances such as forest harvesting (Richards and Minshall, 1992; Hildrew and Giller, 1994), potentially influencing invertebrate population dynamics and community organisation (Levin, 1992). The effects on invertebrate communities during and immediately after forest harvesting may therefore be different from those six months or six years after harvesting, and may also differ throughout the year. Temporal studies are poorly represented in New Zealand (e.g., Winterbourn, 1978; Towns, 1981; Scarsbrook and Townsend, 1993) and there appear to be no studies on the "recovery" of invertebrate communities immediately after forest harvesting.

The influence of spatial and temporal scale on communities is important both from an ecological and management perspective. Ecologically, the influence of scale helps to elucidate the most important influences on community structure. Furthermore, how invertebrates respond to disturbance at several scales will bring greater understanding to the relationship between different scales (Levin, 1992; Allan et al., 1997); the key to what determines patterns and process of communities (Fisher, 1994). This understanding can then be incorporated into management planning as a predictive and monitoring tool (Larsen et al., 1988; Levin, 1992; Richards and Minshall, 1992). Regional influences may need to be acknowledged (Allan et al., 1997), and forest management plans adjusted on a regional basis. Factors operating over time must also be considered so that influences of disturbances on communities can be understood within the framework of natural variation (Larsen et al., 1988).

To this end, the effects of forest age on stream macroinvertebrate community structure within and between regions, was investigated in Chapter Two and comparisons made between streams in exotic forest, native forest and pasture catchments. Chapter Three examined the short term effects of forest harvesting, including before and after harvest monitoring, and the importance of seasonal changes on community responses to forest harvesting and other land use practices. Sedimentation, identified as an important effect of harvesting from the survey and monitoring studies, was investigated further in Chapter Four using an experimental approach to quantify the effect of sedimentation on macroinvertebrate community structure in streams draining exotic forest and pasture. Chapter Five discusses the findings of previous chapters from an overall perspective, and Chapter Six is a Management Report completed for Rayonier New Zealand Ltd. which summarises findings and gives recommendations to minimise the effects of forest harvesting on stream communities.

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CHAPTER TWO

The effects of exotic forestry on stream invertebrate communities: are they scale dependent?

ABSTRACT

The effects of exotic forestry on stream macroinvertebrate communities was assessed in four regions, Lismore, Tawarau, Pirongia and Te Wera, of central North Island, New Zealand. Forty two streams draining exotic *Pinus radiata* (of differing age), native forest or pasture were sampled with a kick net and quadrate in January and February 1997. Recent forest harvesting (within six years) was associated with increases in the abundance of macroinvertebrates but taxonomic diversity was similar to other streams of differing land use. Community composition also changed in response to forest age, with streams in mature pine having taxa characteristic of higher water quality e.g., Ephemeroptera, Plecoptera and Coleoptera, and streams in recently harvested catchments having taxa characteristic of lower water quality e.g., Oligochaeta, Mollusca and Crustacea.

Invertebrate communities in streams draining mature exotic forest were similar to communities in streams draining native forest whereas stream communities in recently harvested forest were similar to those in pasture. However, community responses to forestry were different between the four regions. Two regions (Lismore and Te Wera) showed distinct differences in invertebrate community composition between mature exotic/native forest and pasture/harvested land use types, but Tawarau and Pirongia forests showed little or no difference in community composition. It appears then that recent forest harvesting does influence benthic invertebrate communities but regional influences e.g., geology and climate, are equally important in determining the extent of the response.

Keywords: community composition, forestry, harvesting, land use, macroinvertebrates, regions, spatial scale, water quality.

INTRODUCTION

Exotic forest harvesting creates major modifications to the environment, including the streams and rivers draining affected catchments. Harvesting is known to cause major disturbance to waterways through changes in water chemistry, light penetration, water temperature, sediment addition, organic matter and nutrient input, and hydrological regimes (Campbell and Doeg, 1989). A number of studies overseas have found forest harvesting causes changes in community composition, increasing abundance while decreasing diversity of macroinvertebrate communities (e.g., Campbell and Doeg, 1989; Growns and Davis, 1991; Ormerod et al., 1993; Davies and Nelson, 1994). However, similar studies in New Zealand have found that if such changes in communities do occur, they are generally short term (e.g., Graynoth, 1979; Winterbourn and Rounick, 1985; Winterbourn, 1986; Collier et al., 1989). Much of this work has focused on the effects of mature exotic forest however, not harvesting (e.g., Quinn et al., 1994; Harding and Winterbourn, 1995; Friberg and Winterbourn, 1997; Friberg et al., 1997; Quinn et al., 1997; Townsend et al., 1997).

Furthermore, regional effects such as geographic location (e.g., geology, altitude, climate) and watershed characteristics (e.g., topography, catchment vegetation) may be more influential to overall stream invertebrate community structure than land use per se (Biggs et al., 1990; Hildrew and Giller, 1994; Richards et al., 1996; Friberg et al., 1997; Harding et al., 1997; Richards et al., 1997). The response of stream communities to forestry practices is therefore likely to differ between regions within New Zealand, as they appear to between New Zealand and overseas.

The aim of my study is therefore two fold; firstly to examine whether the effects of exotic forestry on stream communities differs with forest age and secondly, whether or not this response differs between regions of New Zealand. To achieve this I examined the effects of forest age on stream water quality, macroinvertebrates and periphyton in four west coast forests of central North Island, New Zealand.

STUDY SITES

Forty two streams were sampled during January and February 1997, in or around four exotic forests, Lismore, Tawarau, Pirongia and Te Wera, on the west coast of central North Island, New Zealand (Fig. 2.1, Appendix 2.1). All streams were first to third order with catchments draining at least 90% *Pinus radiata,* 90% native forest or 90% pasture (Plate 2.1 and 2.2). To assess differences between streams in recently harvested exotic forest and mature exotic forest, sites were selected according to forest age. Catchments with forests harvested less than six years before sampling were categorised as "harvested", and those greater than 10 years old categorised as "mature" (Plate 2.1). All harvested streams were clearfelled to the stream edge with little (< 5 m) or no riparian strip.

Lismore forest, located near Wanganui, ranges in altitude from 80 to 350 m a.s.l. and has the lowest rainfall of the four forests (mean annual rainfall = 1000 mm (Kirkpatrick and Phillips, 1996)). Geology consists predominantly of marine sand, siltstone and limestone of Quaternary age (Department of Scientific and Industrial Research, 1984). Te Wera forest is located in the Taranaki region and has an altitude range of 150 to 374 m a.s.l. with similar geology to Lismore, but also some Miocene pumiceous and andesitic tuff. Rainfall is higher in this area averaging 1400 mm per year. Tawarau forest, located in the Waikato, has an altitude range of 200 to 300 m a.s.l. with similar geology to Te Wera, but also some Quaternary agglomerate and andesite. Mean annual rainfall is higher than Te Wera forest (2400 mm). Pirongia forest, also in the Waikato, north-east of Tawarau forest, has a distinctively different geology comprising andesite, agglomerate and breccia of Quaternary age. The altitude range of Pirongia forest is between 100 and 300 m a.s.l. and rainfall averages 2000 mm per year.

For Te Wera, Tawarau and Pirongia forests, native and pasture sites were within 1 km of their respective forests. Streams draining pasture in the Lismore area were similarly close, but native forest sites were located approximately 16 km away (to the furthest stream) and had a slightly different geology of mudstone, siltstone and conglomerate.

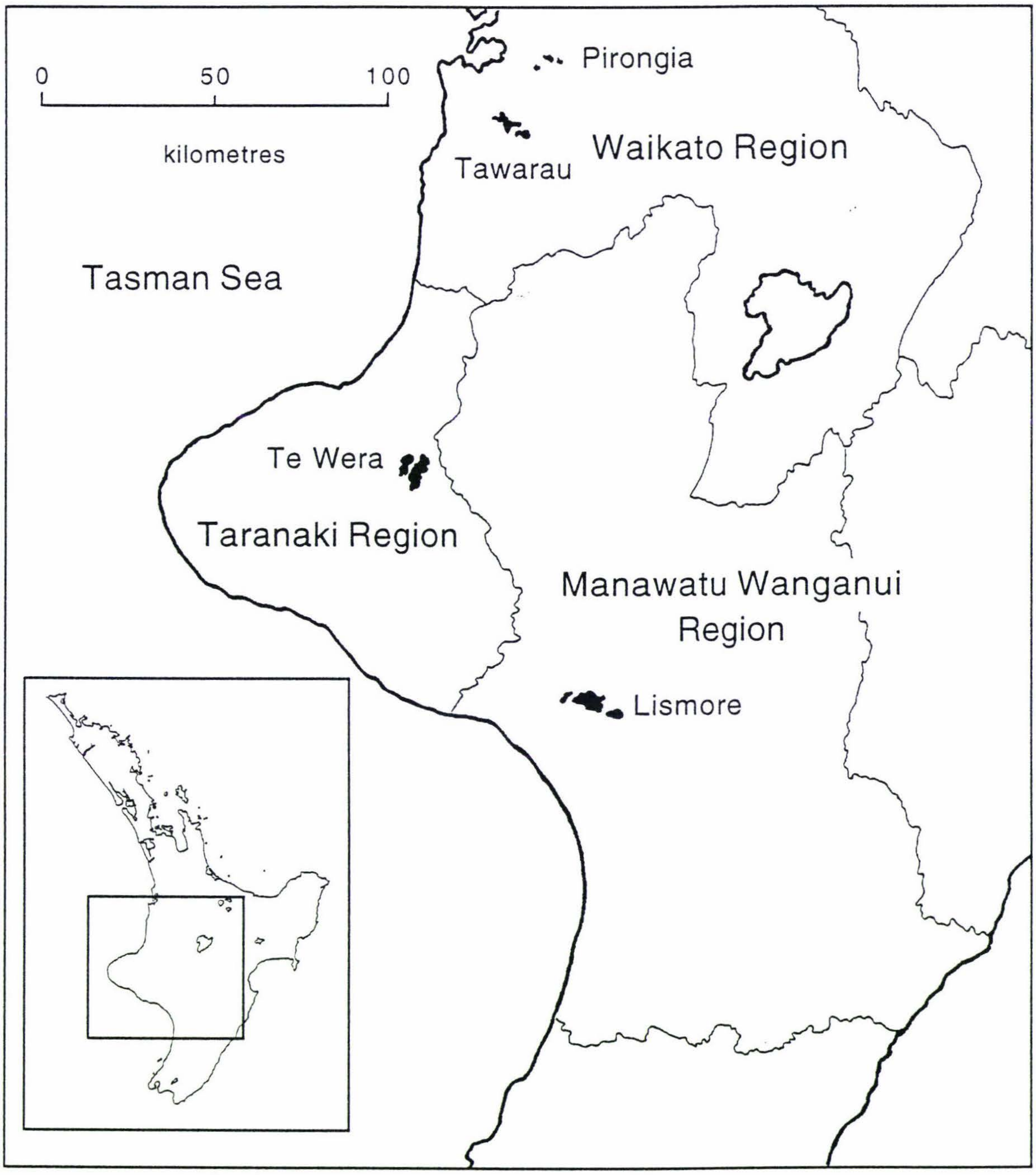

Figure 2.1. Location of the four exotic forests, Lismore, Tawarau, Pirongia and Te Wera, on the west coast of central North Island, within which 42 streams were sampled between January and February 1997, to assess effects of exotic forest harvesting on stream communities.

PLATE 2.1

Plate 2.1. A mature exotic forest site at Lismore (left) and harvested exotic forest site at Lismore (right) from a survey of 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997.

PLATE 2.2

Plate 2.2. A pasture site at Tawarau (left) and a native site at Te Wera (right) from a survey of 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997.

METHODS

Sampling protocol

At each site four samples were collected with a 250 μm-mesh kick net from a 0.1 m^2 quadrate delineated area, and stored in 70% alcohol until sorting. Samples were sieved through 500 μm mesh and macroinvertebrates sorted from particulate matter in a white tray, then identified and enumerated with a dissection microscope (×40). Macroinvertebrates were identified to the lowest possible taxonomic level with keys from McFarlane (1951), Winterbourn (1973), Cowley (1978), and Winterbourn and Gregson (1989). Those taxa that could not be identified to species level were separated into apparent morphospecies.

The remaining particulate matter was sieved into coarse particulate organic matter (CPOM) (> 1 mm) and medium particulate organic matter (MPOM) (250 μm - 1 mm). Samples were dried at 80 °C for five days, weighed and then combusted at 600 °C for two hours, and weighed again to obtain ash free dry weight (AFDW).

Four small stones (mean surface area = 33 cm^2) were collected concurrently with the macroinvertebrate sampling and frozen until pigment extraction in 90% acetone at 5 °C for 24 hours. Absorbency was read with a Jenway 6105 UV/Visible Spectrophotometer at 410, 430, 665 and 750 nm. Pigment concentration was calculated following Moss (1967a and b) and adjusted for surface area following Graham et al. (1988).

Conductivity and dissolved oxygen for each site were measured in the field using an ORION model 122 conductivity meter and a YSI model 58 dissolved oxygen meter respectively. Stream stability of each site was assessed using the Pfankuch Stability index (Pfankuch, 1975). Substrate composition (silt, sand (< 0.2 cm diameter), gravel (0.2-6.0 cm), small cobbles (6.1-12.0 cm), large cobbles (12.1-26.0 cm), boulders (> 26 cm) and bedrock), the percentage channel cover and type of riparian vegetation (native forest, exotic woodland, scrub, crop/pasture, other) were assessed visually. Water samples were taken back to the laboratory for pH (using a ORION model 250A pH meter) and suspended solids analysis. The latter was determined by the weight of solids collected after filtering a sample through a Whatman GF/C filter (pore size = 0.7 μm) and drying overnight at 105 °C. Width (four measurements), depth (five measurements midstream), and current velocity (five measurements midstream using a velocity head rod) were also recorded at each site.

Diversity indices

To examine differences in diversity of macroinvertebrate communities between regions and land use types, three diversity indices were calculated for each site. Margelef's index (Clifford and Stephenson, 1975) is a measure of taxonomic richness given by:

$$D = \frac{(S - 1)}{\ln N}$$

where N = the total number of individuals collected and S = the number of taxa. The Berger-Parker dominance index (Berger and Parker, 1970) and the Simpson's index (Simpson, 1949) are both measures of evenness (or dominance). The Berger-Parker index is given by:

$$D = \frac{N_{max}}{N}$$

where N_{max} = the number of individuals in the most abundant taxon, and N = the total number of individuals collected. Simpson's index is given by:

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

where n_i = the number of individuals in the i^{th} taxon and N = the total number of individuals. An increase in the Margelef's index value indicates an increase in diversity, whereas an increase in the Berger-Parker or Simpson's indices indicates a decrease in diversity (Death and Winterbourn, 1995).

Biotic indices

To examine differences in water quality between region and land use type, the Macroinvertebrate Community Index (MCI), Quantitative Macroinvertebrate Community Index (QMCI) and the Percentage of Ephemeroptera, Plecoptera and Tricoptera (% EPT) were calculated. The MCI (Stark, 1985) is an index based on the presence of macroinvertebrate taxa which have previously been allocated a score (1 to 10) based on their pollution tolerances (1 = highly tolerant, 10 = highly sensitive). MCI values are calculated from:

$$MCI = \frac{\text{site score}}{\text{N}} \times 20$$

where the site score = the sum of individual taxon scores for all taxa present in a sample and N = the number of scoring taxa. Scores greater than 120 may be regarded as pristine and scores less than 80 are severely polluted. The QMCI is a quantitative variant of the MCI (Stark, 1993) and is calculated from:

$$QMCI = \sum_{i=1}^{i=S} \frac{(\text{n}_i \times \text{a}_i)}{\text{N}}$$

where S = the total number of taxa in the sample, n_i = the number of individuals in the i^{th} scoring taxon, a_i = the score for the i^{th} taxon and N = the total number of individuals collected. The % EPT (Lenat, 1988) is calculated from:

$$\%EPT = \frac{(\text{E} + \text{P} + \text{T})}{\text{N}}$$

where E = the number of Ephemeroptera, P = the number of Plecoptera, T = the number of Tricoptera and N = the total number of individuals. Ephemeroptera, Plecoptera and Tricoptera characteristically consist of taxa that prefer pristine conditions, hence a high index value indicates more pristine conditions.

Statistical analyses

Statistical differences between treatments (region, land use type and stream) were analysed using a three-way Analysis of Variance (ANOVA) and Duncan multiple range a posteriori means test (P=0.05) with SAS (SAS, 1989). Region and land use type were treated as fixed effects and streams within each group as random. Data were log transformed ($\log_{10}(x+1)$) before analysis if necessary, to increase variance homogeneity.

Ordination of site communities (log transformed data) was also carried out with Detrended Correspondence Analysis (DECORANA) using the PC-ORD statistical package (McCune and Mefford, 1995) and environmental variables correlated with the DECORANA axes using Pearson's correlation coefficient.

RESULTS

Physiochemical characteristics

Streams were similar in most physical characteristics and water chemistry between regions and land use type (Table 2.1, Appendix 2.2). Streams in harvested exotic forest however, tended to have slower velocity, less riparian cover ($F_{3,23}$=20.16, P<0.001) (and therefore higher water temperatures), increased sediment deposition ($F_{3,23}$=3.52, P=0.03) and decreased stability, compared to streams in mature exotic forest. Streams draining pasture had similar characteristics to those in harvested exotic forest although water velocity was greater at pasture sites. Streams draining native forest were similar to those in mature exotic forest with both having lower water temperatures, a high percentage of riparian cover and greater stability than pasture or harvested exotic forest sites.

There were also differences between regions. Lismore forest streams had higher amounts of silt ($F_{3,23}$=7.76, P=0.001), pH and conductivity compared to other regions (Table 2.1). Pirongia had a greater proportion of cobbles and boulders, and Tawarau and Te Wera were most similar, being dominated by sand and cobble substrate.

Table 2.1. Mean physiochemical characteristics (with ranges in parentheses) for land use types in each region for 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997. *F* ratios and *P* values from a three way ANOVA testing the null hypothesis that physiochemical characteristics do not differ between land use type and regions are also given, degrees of freedom in all cases is 3, 23. Cond = conductivity, DO = dissolved oxygen, Temp = temperature and SS = suspended solids.

		No. of streams	Width (m)	Depth (cm)	Velocity (m/s)	Cond (μS/cm)	DO (mg/l)	Temp (°C)	Stability	pH	SS (mg/l)
Lismore	**Mature**	2	1.8 (1.4-2.2)	3.5 (1.1-5.8)	0.29 (0.28-0.31)	578 (491-664)	10.4 (9.3-11.4)	15.1 (11.7-18.4)	118 (105-130)	7.15 (7.10-7.20)	0.37 (0.09-0.64)
	Harvested	3	1.1 (0.5-2.0)	11.9 (9.2-16.4)	0.20 (0.09-0.35)	530 (359-661)	6.9 (2.9-9.1)	16.2 (15.2-16.8)	102 (83-133)	7.38 (7.09-7.83)	0.14 (0.07-0.25)
	Pasture	3	1.6 (0.5-2.2)	17.6 (5.3-31.8)	0.47 (0.18-0.75)	398 (322-479)	9.5 (9.2-9.5)	16.6 (15.8-17.4)	98 (76-120)	7.51 (7.20-7.78)	0.08 (0.07-0.09)
	Native	3	1.0 (0.3-1.8)	12.0 (5.1-20.4)	0.25 (0.20-0.37)	263 (247-272)	10.4 (10.1-11.1)	14.0 (13.2-14.5)	70 (64-82)	7.78 (7.62-7.94)	0.13 (0.02-0.30)
Tawarau	**Mature**	5	1.1 (0.02-1.6)	20.8 (6.2-52.7)	0.31 (0.25-0.36)	139 (101-191)	9.9 (9.4-10.6)	12.0 (11.4-12.5)	92 (62-114)	7.07 (6.84-7.23)	0.06 (0.03-0.09)
	Harvested	1	1.2	15.5	0.18	107	9.3	13.0	141	7.21	0.02
	Pasture	2	1.6 (1.5-1.7)	33.7 (32.2-35.1)	0.32 (0.318-0.319)	106 (90-122)	9.2 (8.7-9.6)	14.6 (14.3-14.8)	84 (77-91)	6.74 (6.66-6.81)	0.07 (0.04-0.10)
	Native	2	1.5 (1.5-1.6)	17.3 (16.1-18.4)	0.35 (0.33-0.37)	158 (156-159)	9.9 (9.4-10.4)	12.3	100 (84-115)	7.22 (7.19-7.24)	0.05 (0.04-0.05)
Pirongia	**Mature**	3	1.4 (1.3-1.6)	13.1 (10.4-16.6)	0.53 (0.45-0.62)	92 (68-128)	9.5 (9.2-9.7)	13.6 (12.4-15.0)	68 (63-74)	7.00 (6.90-7.10)	0.03 (0.02-0.05)
	Pasture	1	2.8	14.9	0.65	401	9.7	19.9	95	7.21	0.02
	Native	1	0.9	6.6	0.20	188	10.1	12.3	72	7.21	0.06
Te Wera	**Mature**	5	1.0 (0.4-1.9)	17.0 (6.9-33.6)	0.40 (0.17-0.65)	112 (93-147)	7.9 (8.1-9.8)	14.4 (13.3-15.2)	93 (76-123)	7.01 (6.82-7.24)	0.21 (0.03-0.39)
	Harvested	5	1.3 (0.4-2.0)	17.6 (3.1-29.3)	0.21 (0.16-0.29)	111 (66-134)	7.0 (5.0-9.7)	16.2 (14.2-17.7)	103 (87-114)	7.36 (6.70-8.72)	0.61 (0.01-2.83)
	Pasture	3	1.1 (0.8-1.5)	31.3 (18.6-44.5)	0.57 (0.27-0.99)	92 (88-97)	8.1 (7.5-8.7)	16.2 (15.5-17.2)	108 (101-118)	6.77 (6.61-7.02)	0.03 (0.01-0.05)
	Native	3	0.7 (0.4-0.9)	15.2 (8.0-24.9)	0.30 (0.13-0.58)	72 (53-88)	8.6 (7.3-10.3)	13.5 (13.2-13.7)	73 (61-85)	7.02 (6.85-7.22)	0.10 (0.04-0.17)
Land use type		***F* ratio**	2.12	1.74	4.27	2.10	0.52	14.92	3.39	1.69	0.42
		***P* value**	0.13	0.19	0.02	0.13	0.67	<0.001	0.04	0.20	0.74
Region		***F* ratio**	1.99	2.55	0.53	92.37	1.91	6.66	1.08	3.99	0.32
		***P* value**	0.14	0.08	0.67	0.001	0.16	0.002	0.38	0.02	0.81

There were also interactions between land use type and region for some physiochemical characteristics. For example, in Lismore, silt was higher in streams draining mature and harvested exotic forest, whereas in Tawarau, streams in harvested exotic forest had higher silt and in Te Wera, pasture sites had higher silt ($F_{8,23}$=8.09, P<0.001). Pirongia forest however, had highest silt in the stream draining native forest. Conductivity did not differ between land use types but exhibited an interaction between land use type and region ($F_{8,23}$=8.03, P<0.001) e.g., in Te Wera and Lismore forests, streams in mature and harvested exotic forest had higher conductivity whereas in Pirongia, the stream draining pasture had the highest (Table 2.1).

Particulate organic matter (POM)

Mean total POM ranged from 0.45 to 13.93 g AFDW/0.1 m^2 (Appendix 2.3) and was not affected by land use type ($F_{3,27}$=0.19, P=0.90; $F_{3,27}$=1.27, P=0.30; and $F_{3,27}$=0.78, P=0.52, for total POM, CPOM and MPOM respectively). Regional differences were significant for total POM ($F_{3,27}$=9.54, P<0.001), CPOM ($F_{3,27}$=8.73, P<0.001) and MPOM ($F_{3,27}$=7.33, P=0.001) with highest levels in Tawarau forest and lowest levels in Lismore.

Periphyton

Algal biomass (measured as chlorophyll *a* and phaeophytin concentration) on stones did not show a significant difference between land use types ($F_{3,149}$=2.26, P=0.08), although streams draining native forest tended to have lower periphyton biomass than other land use types (Fig. 2.2a). There was a significant difference between regions ($F_{3,149}$=5.40, P=0.002), with the lowest biomass in Te Wera and the highest in Tawarau and Lismore (Fig. 2.2b). There was also a significant interaction between region and land use type ($F_{8,149}$=4.78, P<0.001) (Fig. 2.2c), with streams in mature exotic forest having highest biomass in Tawarau, Pirongia and Te Wera forests, and streams in pasture and harvested exotic forest catchments having highest biomass in Lismore.

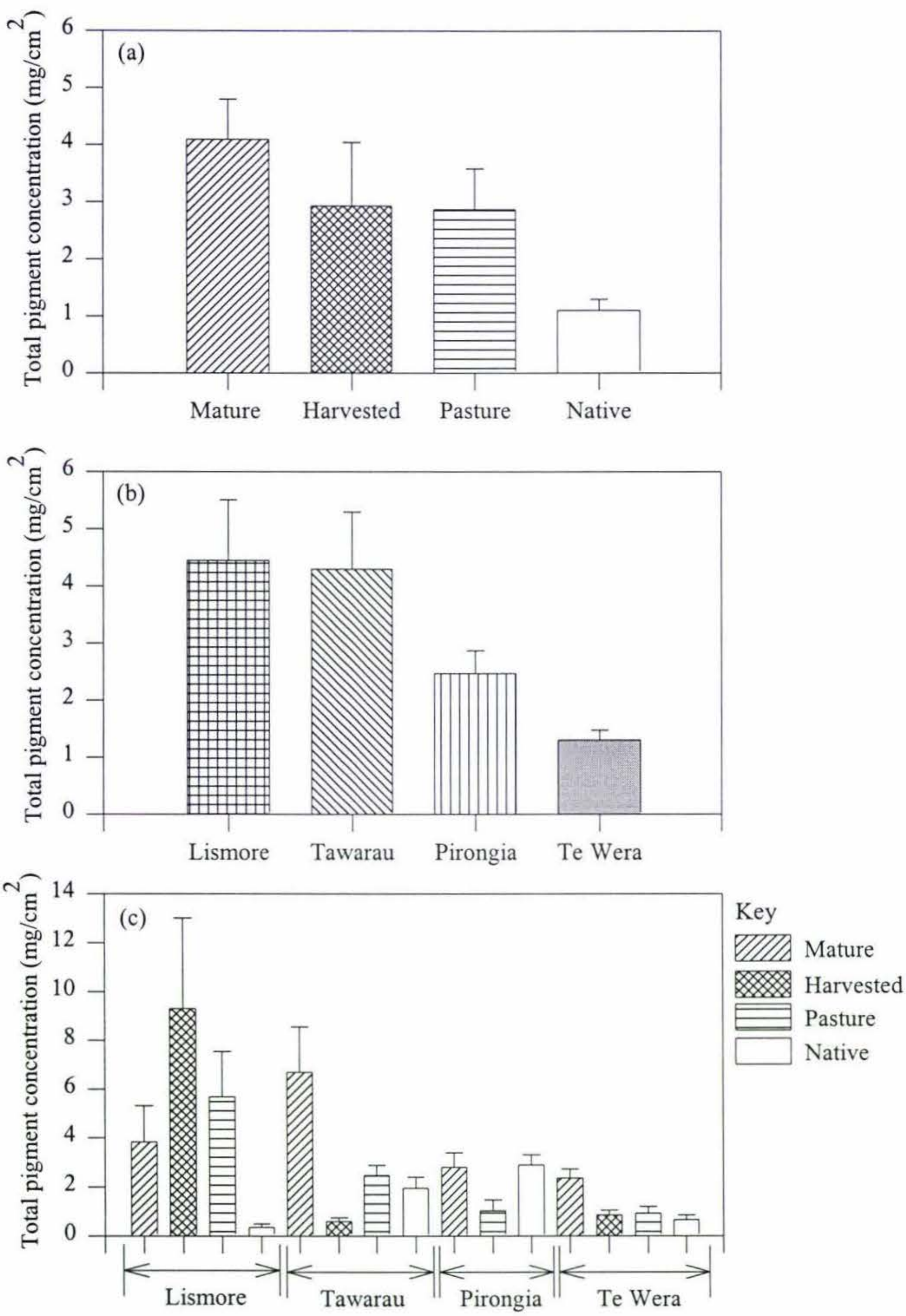

Figure 2.2. Mean total pigment (chlorophyll *a* and phaeophytin) concentrations (± 1 SE) for (a) land use type, (b) region and (c) land use types within each region, on stones collected at 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997.

Macroinvertebrates

Total number of individuals

The total number of individuals collected from any one site (four replicates combined) ranged between 12 and 7297. Lismore forest had the highest mean number of individuals overall and Tawarau forest the lowest ($F_{3,149}$=10.63, P<0.001) (Fig. 2.3b). Streams draining pasture had greater numbers of individuals than streams in all other land use types ($F_{3,149}$=19.31, P<0.001) (Fig. 2.3a). Streams in harvested exotic forest had lower numbers of individuals than those in pasture but higher numbers than streams in mature exotic and native forest, which were not significantly different. Again however, there was a significant interaction between region and land use type ($F_{8,149}$=13.87, P<0.001) reflecting a different response to land use between regions (Fig. 2.3c).

Total number of taxa

A total of 167 macroinvertebrate taxa were collected from the 42 sites. Of these, 12 taxa were Ephemeroptera, 12 Plecoptera, 35 Tricoptera, 40 Diptera, 16 Coleoptera, 9 other insect taxa and 43 were non-insects. The maximum number of taxa collected from any one site (four replicates combined) was 41 (TaP1) and the minimum 5 (LF1). Overall, the Pirongia region had the highest number of taxa, followed by Te Wera and Tawarau, with Lismore having the least ($F_{3,149}$=25.68, P<0.001) (Fig. 2.4b).

Land use effects on taxonomic richness were also significant ($F_{3,149}$=6.00, P<0.001) with the greatest difference occurring between pasture sites (highest number of taxa) and native sites (lowest number of taxa). Streams draining mature and harvested exotic forest were not significantly different from each other (Fig. 2.4a). However, the effect of land use type on taxonomic richness differed between regions ($F_{8,149}$=9.82, P<0.001) e.g., in the Lismore region, streams in mature exotic forest had very low mean taxa numbers (2 taxa) compared with harvested (13 taxa), pasture (11 taxa) and native (9.5 taxa) site groups (Fig. 2.4c). But in the Tawarau region, harvested exotic and native

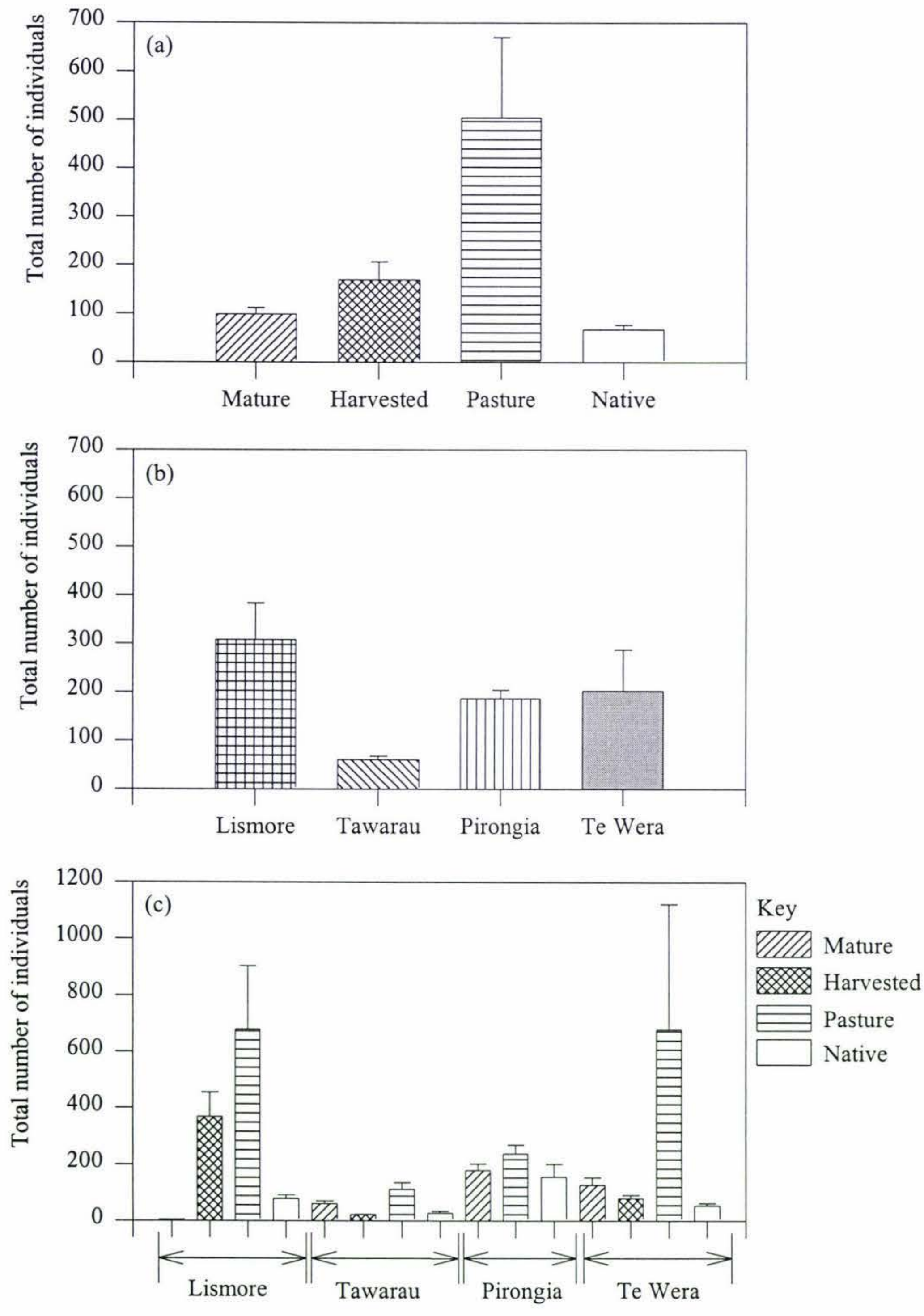

Figure 2.3. Mean total number of individuals (± 1 SE) for (a) land use type, (b) region and (c) land use types within each region, for macroinvertebrates collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997.

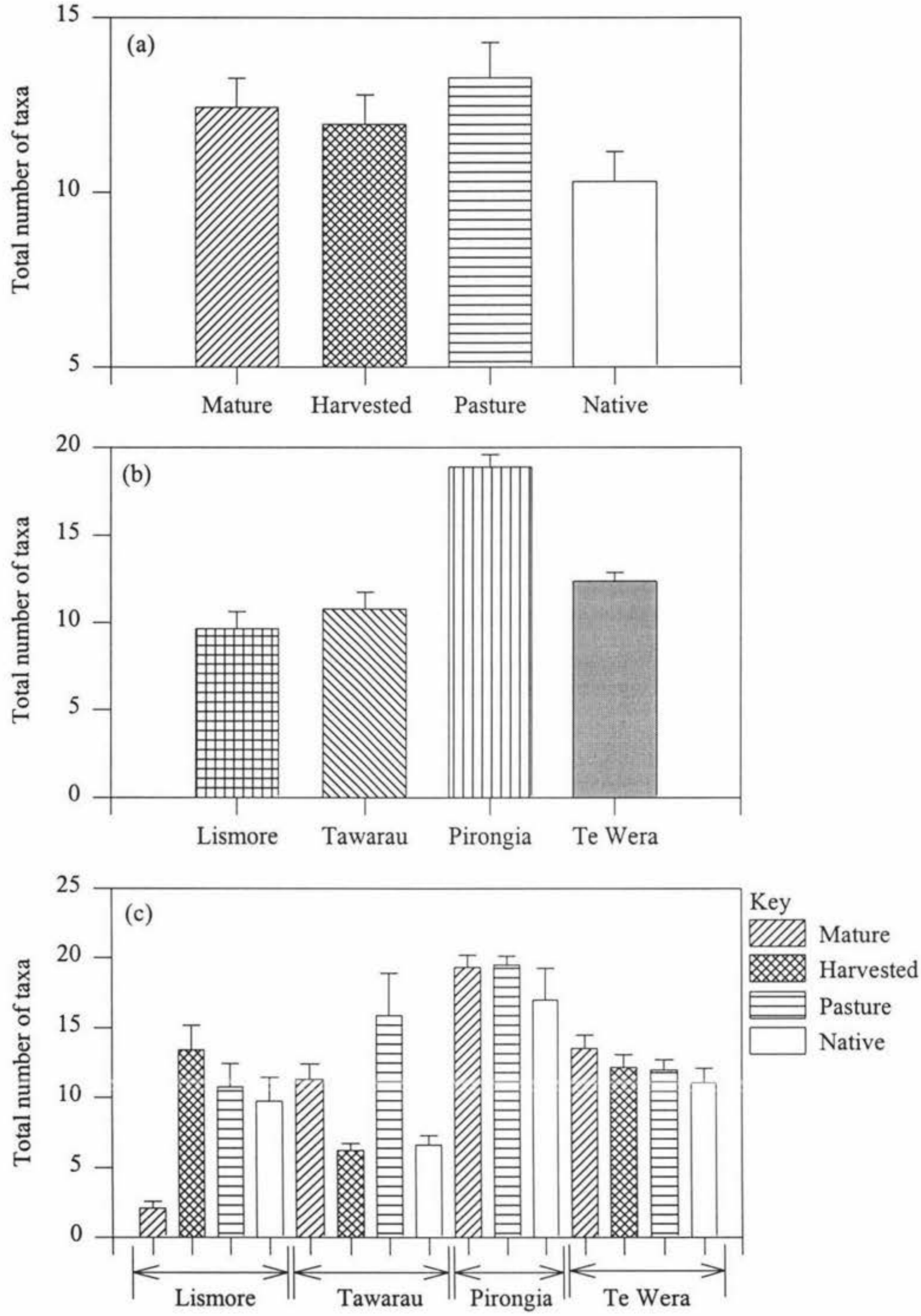

Figure 2.4. Mean total number of taxa (± 1 SE) for (a) land use type, (b) region and (c) land use types within each region, for macroinvertebrates collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997.

forest site groups had lower numbers of taxa compared to mature exotic forest and pasture site groups.

Diversity indices

There were no significant differences in the three diversity measures between land use types ($F_{3,149}$=0.52, P=0.67; $F_{3,149}$=0.93, P=0.43; $F_{3,149}$=1.96, P=0.12, for Margelef's, Berger-Parker and Simpson's indices respectively). But differences in taxonomic diversity were found between regions. Margelef's index ($F_{3,149}$=16.26, P<0.001) had greatest values in Pirongia and lowest values in Lismore. The Berger-Parker dominance index and Simpson's index both revealed that streams in Lismore forest were more heavily dominated by Amphipoda, *Potamopyrgus antipodarum*, and in native streams by *Zephlebia dentata*, than other regions ($F_{3,149}$=7.56, P<0.001 and $F_{3,149}$=7.29, P<0.001 respectively). Again there is a significant interaction between region and land use type for both the Margelef's and Simpson's indices ($F_{8,149}$=2.98, P=0.004 and $F_{8,149}$=2.80, P=0.01 respectively) (Fig. 2.5).

Relative abundance

Streams in mature exotic and native forest had greater numbers of Ephemeroptera ($F_{3,27}$=17.21, P<0.001) (Fig. 2.6a), whereas streams draining harvested exotic forest and pasture had greater numbers of Crustacea ($F_{3,27}$=4.12, P=0.02). Streams in pasture also had a higher abundance of Mollusca than all other land use types ($F_{3,27}$=4.22, P=0.01).

Of the regions, streams in Lismore were characterised by the highest abundance of Mollusca ($F_{3,27}$=3.77, P=0.02) but the lowest abundance of Ephemeroptera ($F_{3,27}$=6.06, P=0.003) (Fig. 2.6b). Tawarau streams also had high numbers of Mollusca, as well as Diptera ($F_{3,27}$=6.99, P=0.001) and Plecoptera ($F_{3,27}$=6.00, P=0.003). Pirongia streams had high numbers of Ephemeroptera and Coleoptera ($F_{3,27}$=5.56, P=0.004), and Te Wera streams had similar relative abundances of Diptera to that of Tawarau. An interaction between region and land use type occurred for Ephemeroptera ($F_{8,27}$=3.72, P=0.05) e.g.,

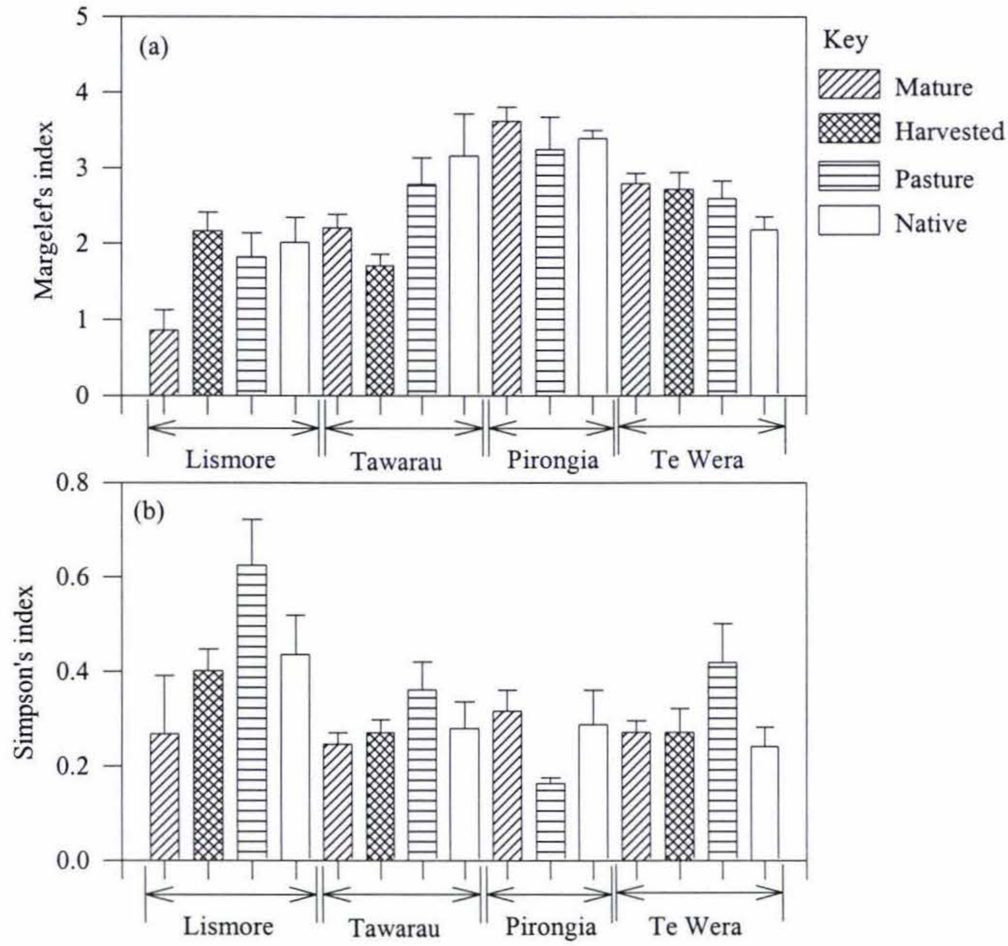

Figure 2.5. Mean Margelef's (a) and Simpson's (b) indices (± 1 SE) for land use types within each region, for macroinvertebrates collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled during January and February 1997.

Lismore streams draining native forest were dominated by Ephemeroptera, whereas streams in the other three land use types in Lismore comprised mainly Diptera, Mollusca and others. All other regions had more Ephemeroptera in streams draining mature exotic and native forest than those draining harvested exotic forest and pasture.

Biotic indices

For all indices, streams in mature exotic and native forest had significantly higher scores than streams in harvested exotic forest and pasture ($F_{3,35}$=16.43, P<0.001; $F_{3,35}$=6.66,

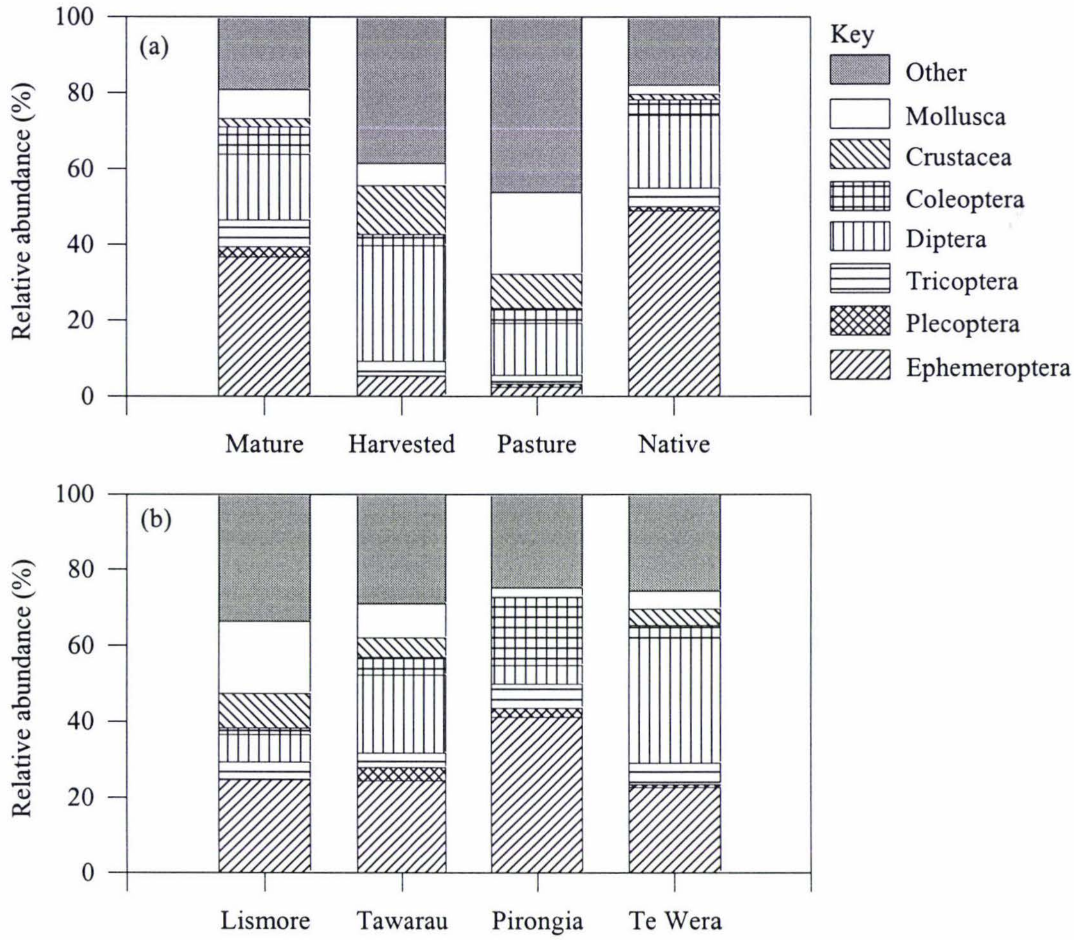

Figure 2.6. Mean relative abundance of the main taxonomic orders for (a) land use type and (b) region, for macroinvertebrates collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997.

P=0.001; $F_{3,27}$=12.74, P<0.001, for MCI, QMCI and % EPT respectively) (Fig. 2.7a). Significant differences between regions occurred for the MCI only ($F_{3,35}$=3.75, P=0.02), with streams in Tawarau and Pirongia regions having higher MCI scores (Fig. 2.7b).

Ordination

Initial ordination of sites separated one Tawarau pasture site (TaP2) from all other sites. This site was removed for subsequent analysis.

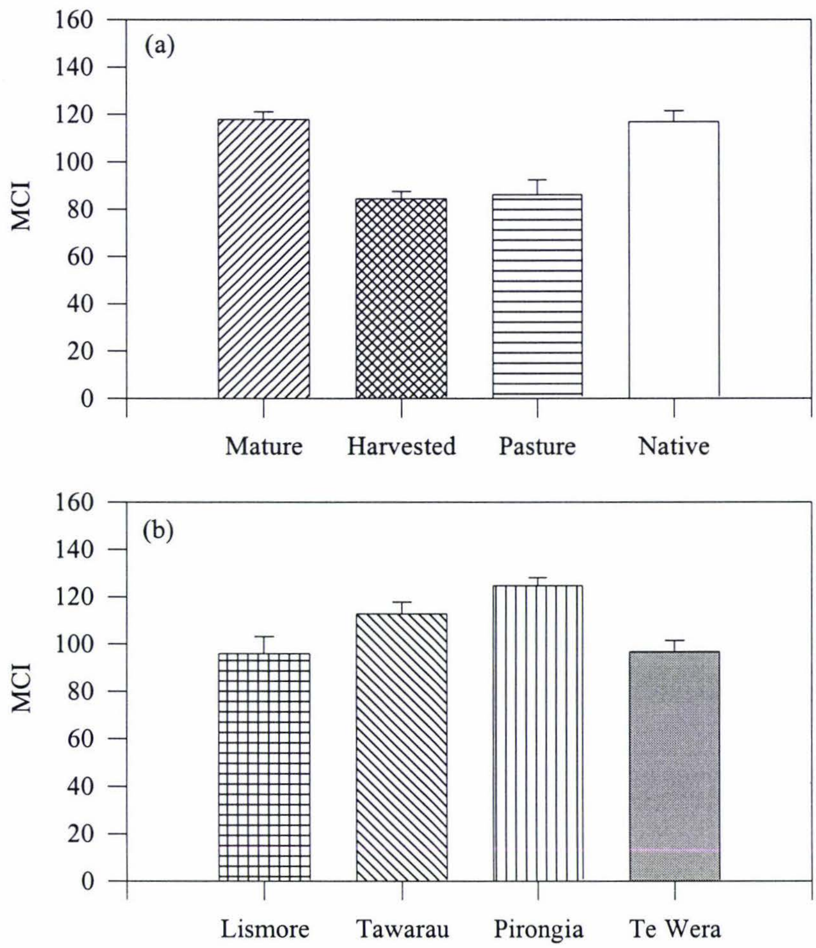

Figure 2.7. Mean Macroinvertebrate Community Index (MCI) (± 1 SE) for (a) land use type and (b) region, for macroinvertebrates collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997.

Distinctly different communities were not apparent in a plot of DECORANA scores for axis one and two (Fig. 2.8), but axis one generally graded sites from those in harvested exotic forest and pasture catchments (left of axis one) to those in native and mature exotic forest. The only exceptions were two streams draining pasture (in Tawarau and Pirongia), which occurred with the native and mature exotic forest sites. Axis one accounted for 38.6%, and axis two, 16.8%, of the variation in the data.

There were similarly no distinct regional groups of communities (Fig. 2.8), but rather a differential effect of land use type between regions as indicated in the univariate analyses. For example, streams in Lismore and Te Wera exhibited distinct communities

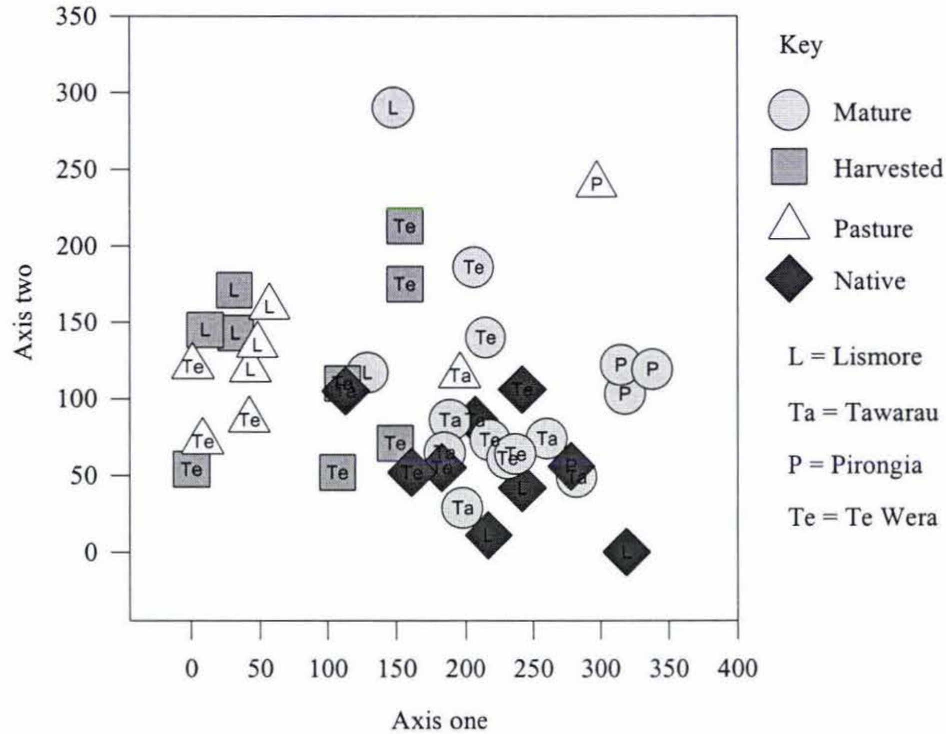

Figure 2.8. Axis one of a Detrended Correspondence Analysis as a function of axis two for macroinvertebrate communities collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled during January and February 1997.

in relation to land use, whereas in Pirongia, all streams were situated together to the right of axis one, with a weaker trend along axis two related to land use. In Tawarau, streams showed no differences in community structure between land use types.

Axis one correlated with a number of site characteristics (Table 2.2) indicating that communities to the left of axis one were those at sites with high to medium amounts of silt, high temperatures and greater instability. MCI, QMCI and % EPT were all positively correlated with axis one, suggesting that macroinvertebrate communities to the right of axis one were those with taxa characteristic of moderate to good water quality. This was also reflected in the taxa correlated with the DECORANA axes (Table 2.3), with most taxa positively correlated with axis one, taxa characteristic of pristine conditions e.g., *Deleatidium* sp., *Zephlebia dentata*, *Aphrophila neozelandica* and *Archichauliodes diversus*; and those negatively correlated, taxa characteristic of lower water quality e.g., Oligochaeta and *Potamopyrgus antipodarum*. The years since

Table 2.2. Variables correlated (* = $P<0.05$) with Detrended Correspondence Analysis axes for macroinvertebrate communities collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997.

Parameter	Axis 1	Axis 2	Axis 3
% boulders	0.36 *	-0.43 *	-0.25
% cover	0.35 *	-0.22	-0.50 *
% EPT	0.78 *	-0.55 *	-0.30
% gravel	0.38 *	-0.03	0.12
% native	0.28	-0.46*	-0.27
% pasture	-0.42 *	0.34 *	0.32 *
% sand	-0.33 *	0.20	0.20
% silt	-0.54 *	0.35 *	-0.11
% small cobbles	0.42 *	-0.30	-0.01
Altitude	0.27	-0.36*	0.03
Bed stability	-0.45 *	0.45 *	0.14
Conductivity	-0.32 *	0.49 *	-0.20
Depth	-0.17	-0.20	0.34 *
Dissolved oxygen	0.38 *	0.05	-0.17
MCI	0.87 *	-0.41 *	-0.31 *
QMCI	0.85 *	-0.46 *	-0.32 *
Stability	-0.47 *	0.44 *	0.15
Temperature	-0.43 *	0.65 *	0.37 *
Velocity	0.11	0.15	0.35 *
Width	-0.04	0.43 *	0.30
Years since harvesting	0.41 *	-0.46 *	-0.32 *

Table 2.3. Taxa correlated (* = $P<0.05$) with Detrended Correspondence Analysis axes for macroinvertebrate communities collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled between January and February 1997.

Taxon	Axis 1	Axis 2	Axis 3
Acari sp. 1	-0.47 *	-0.07	-0.07
Acari sp. 3	-0.47 *	0.13	-0.27
Amphipoda	-0.46 *	0.10	-0.47 *
Aphrophila neozelandica	0.48 *	0.07	0.07
Archichauliodes diversus	0.53 *	0.08	0.06
Austroclima sepia	0.50 *	-0.22	0.05
Chironomid sp. 1	-0.03	0.49 *	-0.04
Chironominae	-0.48 *	-0.21	0.59 *
Coloburiscus humeralis	0.56 *	-0.12	-0.04
Deleatidium sp.	0.77 *	-0.10	-0.08
Elmidae	0.63 *	0.10	-0.04
Helicopsyche sp.	0.53 *	-0.28	-0.08
Hydrobiosis parumbripennis	0.40 *	0.41 *	0.09
Medium Oligochaetes	-0.65 *	-0.08	0.34 *
Orthocladiinae	-0.09	0.42 *	0.07
Orthopsyche fimbriata	0.54 *	-0.14	-0.07
Ostracod sp. 1	-0.47 *	0.08	0.02
Paralimnophila skusei	-0.44 *	0.28	-0.13
Paramephrops planiforms	0.17	-0.42 *	-0.36 *
Physa sp.	-0.41 *	0.20	-0.22
Pisidium casertanum	-0.48 *	0.04	0.15
Polyplectropus sp.	-0.52 *	-0.10	0.42 *
Potamopyrgus antipodarum	-0.58 *	0.2	0.17
Pycnocentria funerea	0.42 *	-0.23	0.01
Sigara sp.	-0.53 *	0.11	0.33 *
Small Oligochaetes	-0.72 *	0.20	0.19
Zelandoperla decorata	0.44 *	0.21	0.02
Zephlebia dentata	0.50 *	-0.65 *	-0.44 *

harvesting (set at 100 years for native forest and one year for pasture) was also strongly correlated with the axes and may explain why there was no distinct separation between exotic forest types arbitrarily assigned as harvested (less than six years) or mature (greater than 10 years).

DISCUSSION

Increased light and temperature as a result of deforestation have been known to cause increases in periphyton biomass in streams (Biggs et al., 1990; Holopainen and Huttunen, 1992; Quinn et al., 1994). But in this study, differences in periphyton biomass between open sites (harvested exotic forest and pasture) and shaded sites (native and mature exotic forest) were not highly significant, with only streams in native forest exhibiting significantly lower periphyton levels. Although periphyton levels were still relatively high in streams draining other land use types, some other factor, such as nutrients or sedimentation, may be limiting periphyton biomass in streams draining harvested exotic forest and pasture. Nutrients (especially nitrogen) are the limiting factor to periphyton biomass in many New Zealand streams (Biggs, 1995), and although forest harvesting and agricultural development are known to increase nutrient concentrations (Graynoth, 1979; Mosley and Rowe, 1981; Winterbourn, 1986), they may still be too low to boost periphyton levels. Conductivity, a surrogate of nutrient levels (Mosley and Rowe, 1981; Biggs, 1995), certainly did not differ between land use types in this study and was moderate to low in Tawarau, Pirongia and Te Wera, suggesting possible nutrient limitation. However, conductivity was extremely high in streams from the Lismore region, and may allow for greater periphyton biomass in those streams draining pasture and harvested exotic forest.

Sediment deposition and substrate instability were high in streams draining both harvested exotic forest and pasture, and this may keep periphyton levels below those at mature exotic forest sites (Harding and Winterbourn, 1995; Friberg et al., 1997). There were also high abundances of grazers at harvested exotic forest and pasture sites, which have been shown elsewhere to be important regulators of periphyton biomass (Steinman, 1992; Botts, 1993; Allan, 1995). Therefore, an interaction between these factors (i.e., light, nutrients, grazers and substrate) may be determining periphyton biomass between land use types and overriding expected increases in biomass due to forest harvesting.

Several aspects of the stream macroinvertebrate communities were affected by the harvesting of surrounding exotic forest. Numbers of individuals were higher in streams at harvested compared to mature exotic forest sites. This increase in abundance could be a result of increased algal biomass due to changes in shading and water temperature (Behmer and Hawkins, 1986). Although periphyton biomass was lower in streams draining harvested exotic forest than in streams draining mature exotic forest, these high numbers of invertebrates may be moderating increases in periphyton biomass due to grazing pressure (Allan, 1995).

Diversity was not affected by forest age or type in this study. Similar findings have been found elsewhere (Winterbourn and Rounick, 1985; Carlson et al., 1990; Growns and Davis, 1991) but not in all studies (Graynoth, 1979; Newbold et al., 1980; Murphy and Hall, 1981; Ormerod et al., 1993). However, macroinvertebrate community composition did differ between land use types. Generally, Ephemeroptera, Mollusca and Crustacea were most responsive to forest age or land use type. Ephemeroptera were less common in streams draining harvested exotic forest and pasture catchments where Crustacea and Mollusca occurred in higher abundances. The MCI, QMCI and % EPT all indicated a similar situation; that pollution sensitive groups decreased in streams draining harvested exotic forest and pasture (Quinn and Hickey, 1990b; Quinn et al., 1994; Collier, 1995; Harding and Winterbourn, 1995).

The ordination failed to show differences in community structure between streams draining mature exotic and those draining native forest. Friberg et al. (1997) also found that macroinvertebrates appear indifferent to forest type and this is consistent with the perception that New Zealand stream macroinvertebrates are habitat generalists (Winterbourn and Rounick, 1985; Parkyn and Winterbourn, 1997) i.e., the type of riparian vegetation has less effect in determining community structure, via its influences on habitat and food quantity and quality, if physical influences are similar (e.g., shading, substrate characteristics and stream bed stability). However, harvesting does affect invertebrate communities initially, causing shifts in community structure comparable to those communities in streams draining pasture, communities consisting predominantly of pollution tolerant taxa such as *Potamopyrgus antipodarum*, Amphipoda and Ostracoda. The results of the present and other New Zealand studies (Graynoth, 1979; Winterbourn, 1986; Collier et al., 1989; Harding and Winterbourn, 1995; Friberg et al., 1997) indicate that benthic invertebrate communities comparable to those in native forest are re-established following reforestation with exotic conifers. This study indicates that this is a gradual reversion however, correlated with the recovery of riparian vegetation.

Several variables may be responsible for these differences in community structure between sites with open and those with closed canopy. Percentage of riparian cover, light and thermal regimes, substrate characteristics (especially the amount of fine sediments), stream stability and years since harvesting were all correlated with trends in community structure. Removal of riparian cover may be an important factor as it influences light and water temperature regimes (Campbell and Doeg, 1989), which in turn may influence periphyton biomass if other factors are not limiting (i.e., nutrients or sedimentation) (Quinn et al., 1997). This may result in an increase in taxa that depend on autochthonous in-stream production, in contrast to communities in forested streams that rely more on allochthonous inputs as food (Winterbourn and Rounick, 1985). However, forest harvesting also increases the amount of woody debri which may also be important to community structure (Quinn et al., 1997), and may account for some of the increase in numbers of invertebrates in streams with harvested catchments (Noel et al., 1986). Sedimentation and lower stream bed stability are usually associated with the absence of EPT taxa, reducing suitable habitat and food resources for such taxa (Waters, 1995). A combination of sedimentation and removal of riparian vegetation may therefore result in a community of pollution tolerant taxa.

Although there were often no clear differences in stream communities between land use types when all regions were considered concurrently, there were frequently marked differences between land use types within regions. Several aspects of community structure exhibited interactions between region and land use type e.g., number of taxa, number of individuals, relative abundance of Ephemeroptera, Margelef's and Simpson's indices. Furthermore, in the ordination communities were not distinctly different between land use type or region but within a region, showed stronger responses toward land use type e.g., in Lismore and Te Wera. Periphyton also showed a significant interaction between region and land use type.

The important underlying influences between regions are geology, topography and climate (Biggs and Gerbeaux, 1993; Allan et al., 1997; Richards et al., 1997) and it is likely these influences are acting from several spatial scales. Macro-scale features (e.g., climate and geology) should determine long term features of the community operating over a period greater than one year, whereas micro-scale features (e.g., nutrient and hydraulic regimes) should control short term features of communities operating over a period of less than one year (Biggs and Gerbeaux, 1993). However, micro-scale features are often conditional on macro-scale features, and communities may reflect interactions between fixed (e.g., geology, hydrology) and management influenced (e.g., land use) landscape features (Richards et al., 1997). The stream substrate has been identified as an important influence on invertebrate community structure (Minshall, 1984) and changes in substrate are also an important effect of many anthropogenic disturbances e.g., forestry and agriculture (Campbell and Doeg, 1989; Waters, 1995). Stream substrate type is strongly influenced by geology (Richards et al., 1996; Richards et al., 1997) and therefore, local communities will be indirectly influenced regionally through geological influences on substrate characteristics. This study has identified the importance of sedimentation as an effect of forest harvesting on stream communities. The different response of macroinvertebrates between regions may therefore reflect the different substrate types and erodibility of the underlying geology. Stream communities in Lismore and Te Wera (both underlain with sandstone, siltstone and limestone geology) will therefore show greater changes after harvesting due to greater potential for sedimentation in these regions. Stream communities in the Tawarau and Pirongia regions showed little change due to the greater proportion of bedrock and volcanic geology, and therefore stable substrate types.

Friberg and Winterbourn (1997) and Friberg et al. (1997) determined that regional factors were more important in determining algal biomass and production, and invertebrate community structure in streams draining forest (exotic versus native). However, it is evident that land use influences become more important to invertebrate communities when considering land use that differs in more than vegetation type i.e., pasture (open, unstable) versus native (shaded, stable). Rather than being the dominant regulator of community structure, as Friberg et al. (1997) suggest, regional influences may affect the response of communities to changes in the stream environment as a result of forest harvesting. Harding et al. (1997) found that as the level of catchment modification increased, through deforestation and farm development, the distinctiveness of regional stream fauna declined and was eventually lost. Richards et al. (1996) found that geology and climate were important to communities when assessed at larger spatial scales and land use influences became more important determinants of community structure at smaller scales i.e., within regions. The importance of regional influences, and the effects of spatial scale on stream communities with respect to land use, is therefore well demonstrated and may necessitate the incorporation of regional influences in environmental management and monitoring programmes throughout New Zealand.

The influence of regional factors on responses of communities to disturbance may also occur on a larger scale between the North and South Island. Most research into forestry and land use effects on invertebrate communities has occurred in the South Island with communities showing rapid recovery from disturbances such as harvesting (Graynoth, 1979; Rounick and Winterbourn, 1982; Winterbourn, 1986; Friberg and Winterbourn, 1997; Friberg et al., 1997). However, this study indicates that although harvesting effects may be mitigated by reforestation, they are still relatively long term (greater than six years). Although community responses to forest type in the North and South Island are similar with respect to streams draining mature exotic and native forest, response of communities to exotic forest harvesting may be more long term in central North Island streams of this study that have easily erodible geology compared to those in the South Island.

In summary, macroinvertebrate communities are affected by forest harvesting, with higher abundances of enrichment tolerant taxa in streams draining harvested catchments compared to streams draining mature exotic forest. Regional influences, which may not primarily determine community structure, do appear to influence the response of a community to forest harvesting and changes in land use practices due to influences on stream substrate characteristics. Effects seem to be influenced differently at several spatial scales i.e., within a catchment communities may show stronger responses to land use but between regions, responses may be less marked due to influences of the climate and underlying geology. Effects may also be more long term in central North Island streams compared to those in the South Island.

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Appendix 2.1. Site name abbreviations and details for 42 streams sampled in Lismore, Te Wera, Tawarau and Pirongia forests between January and February 1997.

Site abbreviation	Site name	Land use type	NZMS 260 grid reference	Altitude (m)	Stream order
LF1	Lismore exotic 1	Mature	S22: 975524	80	3
LF2	Lismore exotic 2	Harvested	S22: 967511	60	3
LF3	Lismore exotic 3	Harvested	S22: 979494	80	2
LF5	Lismore exotic 5	Mature	S22: 049505	40	2
LF6	Lismore exotic 6	Harvested	S22: 994485	80	1
LP1	Lismore pasture 1	Pasture	S22: 980486	40	2
LP2	Lismore pasture 2	Pasture	S22: 053504	60	3
LP3	Lismore pasture 3	Pasture	S22: 954471	40	3
LN1	Lismore native 1	Native	S21: 948661	40	1
LN2	Lismore native 2	Native	S21: 944654	40	2
LN3	Lismore native 3	Native	S21: 104602	200	2
TaF1	Tawarau exotic 1	Harvested	R16: 825225	280	1
TaF2	Tawarau exotic 2	Mature	R16: 809227	280	1
TaF5	Tawarau exotic 5	Mature	R16: 789228	250	2
TaF6	Tawarau exotic 6	Mature	R16: 773205	260	2
TaF7	Tawarau exotic 7	Mature	R16: 788228	220	1
TaF11	Tawarau exotic 11	Mature	R16: 796231	260	2
TaP1	Tawarau pasture 1	Pasture	R16: 825222	280	2
TaP2	Tawarau pasture 2	Pasture	R16: 822226	280	2
TaN1	Tawarau native 1	Native	R16: 776237	180	1
TaN2	Tawarau native 2	Native	R16: 778236	200	1
PF8	Pirongia exotic 8	Mature	S15: 931412	120	2
PF9	Pirongia exotic 9	Mature	Q15, R15:898417	180	2
PF10	Pirongia exotic 10	Mature	Q15, R15:901415	180	2
PP3	Pirongia pasture 3	Pasture	S15: 913423	120	3
PN3	Pirongia native 3	Native	S15: 921431	180	1
TWF1	Te Wera exotic 1	Harvested	R19: 502207	200	1
TWF2	Te Wera exotic 2	Harvested	Q19: 495211	200	1
TWF3	Te Wera exotic 3	Mature	Q20: 498178	200	1
TWF4	Te Wera exotic 4	Harvested	Q20: 486178	200	1
TWF5	Te Wera exotic 5	Harvested	R19: 522231	220	1
TWF6	Te Wera exotic 6	Mature	R19: 524215	220	1
TWF7	Te Wera exotic 7	Mature	R19: 521201	200	1
TWF8	Te Wera exotic 8	Mature	R19: 509174	180	2
TWF10	Te Wera exotic 10	Harvested	R20: 514182	200	1
TWF11	Te Wera exotic 11	Mature	Q20: 497183	200	1
TWP1	Te Wera pasture 1	Pasture	Q20: 494203	200	1
TWP2	Te Wera pasture 2	Pasture	Q20: 491199	200	1
TWP3	Te Wera pasture 3	Pasture	Q20: 483188	180	1
TWN1	Te Wera native 1	Native	R19: 520224	360	1
TWN2	Te Wera native 2	Native	R20: 515177	200	1
TWN3	Te Wera native 3	Native	Q20: 498183	200	1

Appendix 2.2. Physiochemical parameters for 42 streams in Lismore, Te Wera, Tawarau and Pirongia forests sampled in January and February 1997. Abbreviations in Appendix 1.

Site	Width (m)	Depth (cm)	Velocity (m/s)	Cond (μS/cm)	pH	DO (mg/l)	Temp (°C)	Stability	SS (mg/l)
LF1	1.39	5.78	0.3130	491	7.1	11.42	11.7	105	0.094
LF2	0.79	9.2	0.1512	661	7.09	2.92	15.2	133	0.248
LF3	0.5	10.0	0.0886	570	7.23	8.60	16.6	83	0.072
LF5	2.22	1.14	0.2764	664	7.2	9.27	18.4	130	0.642
LF6	1.95	16.38	0.3530	359	7.83	9.08	16.8	91	0.089
LP1	2.18	31.8	0.4790	393	7.2	9.24	15.8	76	0.071
LP2	0.51	5.32	0.1771	479	7.55	9.73	17.4	97	0.094
LP3	2.0	15.60	0.7510	322	7.78	9.49	16.6	120	0.074
LN1	0.34	5.1	0.2019	247	7.78	10.20	14.5	64	0.305
LN2	0.77	10.58	0.1954	270	7.94	10.06	14.2	65	0.023
LN3	1.76	20.44	0.3665	272	7.62	11.06	13.2	82	0.075
TaF1	1.21	15.46	0.1771	107	7.21	9.30	13.0	141	0.025
TaF2	0.72	9.72	0.2538	134	6.84	9.60	12.2	64	0.057
TaF5	1.5	24.50	0.3671	191	7.15	10.50	11.8	114	0.036
TaF6	1.62	52.72	0.3552	153	7.23	9.40	12.1	114	0.060
TaF7	0.29	6.16	0.2580	101	6.94	9.40	12.5	109	0.092
TaF11	1.15	11.02	0.2947	117.5	7.19	10.6	11.4	62	0.033
TaP1	1.46	35.10	0.3228	90	6.81	9.60	14.8	91	0.039
TaP2	1.74	32.20	0.3186	122	6.66	8.70	14.3	77	0.101
TaN1	1.52	18.40	0.3282	159	7.19	9.40	12.3	115	0.044
TaN2	1.57	16.18	0.3687	156.5	7.24	10.40	12.3	84	0.051
PF8	1.33	16.62	0.4503	68	7.09	9.70	12.4	74	0.018
PF9	1.61	12.24	0.6276	128	7.00	9.20	15.0	66	0.016
PF10	1.29	10.40	0.5157	80	6.90	9.50	13.5	63	0.045
PP3	2.83	14.92	0.6523	401	7.21	9.70	19.9	95	0.016
PN3	0.90	6.62	0.1984	188	7.21	10.10	12.3	72	0.058
TWF1	1.34	29.34	0.1954	123	7.19	7.20	15.5	89	0.007
TWF2	2.02	25.56	0.2904	134	7.12	5.00	16.3	87	0.094
TWF3	0.35	6.90	0.3164	147	7.12	8.10	15.2	80	0.392
TWF4	0.35	3.05	0.1695	101	7.09	8.80	17.4	114	2.831
TWF5	1.60	4.04	0.1954	132.5	8.72	8.60	17.7	112	0.038
TWF6	0.49	7.76	0.1652	102	7.14	9.80	13.3	91	0.278
TWF7	0.93	16.3	0.6475	93	7.20	8.10	15.2	76	0.033
TWF8	1.92	20.52	0.3687	124	7.24	8.90	14.1	123	0.272
TWF10	1.14	25.86	0.1836	66	6.70	9.70	14.2	114	0.083
TWF11	1.05	33.60	0.5262	95	6.82	-	14.1	97	0.088
TWP1	1.00	44.46	0.9963	90	7.02	7.50	15.45	105	0.020
TWP2	1.53	30.88	0.4474	88	6.61	8.70	15.8	101	0.047
TWP3	0.84	18.64	0.2725	97	6.67	-	17.2	118	0.010
TWN1	0.94	12.80	0.1878	76	6.85	8.20	13.6	85	0.088
TWN2	0.44	7.96	0.1328	53	7.22	7.30	13.2	73	0.173
TWN3	0.59	24.90	0.5839	88	7.00	10.30	13.7	61	0.043

Appendix 2.3. Mean particulate organic matter (POM) (with range in parentheses) for land use types within each region from benthic macroinvertebrate samples during January and February 1997.

Region	Land use type	CPOM (g AFDW/0.1m^2)	MPOM (g AFDW/0.1m^2)	Total POM (g AFDW/0.1m^2)
Lismore				
	Mature	0.23 (0.19-0.27)	0.22 (0.13-0.32)	0.45 (0.13-0.32)
	Harvested	0.53 (0.17-1.00)	1.26 (0.18-3.21)	1.79 (0.17-3.21)
	Pasture	1.74 (0.34-1.17)	0.09 (0-0.14)	1.83 (0-1.17)
	Native	0.54 (0.27-0.79)	0.15 (0.06-0.23)	0.69 (0.06-0.79)
Tawarau				
	Mature	2.42 (0.47-4.51)	1.33 (0.67-1.75)	3.75 (0.47-4.51)
	Harvested	7.21	6.71	13.93
	Pasture	2.35 (1.36-3.34)	3.07 (1.02-5.11)	5.41 (1.02-5.11)
	Native	6.39 (3.56-9.21)	0.97 (0.86-1.07)	7.35 (0.86-9.21)
Pirongia				
	Mature	1.56 (1.31-1.86)	0.50 (0.22-0.72)	2.07 (0.22-1.86)
	Pasture	0.94	0.17	1.11
	Native	1.81	0.43	2.24
Te Wera				
	Mature	3.41 (1.84-5.40)	1.30 (0.39-2.30)	4.71 (0.39-5.40)
	Harvested	1.06 (0.38-1.70)	0.58 (0.14-1.20)	1.64 (0.14-1.70)
	Pasture	3.55 (1.56-5.48)	1.32 (0.77-1.85)	4.88 (0.77-5.48)
	Native	1.98 (1.48-2.98)	0.70 (0.62-0.78)	2.68 (0.62-2.98)

CHAPTER THREE

Short term response of stream invertebrates to forest harvesting: comparison of pre and post harvest conditions

ABSTRACT

Twelve streams were sampled every two months from February to December 1997, in three exotic forests, Lismore, Te Wera and Tawarau, on the west coast of central North Island, New Zealand, to monitor changes in macroinvertebrate community structure before and after harvesting, and to assess the effects of seasonal influences on macroinvertebrate community response to forest harvesting and land use management.

Taxonomic richness and diversity were similar in streams draining mature and harvested exotic forest, but total abundance increased after harvesting. Community composition changed from Ephemeroptera-dominated to one consisting mainly of pollution tolerant taxa e.g., Oligochaeta, Mollusca and Diptera. However, road construction associated with forest harvesting had little effect on the stream invertebrate community with only the number of individuals increasing after construction commenced. Multivariate analyses separated communities according to canopy cover i.e., streams draining pasture and harvested exotic forest had similar communities, as did streams draining native and mature exotic forest. Changes in communities over time revealed that initial impacts of forest harvesting (i.e., decreased invertebrate abundance due to sedimentation) were rapidly ameliorated, and the pollution tolerant communities that established after harvesting were determined by the long term effects of riparian vegetation removal and sedimentation. Seasonal influences were only evident for periphyton biomass and the number of invertebrate taxa, and did not affect the response of macroinvertebrate communities to forest harvesting.

Keywords: community composition, forestry, harvesting, macroinvertebrates, monitoring, periphyton, road construction, season, temporal scale.

INTRODUCTION

Harvesting of exotic forest has long been recognised to change stream macroinvertebrate community structure from being dominated by taxa that are less tolerant of low water quality to those which are more tolerant. Many studies have revealed such effects through comparisons of mature forest catchments with harvested catchments at varying stages of recovery (e.g., Davies and Nelson, 1994; Fore and Karr, 1996; Brown et al., 1997; Stone and Wallace, 1998). However, few studies have looked at immediate effects of harvesting i.e., within the first six to 12 months (but see Brown et al., 1997). Although some studies have assessed short term effects of harvesting on water chemistry and physical stream characteristics e.g., nutrient concentrations, water temperature, substrate composition, organic matter and suspended solids (Mosley and Rowe, 1981; Garman and Moring, 1991; Neal et al., 1992), only a few have considered the biological responses to initial disturbance (e.g., Holopainen and Huttunen, 1992).

Although rapid recovery of macroinvertebrates to disturbance is well documented (Resh et al., 1988), the immediate effect of harvesting on invertebrates may differ markedly from longer term effects (Newbold et al., 1980; Davies and Nelson, 1994). Such short term changes in community structure cannot be detected with studies conducted two to three years after harvesting.

Road construction is often also an integral part of harvesting that may cause even more detrimental effects to stream invertebrates (Hornbeck and Reinhart, 1964; Mosley, 1980; Harr and Fredriksen, 1988; Campbell and Doeg, 1989). Studies on the biological effects of road construction are also limited. However, the dominant physical effect of road construction, increased sedimentation, generally results in decreased abundance and diversity of macroinvertebrates (Ryan, 1991).

Similarly, seasonal fluctuations in climate (e.g., temperature, light and rainfall) may affect stream invertebrate communities through effects on flow regimes, food sources, habitat and life cycles (Hirsch, 1958; Towns, 1981). Such effects on macroinvertebrate communities may also be important in determining the response of invertebrates to disturbances such as forest harvesting (Hirsch, 1958; Scott et al., 1994). Furthermore, seasonal patterns in stream invertebrate community structure may differ with land use type. Although New Zealand invertebrate fauna appear to be less influenced by seasonal fluctuations compared to many Northern Hemisphere studies (Towns, 1981; Winterbourn et al., 1981; Winterbourn, 1982), recent studies have found that some seasonal changes in communities do occur (Scarsbrook and Townsend, 1993; Fowler and Death, unpublished data).

The aim of this study is to monitor the short term effects of harvesting operations, including road construction, on macroinvertebrate communities, and to examine whether seasonal patterns in community structure differ with land use type. This was achieved by monitoring 12 streams, two of which had catchments harvested during the monitoring period, in three exotic forests in central North Island.

STUDY SITES

To assess the influence of seasonal effects on stream invertebrate communities between land use types, ten streams were sampled, once every two months between February and December 1997, in or around three exotic forests, Lismore, Te Wera and Tawarau, on the west coast of central North Island, New Zealand. All streams were first to third order, with catchments draining 90% *Pinus radiata,* 90% native forest or 90% pasture. Within each forest, a mature exotic forest site (greater than 15 years old), a harvested site (clearfelled to the stream edge within the last year), a native forest site and a pasture site (except Lismore, for the latter two) were sampled.

To assess short term effects of harvesting, one stream in Te Wera (harvested between March and November 1997), and one in Tawarau (road construction from July to December 1997) were sampled every two months before and after disturbance (Plate 3.1). Sampling occasions before harvesting or road construction were termed "pre harvest" and those after harvesting "post harvest". Corresponding "control" streams of mature exotic forest were labelled "control pre" and "control post" to correspond to pre and post harvest samples respectively.

PLATE 3.1

Plate 3.1. Pre and post harvest sites in Te Wera (harvested from March to November) (top), and in Tawarau (road constructed from July to December) (bottom), sampled once every two months from February to December 1997.

METHODS

Sampling protocol

Invertebrate samples and physiochemical measures were collected in the first week of February, April, June, August, October and December 1997, using the methods outlined in Chapter Two (except that three rather than four Surber samples were collected).

Statistical analyses

Statistical differences between treatments (land use type, region, stream, month) were analysed using a four-way Analysis of Variance (ANOVA) and Duncan multiple range a posteriori means test with SAS (SAS, 1989). Land use type, region and month were treated as fixed effects and streams within each group as random. Data were log transformed ($\log_{10}(x+1)$) before analysis, if necessary, to increase variance homogeneity. Pre and post harvest samples and respective controls were compared in a three-way design (pre and post harvest, stream, month).

Classification of communities (log transformed) was conducted with a cluster analysis (Sorenson distance measure and group average linkage method) using the PC-ORD statistical package (McCune and Mefford, 1995). Ordination of site communities (log transformed) was also carried out with Detrended Correspondence Analysis (DECORANA) using the PC-ORD statistical package (McCune and Mefford, 1995) and environmental variables correlated with the axes using Pearson's correlation coefficient.

RESULTS

The Influence of Seasonal Variation

Physiochemical characteristics

All streams were similar in physiochemical nature (Table 3.1 and 3.2), although overall streams in harvested catchments had higher levels of silt and sand, and were less stable compared to those in mature exotic forest and pasture (Table 3.2). Of the physiochemical characteristics, only dissolved oxygen (increasing during winter) and water temperature (decreasing during winter) changed seasonally (Table 3.1 and 3.2).

Total particulate organic matter (POM) was not significantly different between land use types ($F_{3,41}$=1.17, P=0.33). Medium particulate organic matter (MPOM) was greatest in streams draining harvested catchments however, and least in streams draining native forest ($F_{3,41}$=12.54, P<0.001) (Appendix 3.1).

Periphyton

Periphyton biomass ranged from 0.02 to 14.85 mg/cm^2, with streams in native and harvested exotic forest having the lowest periphyton biomass and streams in mature exotic forest the highest ($F_{3,174}$=14.51, P<0.001) (Fig. 3.1a). There were also differences between seasons ($F_{5,174}$=9.78, P<0.001), with higher periphyton biomass in summer (December to March) and autumn (April, May), and lowest biomass in spring (September, October) and winter (June to August) (Fig. 3.1b). Seasonal changes in periphyton biomass were different between land use types ($F_{15,159}$=1.77, P=0.04) with biomass changing the most throughout the year in streams draining mature exotic forest (Fig. 3.1b).

Table 3.1. Mean physiochemical characteristics (with range in parentheses) for land use type and pre and post harvest streams with respective controls, at 12 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997. Cond = conductivity, DO = dissolved oxygen, Temp = temperature, SS = suspended solids.

	No. of streams	Stream order	Altitude (m)	Width (m)	Depth (cm)	Velocity (m/s)	pH	Cond (μS/cm)	DO (mg/l)	Temp (°C)	Stability	SS (mg/l)
Land use type												
Mature	3	2 (1-3)	177 (80-250)	1.30 (0.74-1.76)	15.5 (4.7-33.6)	0.41 (0.20-0.75)	7.36 (6.82-8.10)	282 (95-557)	10.70 (8.75-14.30)	10.7 (5.0-14.1)	105 (97-114)	0.07 (0.001-0.30)
Harvested	3	2.5 (1-3)	187 (60-280)	1.55 (0.78-2.54)	17.9 (4.0-44.0)	0.26 (0-1.20)	7.31 (6.65-8.72)	289 (92-684)	8.07 (1.10-12.00)	11.9 (5.2-17.7)	129 (112-141)	0.06 (0.004-0.40)
Pasture	2	1.5 (1-2)	240 (200-280)	1.13 (0.61-1.74)	19.4 (9.0-32.2)	0.76 (0.15-1.80)	7.17 (6.61-8.20)	108 (80-129)	9.39 (7.61-11.20)	13.5 (8.4-17.4)	89 (77-101)	0.03 (0.01-0.10)
Native	2	2 (1-3)	190 (180-200)	0.87 (0.46-1.52)	16.6 (11.5-24.9)	0.45 (0.25-0.80)	7.39 (7.00-8.30)	145 (88-169)	10.84 (8.63-12.90)	10.4 (3.7-13.9)	88 (61-115)	0.03 (0-0.13)
Te Wera pre/post												
Control pre	1	1	200	0.96 (0.86-1.05)	27.3 (21.0-33.6)	0.39 (0.25-0.53)	6.94 (6.82-7.06)	133 (95-170)	10.30	13.5 (12.8-14.1)	97	0.03 (0.001-0.09)
Control post				0.91 (0.74-1.33)	21.8 (20.6-22.8)	0.24 (0.20-0.30)	7.12 (6.87-7.70)	140 (115-168)	10.56 (8.75-12.40)	9.0 (5.0-12.2)	97	0.02 (0.01-0.02)
Pre harvesting	1	1	220	0.49	7.8	0.17	7.14	102	9.80	13.3	91	0.28
Post harvesting				0.94 (0.51-1.31)	22.3 (8.0-27.8)	0.10 (0.00-0.25)	6.92 (6.42-8.20)	132 (86-194)	8.29 (3.95-10.80)	9.5 (3.8-11.9)	141	0.04 (0.01-0.10)
Tawarau pre/post												
Control pre	1	2	250	1.48 (1.47-1.50)	24.6 (24.5-24.7)	0.36 (0.35-0.37)	7.28 (7.15-7.41)	193 (191-198)	11.75 (10.50-13.00)	11.9 (11.8-11.9)	114	0.05 (0.04-0.07)
Control post				1.41 (1.40-1.43)	13.5 (11.8-15.7)	0.68 (0.55-0.75)	7.52 (7.17-8.10)	203 (199-206)	10.25 (8.90-11.10)	10.6 (9.4-11.8)	114	0.017 (0.008-0.02)
Pre harvesting	1	2	260	1.12 (1.08-1.15)	11.1 (11.0-11.1)	0.47 (0.29-0.65)	7.36 (7.19-7.53)	116 (115-117)	11.75 (10.60-12.90)	12.0 (11.4-12.5)	62	0.03 (0.02-0.03)
Post harvesting				0.87 (0.73-1.05)	13.0 (11.2-14.4)	0.60 (0.40-0.80)	7.53 (7.22-8.10)	122 (120-126)	10.63 (9.70-11.60)	10.4 (9.4-12.1)	94	0.02 (0.02-0.03)

Table 3.2. *F* ratios and *P* values (* = $P<0.05$) from an ANOVA testing the null hypothesis that physiochemical characteristics do not differ between land use type and month (sampling occasion) for 12 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997 (includes pre and post harvest streams).

Characteristic	**Land use type**		**Month**			
	***F* value**	***P* value**	***F* value**	***P* value**		
Velocity	4.23 *	0.01	0.71	0.62		
pH	0.73	0.54	1.27	0.29		
Conductivity	1.68	0.18	0.22	0.95		
Dissolved Oxygen	2.13	0.12	4.14 *	0.003		
% Cover	137.88 *	<0.001	0.04	1.00		
Temperature	2.23	0.10	4.22 *	0.003		
Silt	7.63 *	<0.001	0.00	1.00		
Sand	23.03 *	<0.001	0.00	1.00		
Stability	13.53 *	<0.001	0.34	0.89		
Suspended Solids	0.31	0.82	0.96	0.45		
	Pre/post harvest / Control		**Means test (different letters show significant differences)**			
	***F* value**	***P* value**	**Control pre**	**Control**	**Pre harvest**	**Post harvest**
Velocity	0.32	0.81				
pH	0.94	0.45				
Conductivity	9.50 *	0.002	A	A	B	B
Dissolved Oxygen	1.07	0.40				
% Cover	3.18	0.06	B	B	A	B
Temperature	11.13 *	0.001	A	B	A	B
Silt	0.64	0.60				
Sand	0.84	0.50				
Stability	1.86	0.19				
Suspended Solids	1.72	0.22				

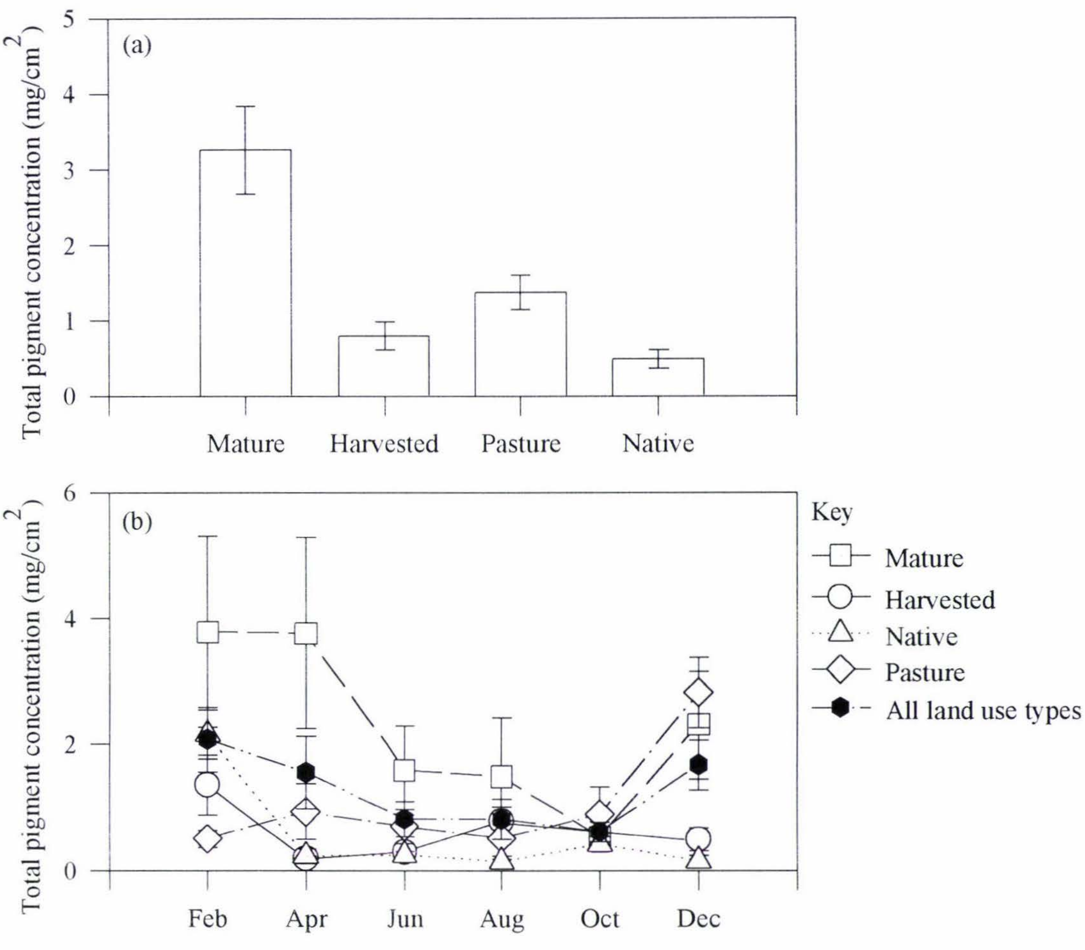

Figure 3.1. Mean total pigment (chlorophyll *a* and phaeophytin) concentrations (± 1 SE) for (a) land use type and (b) land use types at each sampling occasion, from stones collected at 10 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997.

Macroinvertebrates

Streams in native and mature exotic forest tended to have greater relative abundances of Ephemeroptera (mainly *Zephlebia dentata*), Plecoptera and Coleoptera, with these groups having lower abundances or being absent from streams in harvested exotic forest and pasture catchments. The latter sites were dominated by Diptera (Chironomidae) and Crustacea (Amphipoda and Ostracoda).

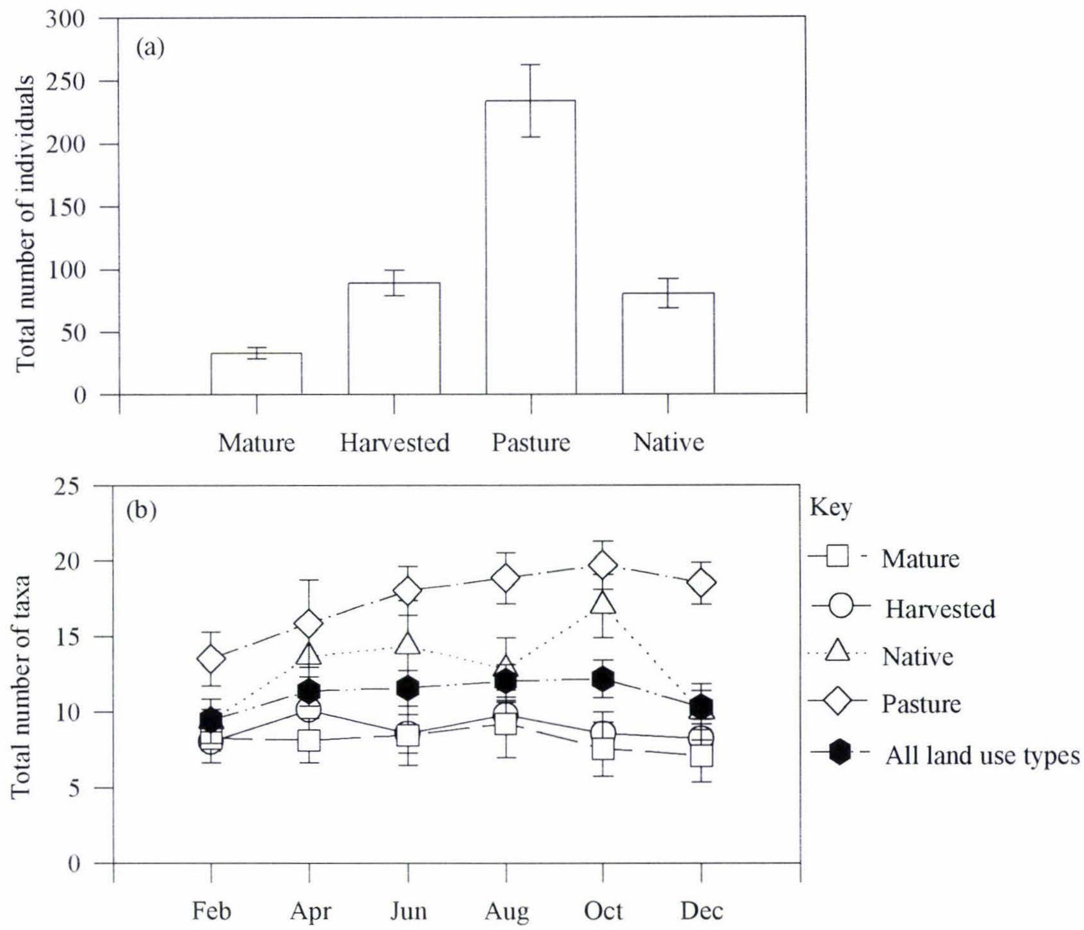

Figure 3.2. Mean total number of individuals (± 1 SE) for land use type (a) and mean total number of taxa (± 1 SE) for land use types at each sampling occasion (b), for macroinvertebrates collected at 10 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997.

The total number of individuals was highest at pasture sites and lowest at mature sites ($F_{3,179}$=41.15, P<0.001) (Fig. 3.2a), but did not differ with season ($F_{5,179}$=1.25, P=0.29). Pasture sites also had the highest number of taxa, followed by native, and then mature and harvested exotic forest sites, which were not significantly different ($F_{3,179}$=26.70, P<0.001) (Fig. 3.2b). Seasonal differences were not highly significant ($F_{5,179}$=2.14, P=0.06) but spring supported the highest number of taxa followed by winter, autumn and then summer having the lowest (Fig. 3.2b). Pasture and native sites had the greatest taxonomic richness and evenness, whereas exotic forest sites were dominated by only one or two taxa e.g., Chironomidae, Ostracoda, *Potamopyrgus antipodarum* or

Oligochaeta. Seasonal influences on the number of taxa or number of individuals did not differ between land use type ($F_{15,164}$=1.16, P=0.31 and $F_{15,164}$=0.99, P=0.47 respectively). However, streams draining native forest showed some variation between sampling occasions (Fig. 3.2b).

The MCI, QMCI and % EPT were all higher in streams draining native and mature exotic forest than in streams draining pasture and harvested exotic forest ($F_{3,49}$=35.15, P<0.001; $F_{3,49}$=32.43, P<0.001; $F_{3,51}$=17.07, P<0.001 for MCI, QMCI and % EPT respectively) (Fig. 3.3), with no seasonal effects apparent ($F_{5,49}$=0.38, P=0.86; $F_{5,49}$=0.54, P=0.74; $F_{5,34}$=0.44, P=0.82 respectively).

<u>**Pre and Post Harvest Effects**</u>

Physiochemical characteristics

Decreases in temperature occurred after harvesting and road construction (Table 3.1 and 3.2). However, this decrease is likely due to seasonal influences as similar changes occurred at control post sampling occasions (Table 3.1). Conductivity was higher after harvesting (Table 3.1), with control streams showing no differences between pre and post sampling occasions (Table 3.2). Particulate organic matter was also higher after harvesting ($F_{3,11}$=4.33, P=0.03) (Appendix 3.1).

Periphyton

Periphyton biomass decreased after harvesting ($F_{3,70}$=6.70, P=0.001) (Fig 3.4). However, this is probably influenced seasonally ($F_{5,70}$=3.53, P=0.01) as periphyton biomass dropped to the same extent in both the control post and post harvest sampling occasions.

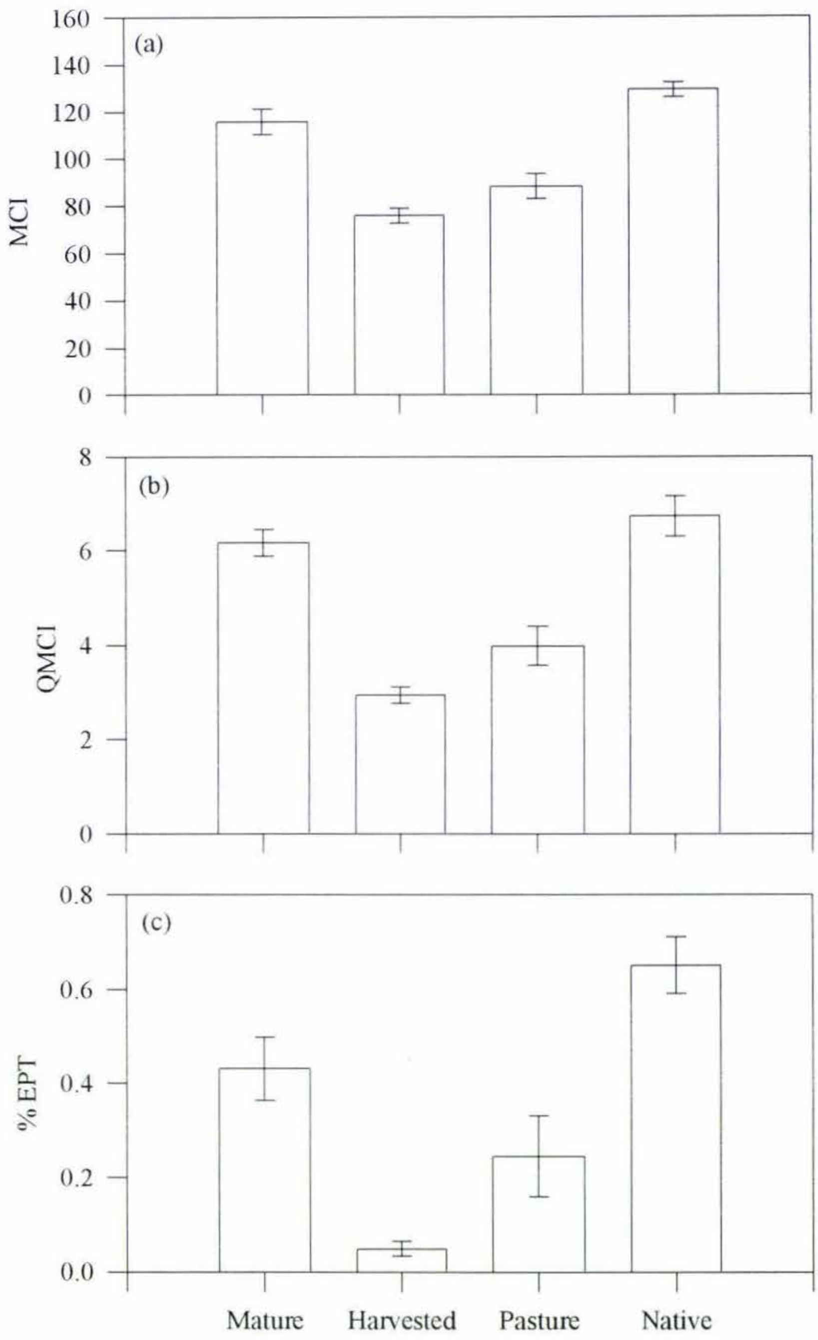

Figure 3.3. Mean biotic index scores (± 1 SE) for each land use type for (a) Macroinvertebrate Community Index (MCI), (b) Quantitative Macroinvertebrate Community Index (QMCI) and (c) the Percentage of Ephemeroptera, Plecoptera and Tricoptera (% EPT), for macroinvertebrates collected at 10 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997.

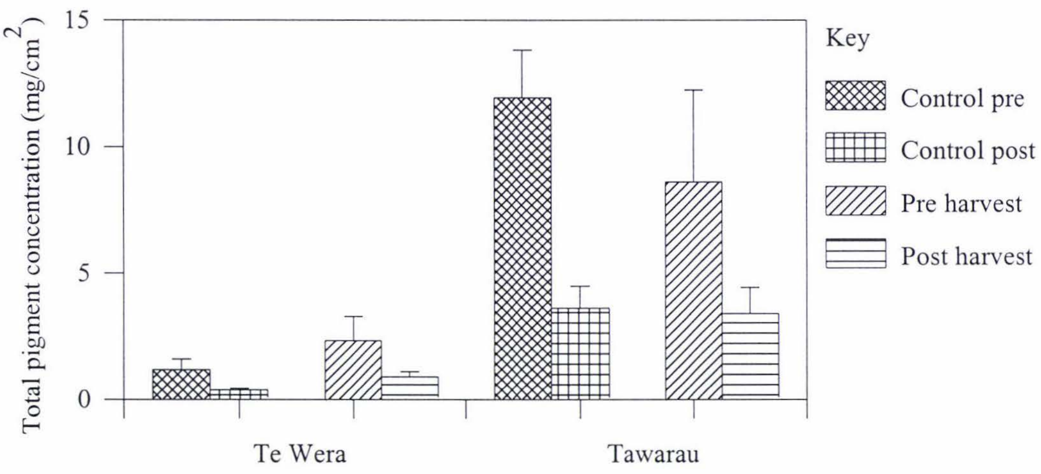

Figure 3.4. Mean total pigment (chlorophyll *a* and phaeophytin) concentrations (± 1 SE) for pre and post harvest streams with respective controls, from stones collected at four streams in Te Wera and Tawarau forests, sampled every two months from February to December 1997.

Macroinvertebrates

The number of individuals were higher after harvesting for both Te Wera and Tawarau ($F_{3,62}$=7.19, P<0.001) (Fig. 3.5a). Control streams did not show changes of the same magnitude for the corresponding period, and in Tawarau abundances actually dropped in control post samples. However, the number of taxa before and after harvesting in Te Wera and Tawarau showed no significant differences ($F_{3,62}$=0.32, P=0.81) (Fig. 3.5b), and control streams exhibited more change than did the pre and post harvest streams.

At Te Wera, harvesting yielded an increase in evenness but at Tawarau exhibited a decrease (Simpson's index, $F_{3,62}$=3.09, P=0.03) (Fig. 3.6b). There were no significant differences before and after harvesting for Margelef's index ($F_{3,62}$=1.36, P=0.26) although a slight decrease did occur after harvesting in Te Wera (Fig. 3.6a). Control streams showed no differences between corresponding pre and post samples for the Simpson's index but exhibited some variation for the Margelef's index (Fig. 3.6).

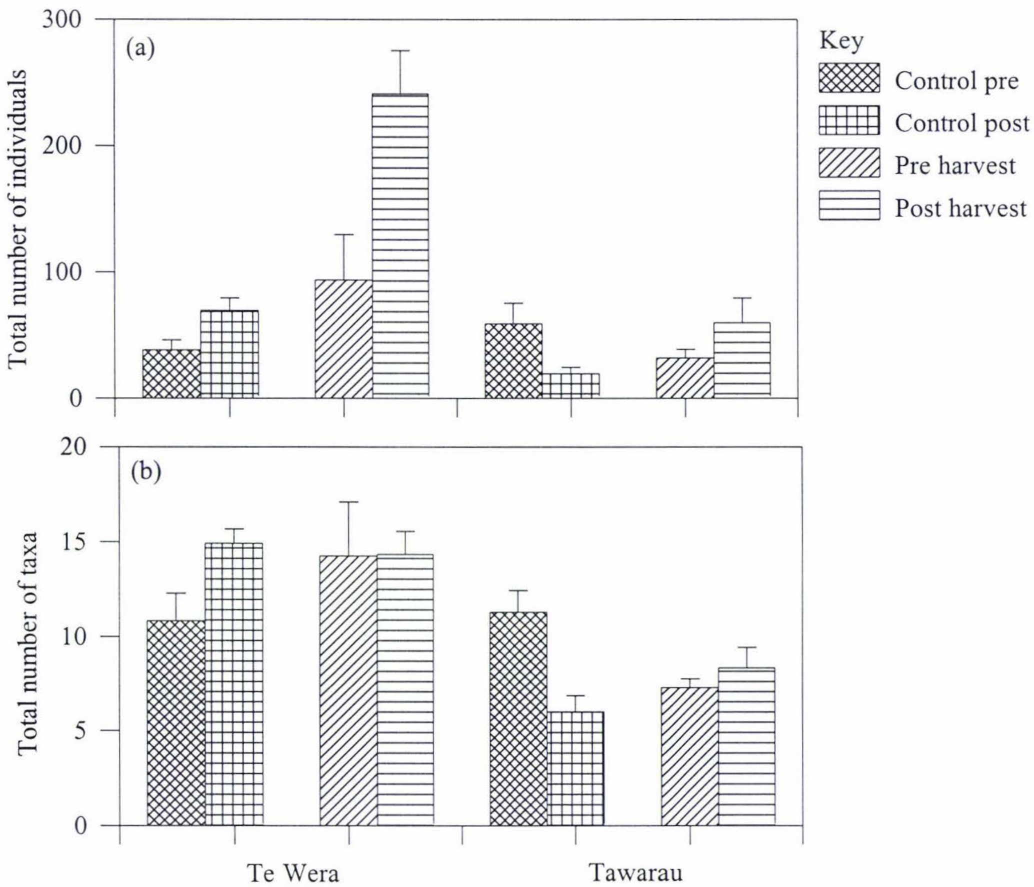

Figure 3.5. Mean total number of individuals (± 1 SE) (a) and mean total number of taxa (± 1 SE) (b) for pre and post harvest streams with respective controls, for macroinvertebrates collected at four streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997.

There were similarly no differences in MCI, QMCI or % EPT before and after harvesting ($F_{3,12}$=2.32, P=0.13; $F_{3,12}$=1.74, P=0.21 and $F_{3,14}$=0.38, P=0.77 respectively). However, for the Te Wera pre and post harvest stream, all indices decreased after harvesting commenced.

Multivariate Analyses

Cluster analysis

Cluster analysis generally grouped stream communities into harvested exotic forest and pasture sites versus mature exotic and native forest sites (Fig. 3.7), with the exception of

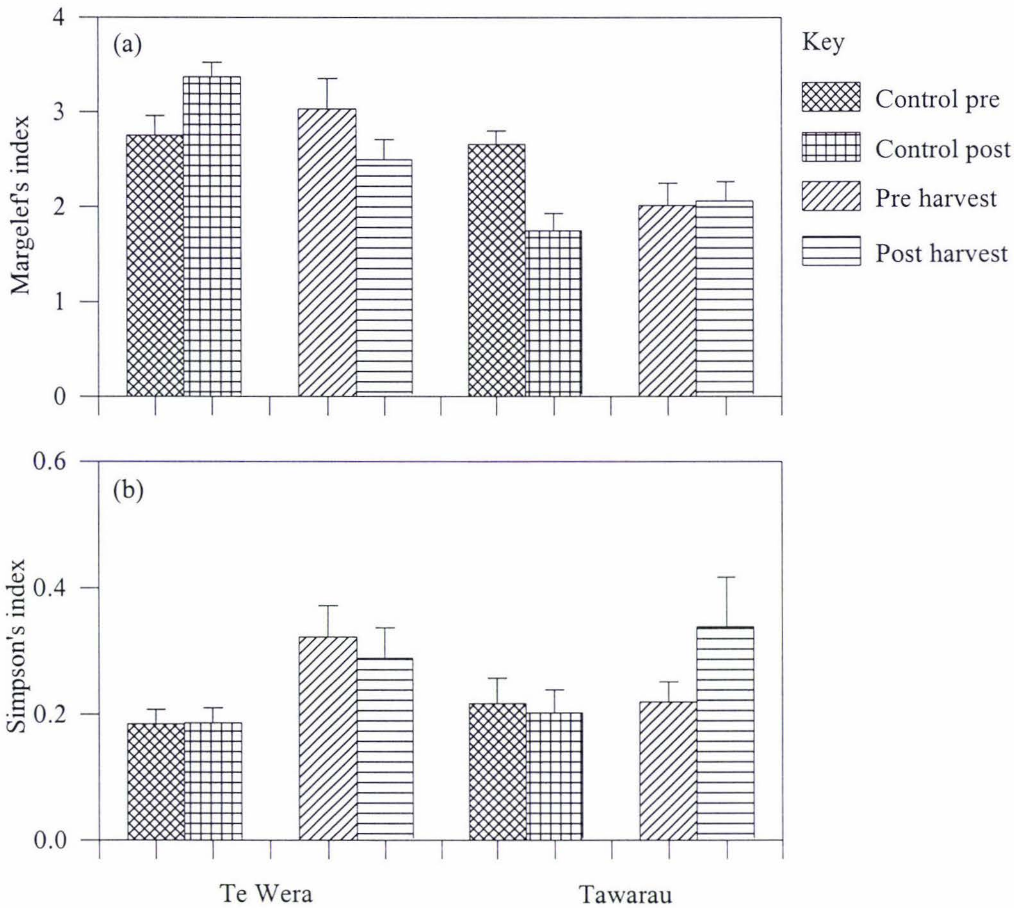

Figure 3.6. Mean diversity index scores (± 1 SE) for pre and post harvest samples with respective controls for (a) Margelef's index and (b) Simpson's index, for macroinvertebrates collected at four streams in Te Wera and Tawarau forests, sampled every two months from February to December 1997.

Tawarau post harvest sampling occasions (Ta-post 3 to 6) which occurred with pre harvest and native Tawarau sampling occasions; and one Te Wera post harvest sample (Te-post 6) which occurred with pre harvest Te-pre 1. Three Lismore mature exotic forest sampling occasions were different from all other sampling occasions.

Within these two broad groups, sites were grouped into regions e.g., at both Te Wera and Tawarau, harvested exotic forest and pasture sites were most similar. This regional division was similar in the mature exotic and native forest site groups except for the Te Wera and Tawarau streams already mentioned (Te-post 6 and Tawarau pre and post harvest sampling occasions).

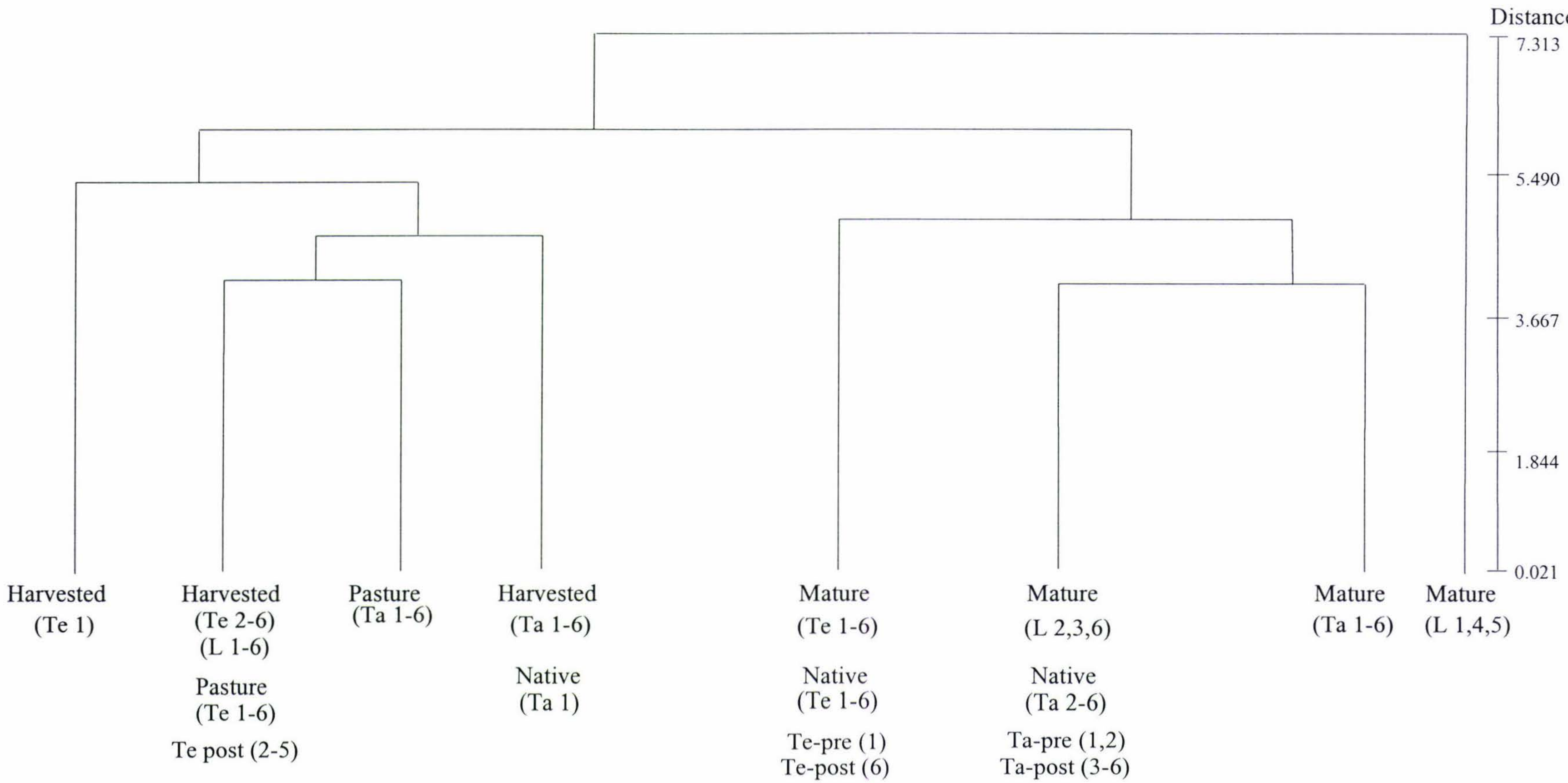

Figure 3.7. Cluster analysis of communities from 12 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997 (includes pre and post harvest streams). Abbreviations are: L = Lismore; Te = Te Wera; Ta = Tawarau; pre = pre harvest sample and post = post harvest sample. Numbers in brackets correspond to sampling occasions i.e., 1 = first sample in February and 6 = last sample in December.

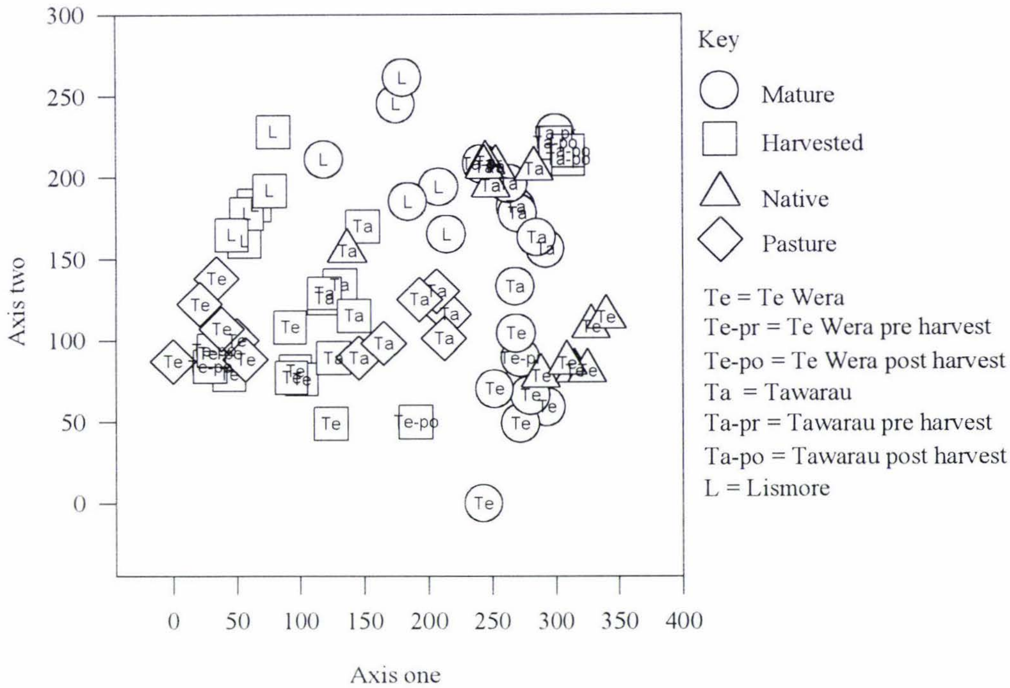

Figure 3.8. Axis one of a Detrended Correspondence Analysis as a function of axis two for macroinvertebrate communities collected from 12 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997 (includes pre and post harvest streams).

Ordination

Generally, communities for each sampling occasion were grouped in the DECORANA by stream and land use type, being separated by land use on axis one and stream on axis two (Fig. 3.8). Although in general, clear divisions were not apparent, within each region (Lismore, Te Wera and Tawarau) all streams draining native and mature exotic forest had similar communities and all streams draining pasture and harvested exotic forest had similar communities (with the exception of Tawarau pre and post road construction). Axes one and two accounted for 44.3% and 13.1% of the variance in the data respectively.

The axes correlated with many of the physical and chemical variables measured (Table 3.3). For example, those correlated with axis one indicated that stream communities in recently harvested or pasture catchments had higher proportions of silt and sand, and decreased stability and riparian cover. The MCI, QMCI and % EPT were also highly

Table 3.3. Variables correlated (* = $P<0.05$) with Detrended Correspondence Analysis axes for macroinvertebrate communities collected from 12 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997 (includes pre and post harvest streams).

Parameter	Axis 1	Axis 2	Axis 3
% cover	0.66 *	0.55 *	0.11
% EPT	0.86 *	0.02	0.09
% exotic	0.33 *	0.05	-0.01
% gravel	0.06	-0.76 *	0.21
% large cobbles	0.52 *	-0.40 *	-0.08
% native	0.49 *	0.18	0.08
% pasture	-0.66 *	-0.15	0.13
% sand	-0.53 *	-0.13	-0.09
% silt	-0.55 *	0.52 *	-0.17
% small cobbles	0.53 *	-0.45 *	0.02
Altitude	0.25 *	-0.36 *	0.13
Bedrock	0.42 *	0.34 *	0.20
Boulders	0.54 *	-0.29 *	-0.02
Conductivity	-0.24 *	0.49 *	-0.21
Dissolved oxygen	0.42 *	-0.09	0.30 *
MCI	0.92 *	0.09	0.30 *
Periphyton	0.25 *	0.31 *	0.28 *
QMCI	0.93 *	0.09	0.20
Stability	-0.58 *	0.11	-0.45 *
Bed stability	-0.60 *	0.13	-0.24 *
Stream order	-0.06	0.63 *	0.03
Suspended solids	-0.08	0.31 *	0.38 *
Temperature	-0.23 *	0.01	0.25 *
Velocity	0.31 *	0.05	0.26 *
Width	-0.49 *	0.25 *	-0.04
Years since harvesting	0.58 *	0.15	0.10

Table 3.4. Taxa correlated (* = $P<0.05$) with Detrended Correspondence Analysis axes for macroinvertebrate communities collected from 12 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997 (includes pre and post harvest streams).

Taxon	Axis 1	Axis 2	Axis 3
Austroclima sepia	0.68 *	-0.15	0.04
Austroperla cyrene	0.43 *	-0.06	0.01
Ceratopogonidae	-0.46 *	-0.33 *	0.05
Chironominae	-0.56 *	-0.34 *	-0.28 *
Deleatidium sp.	0.74 *	0.12	0.14
Elmidae	0.64 *	-0.24 *	0.08
Empididae	0.47 *	-0.33 *	-0.05
Helicopsyche sp.	0.36 *	0.41 *	0.26 *
Ichthybotus hudsoni	0.42 *	-0.23 *	-0.03
Nesameletus sp.	0.44 *	-0.58 *	-0.19
Neurochorema confusum	-0.02	-0.44 *	0.16
Orthocladiinae sp. 1	-0.43 *	-0.36 *	-0.63 *
Orthocladiinae sp. 2	-0.28 *	-0.46 *	0.41 *
Ostracod sp. 1	-0.77 *	-0.32 *	0.03
Oxyethira albiceps	-0.48 *	-0.27 *	0.25 *
Paralimnophila skusei	-0.53 *	-0.16	0.30 *
Pisidium casertanum	-0.42 *	-0.10	0.40 *
Potamopyrgus antipodarum	-0.52 *	0.25 *	0.21
Ptilodactylidae	0.48 *	-0.13	0.04
Scirtidae	0.24 *	-0.44 *	-0.19
Sigara sp.	-0.56 *	-0.08	0.21
Small Oligochaetes	-0.76 *	-0.28 *	-0.25 *
Triplectides cephalotes	0.25 *	-0.40 *	-0.10
Zelandobius furcillatus	0.50 *	-0.27 *	-0.06
Zelandoperla decorata	0.40 *	0.43 *	0.21
Zephlebia dentata	0.68 *	-0.15	0

correlated with axis one (Table 3.3) indicating communities in streams draining harvested and pasture catchments had taxa characteristic of lower water quality (Table 3.4).

DISCUSSION

Harvesting resulted in a number of changes to stream physical characteristics e.g., sediment deposition, but had little effect on water chemistry. Harvesting effects on macroinvertebrate communities were also marked e.g., increases in total abundance and changes in community composition from taxa characteristic of clean water to those tolerant of lower water quality. Results were similar to those found in Chapter Two, suggesting that communities remain persistent throughout the year. However, pre and post harvest monitoring indicated that effects of harvesting and road construction on invertebrates were different.

Stream communities in pasture and native forest catchments had higher taxonomic richness than those in harvested and mature exotic forest, which were similar. And, taxonomic richness did not change before and after harvesting in Te Wera and Tawarau forest streams. This suggests that short term effects of forest harvesting per se do not affect diversity (Winterbourn and Rounick, 1985; Carlson et al., 1990; Growns and Davis, 1991; Brown et al., 1997). The higher diversity at pasture sites may have resulted from passive sampling of high numbers of invertebrates in these streams which inevitably results in higher numbers of taxa (Magurran, 1988).

However, the lack of changes in diversity does not mean that stream communities were unaffected by forest harvesting. Total abundances of invertebrates were higher in streams draining harvested catchments compared to those draining mature exotic forest (including pre and post harvest streams) indicating that harvesting resulted in increased secondary production. As in Chapter Two, primary production (often associated with increased secondary production (Behmer and Hawkins, 1986)) was lower in streams with harvested catchments (compared to mature exotic forest), and also decreased in winter. Nutrients were likely to be limiting in streams draining mature exotic forest, especially in Te Wera streams where conductivity was low. Sedimentation may be more limiting to periphyton biomass in streams draining harvested catchments where conductivity was relatively high (Friberg et al., 1997), and may signify that increases in numbers of invertebrates are due to increases in other food sources (i.e., terrestrial organic matter) (Hawkins et al., 1982; Reed et al., 1994; Quinn et al., 1997). Community composition also differed between mature and harvested exotic forest sites, being similar to stream communities in native forest and pasture catchments respectively.

In the pre and post harvest monitoring invertebrate responses were different for the stream in the harvested catchment (Te Wera) and the stream associated with road construction (Tawarau). The Te Wera stream exhibited community responses similar to those of communities in streams harvested one year previously e.g., decreases in the abundance of Ephemeroptera, Plecoptera and Coleoptera, and increases in the abundance of Crustacea and Mollusca. The Tawarau stream however, showed no significant changes in community structure. At this stream, road construction for harvesting started in July and was still continuing in December. There was a small (2 m) riparian strip of native scrub vegetation still intact which may have reduced harvesting effects. Furthermore, although road construction was occurring 2 m from the stream, the bedrock nature of the surrounding substrate would have lessened sediment input (Biggs and Gerbeaux, 1993).

Changes in substrate i.e., sedimentation after harvesting, may therefore be an important factor determining community composition (Richards et al., 1997), both directly through bed smothering and loss of suitable habitat, and indirectly through its influences on food sources (i.e., periphyton and particulate organic matter) (Ryder, 1989; Wood and Armitage, 1997). Effects may also be dependent on the extent or type of riparian cover as this, and other New Zealand studies (Harding and Winterbourn, 1995; Friberg et al., 1997) have found that communities in shaded streams (mature pine and native forest) were more similar to each other and communities in open streams (tussock, pasture and harvested forest) more similar.

The persistence of harvesting effects may therefore be dependent on several factors. Campbell and Doeg (1989) suggested that long term effects are a result of alteration of the stream riparian vegetation, while short term effects are mostly attributable to the impact of suspended or deposited sediment. As little sediment was deposited in the Tawarau stream, the short term effects of sediment deposition on stream invertebrates seems to have been avoided. However, sediment may also determine long term community structure through its influence on colonisation of invertebrates (i.e., sediment sensitive taxa tend to avoid sandy substrates (Ryder, 1989)). The difference between pre/post harvest streams (Te Wera and Tawarau) may be influenced by regional differences in geology. In Chapter Two, Tawarau stream communities showed little difference between harvested and mature exotic forest catchments, whereas in Te Wera, differences were greater. The lower potential for sedimentation in streams in the Tawarau region means effects of forest harvesting and road construction on stream communities will be less pronounced.

Neither of the post harvest streams exhibited a decrease in total abundance or taxonomic richness immediately after harvesting, indicating that if such a decrease occurred, it was quickly ameliorated and the community that re-established was determined by the longer term effects of riparian vegetation removal and sedimentation (Campbell and Doeg, 1989). However, there was a decrease in the abundance of certain taxa (e.g., Ephemeroptera, Plecoptera and Coleoptera) and a concomitant increase in other taxa (e.g., Crustacea and Oligochaeta) in the Te Wera stream. Therefore, although taxonomic richness and total abundance did not decrease initially, some taxa within the community sensitive to riparian vegetation removal and sedimentation, decreased in abundance and dominance.

Little seasonal influence on macroinvertebrate communities was evident and did not obscure the effects of forest harvesting or land use (results were similar to those found in Chapter Two). It seems therefore, that the time of year sampling is undertaken is not likely to affect results as communities remain relatively similar throughout the year. This is consistent with the general belief that New Zealand macroinvertebrate communities have poorly synchronised life cycles (Winterbourn, 1978; Towns, 1981).

However, Scott et al. (1994) and Scarsbrook and Townsend (1993) found that seasonal influences may affect macroinvertebrate abundance although not general community composition, consistent with the decrease in stream production in winter (Biggs, 1995). In contrast, the number of taxa increased slightly during winter and spring in this study. This increase was largely due to increases in the number of taxa in streams draining pasture and native forest, with numbers of taxa in streams draining mature and harvested exotic forest remaining constant throughout the year. This may correlate with the synchronisation of life cycles of some taxa in the community (Winterbourn, 1978; Towns, 1981).

Periphyton biomass was affected seasonally however, and when harvesting was conducted in winter, biomass was not significantly increased as may be expected after harvesting. In fact, at both post harvest and control post sampling occasions, periphyton biomass decreased. Seasonal influences may therefore determine the response of periphyton to harvesting i.e., in winter, temperatures and light levels may limit periphyton growth at open sites (Winterbourn, 1990). An increase in nutrients and light associated with harvesting may therefore not increase biomass as much as it would in summer. However, if riparian shading is not replaced by the preceding summer, effects on periphyton may become more prominent as light and temperature are no longer limiting (Winterbourn, 1990) and nutrient concentrations may be higher (Mosley and Rowe, 1981; Neal et al., 1992), increasing periphyton growth significantly. However, periphyton biomass in streams draining harvested exotic forest and pasture catchments may also be limited by other factors i.e., sedimentation.

In summary, forest harvesting was associated with lower stability and higher proportions of silt and sand substrates, but had little effect on water chemistry. Periphyton was more affected by seasonal changes in climate than forest harvesting, but macroinvertebrate communities showed significant differences between streams draining mature exotic forest and those draining harvested exotic forest i.e., increased total abundance and greater numbers of taxa tolerant of lower water quality following harvesting. Macroinvertebrate communities sampled within one year after harvesting reflected that of communities two to three years after harvesting (Chapter Two).

Changes in communities over time revealed that initial impacts of forest harvesting on communities (i.e., decreases in abundance and taxonomic richness), if these occur, are rapidly ameliorated i.e., within two months, and communities that re-establish are determined by the long term effects of forest harvesting i.e., removal of riparian vegetation and sedimentation. Effects of road construction on macroinvertebrates differed from that of harvesting the whole catchment, and this could be attributed to geological differences between regions and a small riparian strip.

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Appendix 3.1. Mean particulate organic matter (POM) (with range in parentheses) for each land use type from 12 streams in Lismore, Te Wera and Tawarau forests, sampled every two months from February to December 1997 (includes pre and post harvest streams). CPOM = Coarse POM, MPOM = Medium POM and TPOM = Total POM.

	CPOM ($g\ AFDW/0.1m^2$)	MPOM ($g\ AFDW/0.1m^2$)	TPOM ($g\ AFDW/0.1m^2$)
Land use type			
Mature	3.52 (0.26-8.19)	1.88 (0.06-5.72)	5.40 (0.34-13.91)
Harvested	2.46 (0.17-8.13)	2.72 (0.12-11.60)	5.18 (0.90-19.73)
Pasture	3.61 (0.58-6.43)	1.54 (0.45-2.70)	5.16 (1.04-8.41)
Native	2.84 (0.94-6.58)	0.59 (0.16-1.07)	3.43 (1.33-7.11)
Pre/post harvested			
Te Wera			
Control pre	4.86 (4.31-5.40)	1.54 (1.14-1.93)	6.39 (5.46-7.33)
Control post	5.39 (3.97-7.97)	2.13 (1.58-2.99)	7.52 (5.55-10.96)
Pre harvesting	3.53	1.10	4.63
Post harvesting	3.10 (0.71-4.57)	1.55 (0.98-2.49)	4.66 (1.06-6.42)
Tawarau			
Control pre	2.84 (2.82-2.86)	2.25 (1.71-2.80)	5.09 (4.52-5.66)
Control post	6.00 (3.99-8.19)	3.59 (1.33-5.72)	9.60 (5.57-11.93)
Pre harvesting	0.36 (0.25-0.47)	0.49 (0.31-0.67)	0.85 (0.56-1.15)
Post harvesting	0.43 (0.21-0.86)	0.09 (0.05-0.11)	0.51 (0.26-0.97)

CHAPTER FOUR

The effects of sedimentation on stony stream macroinvertebrates: a land use perspective

ABSTRACT

The effects of sedimentation on stream macroinvertebrate community composition was investigated in three streams draining exotic forest and two draining pasture in Pirongia forest, North Island, New Zealand. Three replicates of four treatments - control (natural stream), flow control (with a flow regulating net), light sedimentation and heavy sedimentation - were placed in each stream for 21 days. Flow regulating nets were placed at the top and bottom of the flow control, light and heavy sediment treatments to limit sediment suspension. Macroinvertebrates were sampled with a Surber sampler (0.1 m^2 area) at the end of the experiment.

Total abundance of macroinvertebrates decreased in light and especially heavy sediment treatments. The EPT taxa were most affected by sedimentation, and *Potamopyrgus antipodarum* and Elmidae least affected. Differences in community composition between control and sediment treatments were distinct within streams. However, land use had an overriding influence between streams. Invertebrates responded differently to sedimentation in the two land use types (exotic forest and pasture). Stream communities in pasture catchments showed no change with increasing sediment, whereas communities in exotic forest demonstrated decreases in the relative abundance of Ephemeroptera and increases in Diptera and Mollusca. Therefore, sedimentation, although important, is not likely to be the dominant determinant of community structure in these streams when land use is also incorporated.

Keywords: community composition, experiment, forestry, land use, macroinvertebrates, pasture, periphyton, sedimentation.

INTRODUCTION

One of the major effects of urban and agricultural development on New Zealand lotic systems is increased sedimentation. Activities such as farming, forestry and mining often result in the introduction of substantial amounts of sediment which can have highly detrimental effects on aquatic organisms (Wiederholm, 1984). Sedimentation can create changes in the substrate, through the filling of interstices and alteration of surface conditions (Graham, 1990). This can affect the benthos in several ways: decreasing the retention of organic material; restricting the flow of water to deeper substrates, with a concomitant decrease in oxygen availability; and decreasing bed stability (Ryder, 1989). Sedimentation may also increase abrasion directly through physical contact or indirectly through removal of periphyton. Sedimentation can further limit primary productivity by limiting light penetration via increased turbidity and bed smothering (Ryder, 1989; Graham, 1990; Wood and Armitage, 1997).

Increases in sediment deposition was identified as an important effect of forest harvesting in the survey conducted in January and February 1997 (Chapter Two), as well as in other studies (Graynoth, 1979; Murphy and Hall, 1981; Winterbourn, 1986; Davies and Nelson, 1994). However, other factors associated with harvesting, such as increased light, water temperature and nutrients, also influence invertebrate communities (Campbell and Doeg, 1989). Therefore, changes in community structure may not be attributed directly to the effects of sediment.

Ryan (1991) reviewed the environmental effects of sediment on New Zealand streams and emphasised the paucity of studies specifically on sedimentation. Most studies deal with anthropogenic sediment inputs (e.g., Graynoth, 1979), but few involve experimental studies of sediment effects on stream communities. Ryder's (1989) work is the most detailed example of a New Zealand experimental study, encompassing the short term effects of sediment addition on invertebrate drift, community composition and colonisation. Overseas, experimental studies are more numerous including both natural stream manipulations (e.g., Rosenberg and Wiens, 1978; Culp et al., 1986) and laboratory studies (e.g., Williams and Mundie, 1978), but all were conducted over a short period of time (i.e., one to two days) and examined immediate effects on invertebrate drift and community structure. Furthermore, most studies only involved the addition of small amounts of sediment which did not necessarily smother the stream bed (e.g., Culp et al., 1986).

Although the effect of sedimentation may be lessened in New Zealand by high rainfall which flushes out most sediment (Winterbourn et al., 1981), many anthropogenic sediment additions are continual, and in forestry where water velocity may be decreased by debri dams, sediment effects may be much more long term (Murphy and Hall, 1981; Davies and Nelson, 1994).

Macroinvertebrate community structure is also known to differ between land use types (e.g., Collier, 1995; Harding and Winterbourn, 1995; Townsend et al., 1997). Therefore, responses to sedimentation may differ between such communities where other factors (e.g., light, water temperature and nutrients) may interact or even override sedimentation in determining community structure (Wiederholm, 1984; Waters, 1995). The objective of this study is to experimentally assess the effects of both light and heavy deposits of sediment in stony streams draining mature exotic forest and pasture catchments over a three week period.

STUDY SITES

Five streams in Pirongia forest, located in the Waikato, New Zealand, were used for the experiment conducted between 22 November and 13 December 1997 (Table 4.1). Three streams had mature exotic forest catchments (*Pinus radiata*) (F1, F2 and F3) and two had pasture catchments (P1 and P2). All streams were second or third order and stream substrates were cobble and gravel dominated. Geology comprises Quaternary andesite, agglomerate and breccia (Department of Scientific and Industrial Research, 1984) and rainfall averages of 2000 mm per year (Kirkpatrick and Phillips, 1996). Of the three streams draining exotic forest, two were highly shaded and the third was more open (~40% cover) with an immediate riparian zone of grass (*Uncinia uncinata*), the pine being 10 m from the stream bank. The two streams draining pasture were both open

Table 4.1. Location and mean physiochemical characteristics (with range in parentheses) of five streams, for the experiment conducted in Pirongia forest between 22 November and 13 December 1997. Cond = conductivity, Temp = temperature; SS = suspended solids.

Site	NZMS 260 grid reference	Altitude (m)	Stream order	Depth (cm)	Width (m)	Velocity (m/s)	Cond (μS/cm)	Temp (°C)	SS (mg/l)
F1	S15: 931412	120	2	5.1 (0.0-12.5)	1.58	0.24 (0.15-0.31)	130	13.07	0.02
F2	Q15, R15: 901415	180	2	6.8 (2.2-11.5)	1.47	0.31 (0.17-0.43)	79	12.57	0.015
F3	Q15, R15: 898417	180	2	6.0 (2.6-13.7)	1.88	0.22 (0.15-0.31)	120	14.03	0.01
P1	S15: 913423	120	2	9.6 (2.5-10.3)	2.06	0.26 (0.15-0.52)	107	15.03	0.01
P2	R16: 900375	120	3	5.7 (2.5-17.6)	1.91	0.30 (0.15-0.34)	77	13.00	0.01

(~28% cover). Stream stability was similar for all sites (assessed visually using the Pfankuch Stability index (Pfankuch, 1975)), as was water chemistry (mean pH = 7.2, mean dissolved oxygen = 10.59 mg/l and Table 4.1).

METHODS

Experimental methodology

Each replicate comprised four treatments (each of 1 m^2 area) of light, heavy or no sedimentation, placed consecutively in a downstream direction (control (natural stream), flow control (velocity regulated), light sedimentation and heavy sedimentation) (Plate 4.1). The flow control, light and heavy sediment treatments all had flow regulating nets of nylon mesh (2 mm mesh size) upstream and downstream, that allowed water and invertebrate movement but slowed water velocity to limit sediment suspension (Plate 4.1). Each replicate was placed at 10 m intervals downstream.

Five litres of sand was added to the 1 m^2 area of the light sediment treatment and 20 litres to the heavy sediment treatment, the latter covering the substrate to a 5 cm depth. Sand was alluvial sediment collected from the Pirongia area and sieved to < 4 mm. The experiment was run for 21 days (22 November to 13 December 1997) with sediment being added every three to four days to maintain consistent levels within treatments.

PLATE 4.1

Plate 4.1. Experimental design of a replica at site F2 showing the four treatments (top), left to right: control, flow control, light and heavy sedimentation, separated by nylon mesh net; and a comparison of light and heavy sediment treatments (bottom), left in five streams in Pirongia forest for 21 days between 22 November and 13 December 1997.

One terra-cotta tile (32 mm^2) was placed in each treatment to determine the effects of sedimentation on periphyton colonisation.

Velocity (using a velocity head rod), depth, width, conductivity (using a ORION model 122 conductivity meter) and dissolved oxygen (using a YSI model 58 dissolved oxygen meter) were measured at each stream on day 1, 14 and 21. Water samples were also collected for pH (using a ORION model 250A pH meter) and suspended solids analysis. The latter was determined by the weight of solids collected after filtering a sample through a Whatman GF/C filter (pore size = 0.7 μm) and drying overnight at 105 °C.

On day 21, one Surber sample (area 0.1 m^2, 250 μm mesh net) was collected from the centre of each treatment and stored in 70% alcohol until sorting. Samples were sieved through 500 μm mesh and macroinvertebrates sorted from particulate matter in a white tray, then identified and enumerated using a dissection microscope (×40). Macroinvertebrates were identified to the lowest possible taxonomic level with keys from McFarlane (1951), Winterbourn (1973), Cowley (1978), and Winterbourn and Gregson (1989). Taxa that could not be identified to species level were separated into apparent morphospecies. Periphyton tiles were frozen until pigment extraction in 90% acetone at 5 °C for 24 hours. Absorbency was read with a Jenway 6105 UV/Visible Spectrophotometer at 410, 430, 665 and 750 nm. Pigment concentration was calculated following Moss (1967a and b) and adjusted for surface area following Graham et al. (1988).

Statistical analyses

Statistical differences between treatments (land use, stream, treatment) were analysed using a three-way Analysis of Variance (ANOVA) and Duncan multiple range a posteriori means test with SAS (SAS, 1989). Land use and treatment were treated as fixed effects and streams within each group as random. Data were log transformed ($\log_{10}(x+1)$) before analysis if necessary, to increase variance homogeneity.

Classification of experimental communities was carried out with Multi-Response Permutation Procedures (MRPP) and ordination with Detrended Correspondence Analysis (DECORANA) (log transformed), both using the PC-ORD statistical package (McCune and Mefford, 1995). Groups for the MRPP were classified according to treatment (controls, light and heavy sediment) and land use (forest and pasture).

RESULTS

Streams were similar in all physical and chemical characteristics (Table 4.1). Despite the flow control barriers, treatments did not differ significantly in velocity ($F_{3,49}$=2.31, P=0.09).

Periphyton biomass on tiles declined with amount of sediment added but was not significantly different between treatments ($F_{3,49}$=1.30, P=0.28) (Fig. 4.1) or land use types ($F_{1,49}$=0.00, P=0.97). However, variation between streams was high ($F_{4,49}$=4.80, P=0.005) and may explain the lack of significance.

Macroinvertebrates

The most common taxa in all streams were *Deleatidium* sp. and Elmidae. *Deleatidium* sp. tended to decrease in abundance with light and heavy sediment ($F_{3,12}$=7.61, P=0.004), whereas abundances of Elmidae larvae were not affected by sediment deposition ($F_{3,12}$=0.80, P=0.80). Of the other common taxa, *Helicopsyche* sp. were rare in heavy sediment treatments ($F_{3,12}$=3.60, P=0.05) and *Zephlebia dentata* less common in light and heavy treatments ($F_{3,12}$=3.86, P=0.04). *Potamopyrgus antipodarum* tended to occur in higher numbers in pasture streams ($F_{1,12}$=78.01, P<0.001) but did not change in abundance with treatment type ($F_{3,12}$=0.35, P=0.79).

Total number of individuals was greatest in the flow control treatment ($F_{3,49}$=12.08, P<0.001), with both the control and flow control treatments having higher densities than either sediment treatment (Fig. 4.2a). Differences also occurred in the number of

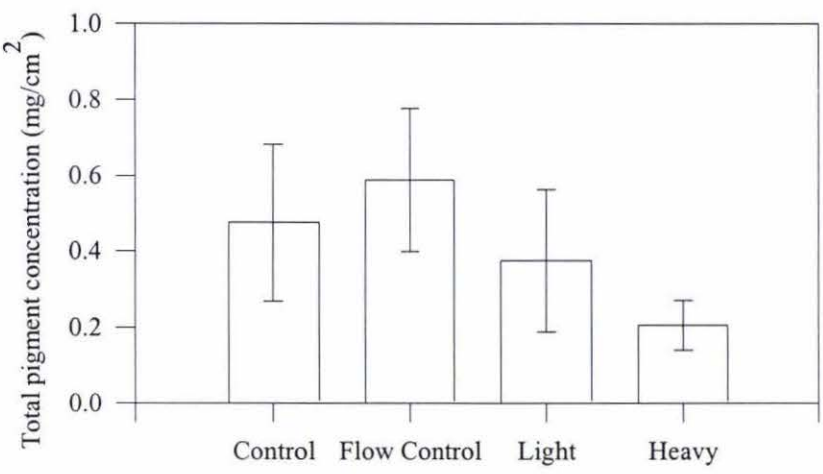

Figure 4.1. Mean total pigment (chlorophyll *a* and phaeophytin) concentrations (± 1 SE) for terra-cotta tiles collected in Pirongia forest after 21 days in streams with heavy, light or no sediment addition (with flow control and without) between 22 November and 13 December 1997.

individuals between land use types, with pasture streams not surprisingly, having higher numbers ($F_{1,49}$=80.12, P<0.001).

Number of taxa also differed significantly between treatments ($F_{3,49}$=5.73, P=0.002) and followed a similar pattern to total abundance, with the heavy sediment treatment having the lowest number of taxa (Fig. 4.2b). The other treatments were not significantly different. Other measures of diversity were not affected by sediment deposition ($F_{3,46}$=1.10, P=0.38; $F_{3,49}$=4.38, P=0.26; $F_{3,49}$=0.82, P=0.49, for Margalef's, Berger-Parker and Simpson's indices respectively).

Changes in relative abundance indicated a switch from an Ephemeroptera ($F_{3,12}$=10.73, P=0.001) to Coleoptera ($F_{3,12}$=4.59, P=0.02) dominated community with increasing amounts of sediment (Fig. 4.3). Diptera also increased in relative abundance in both light and heavy sediment treatments, although differences were not significant ($F_{3,12}$=2.25, P=0.13). Percentage EPT consequently declined ($F_{3,12}$=13.53, P<0.001) with sediment addition. The QMCI also declined significantly in sediment treatments ($F_{3,12}$=8.18, P=0.003) but the MCI did not ($F_{3,12}$=0.13, P=0.94) (Fig. 4.4).

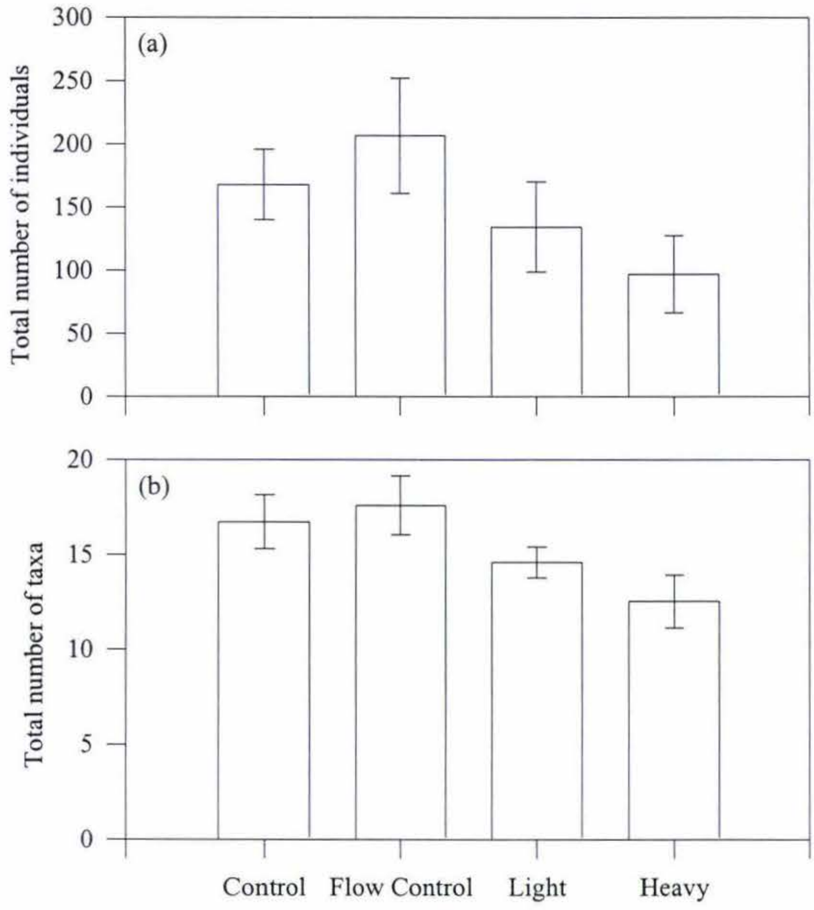

Figure 4.2. Mean total number of individuals (± 1 SE) (a) and total number of taxa (± 1 SE) (b) for samples collected in Pirongia forest after 21 days in streams with heavy, light or no sediment addition (with flow control and without) between 22 November and 13 December 1997.

For the above biotic and diversity indices there were no significant interactions between land use type and degree of sedimentation, suggesting that responses of communities to sedimentation did not differ between land use types.

Ordination of sites separated all P1 treatments from the rest (Fig. 4.5). Most heavy and light sediment treatments were higher on axis two than their respective control and flow control treatments. However, distinct divisions between treatments were not perceptible due to communities at sites F1 and P1 being similar irrespective of treatment type. Sites F2 and P2 had communities most different between sediment and control treatments and communities at site F3 were also different, however, the communities in the flow

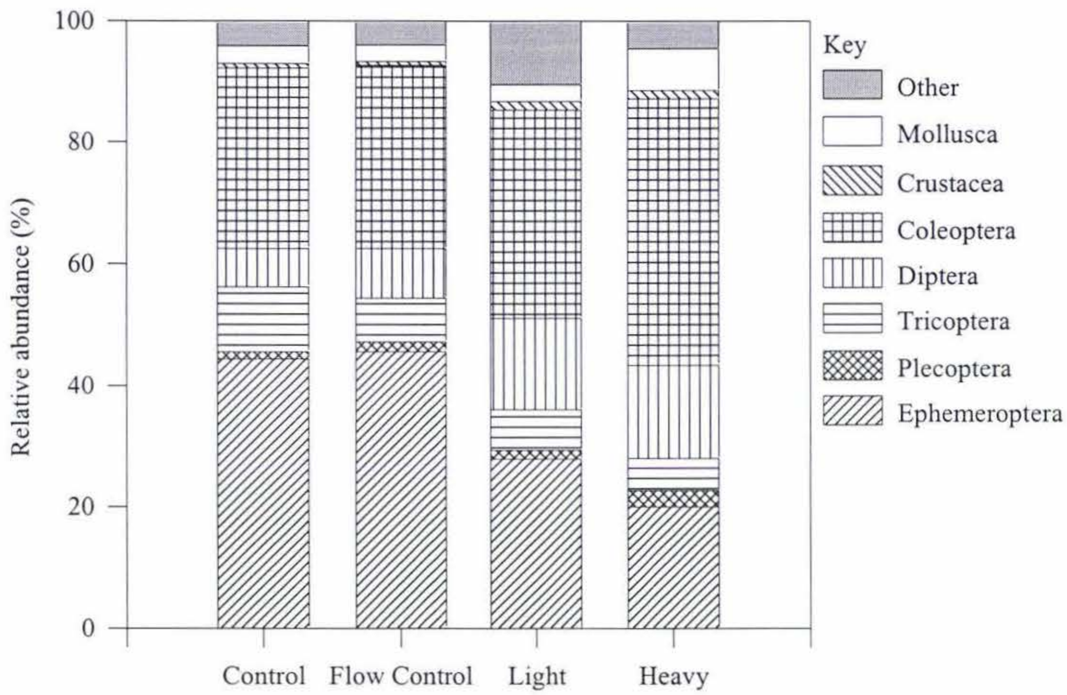

Figure 4.3. Mean relative abundance of taxonomic orders for each treatment from macroinvertebrate samples collected in Pirongia forest after 21 days in streams with heavy, light or no sediment addition (with flow control and without) between 22 November and 13 December 1997.

control treatment occurred closer to those in light and heavy sediment treatments than to the other control treatment. Axis one accounted for 65% of the variation in the data and axis two, 21.3%.

Most correlations of environmental variables with axis one were determined by the characteristics of site P1 (Table 4.2) e.g., increased temperatures, decreased stability and lower water quality (biotic indices). Taxa positively correlated with axis one were those most distinct in the community at P1 compared to all other sites e.g., Elmidae, Eriopterini sp. 2, *Nesameletus* sp., *Oecetis unicolor* and *Zelandobius furcillatus* (Table 4.3). Axis two was correlated with several environmental variables (Table 4.2) although the volume of sand added was not significantly correlated ($P<0.05$). Stream communities higher on axis two were associated with higher conductivity, lower velocity and more exotic riparian vegetation and cover. *Deleatidium* sp., *Potamopyrgus antipodarum* and *Olinga feredayi* were more common lower on axis two, and

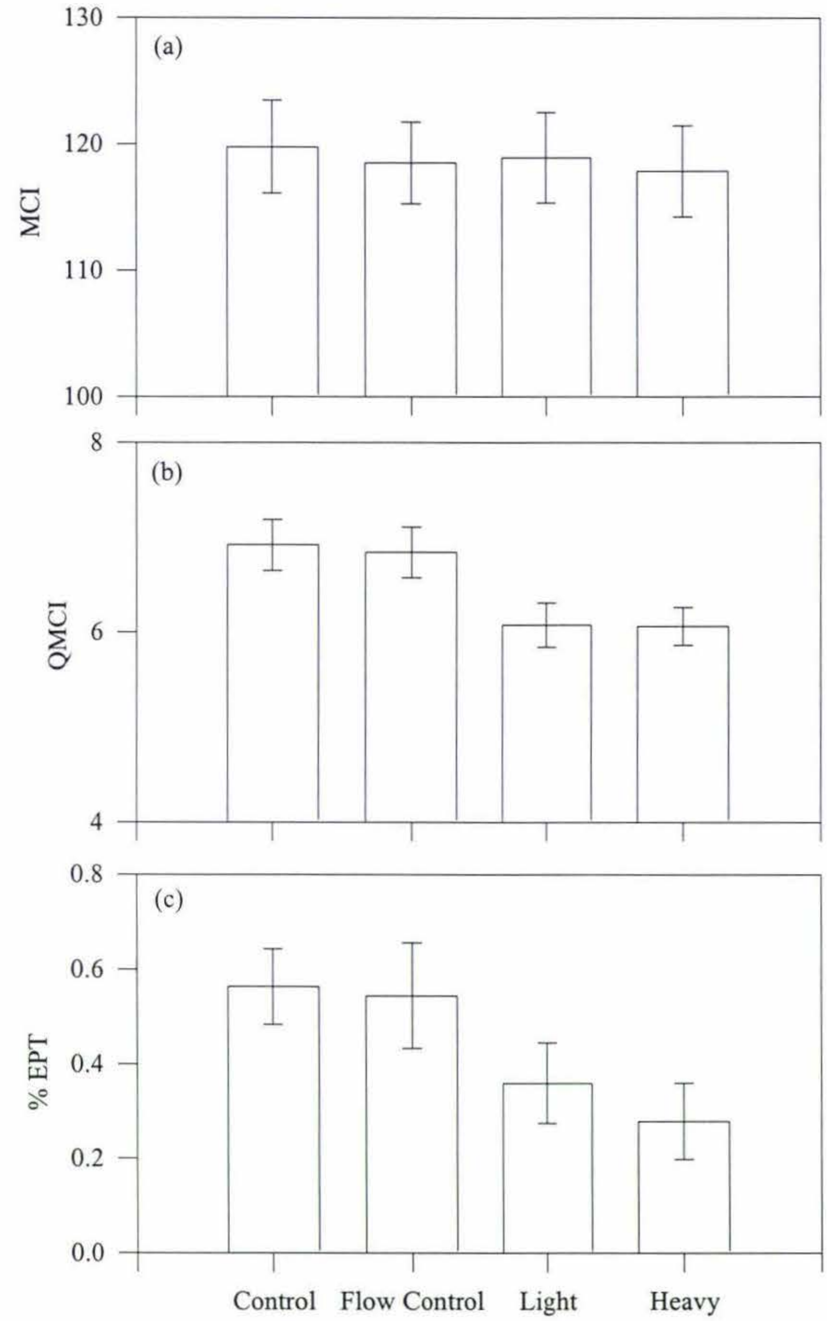

Figure 4.4. Mean biotic index scores (± 1 SE) for (a) the Macroinvertebrate Community Index (MCI), (b) the Quantitative Macroinvertebrate Community Index (QMCI) and (c) Percentage of Ephemeroptera, Plecoptera and Tricoptera (% EPT) for samples collected in Pirongia forest after 21 days in streams with heavy, light or no sediment addition (with flow control and without) between 22 November and 13 December 1997.

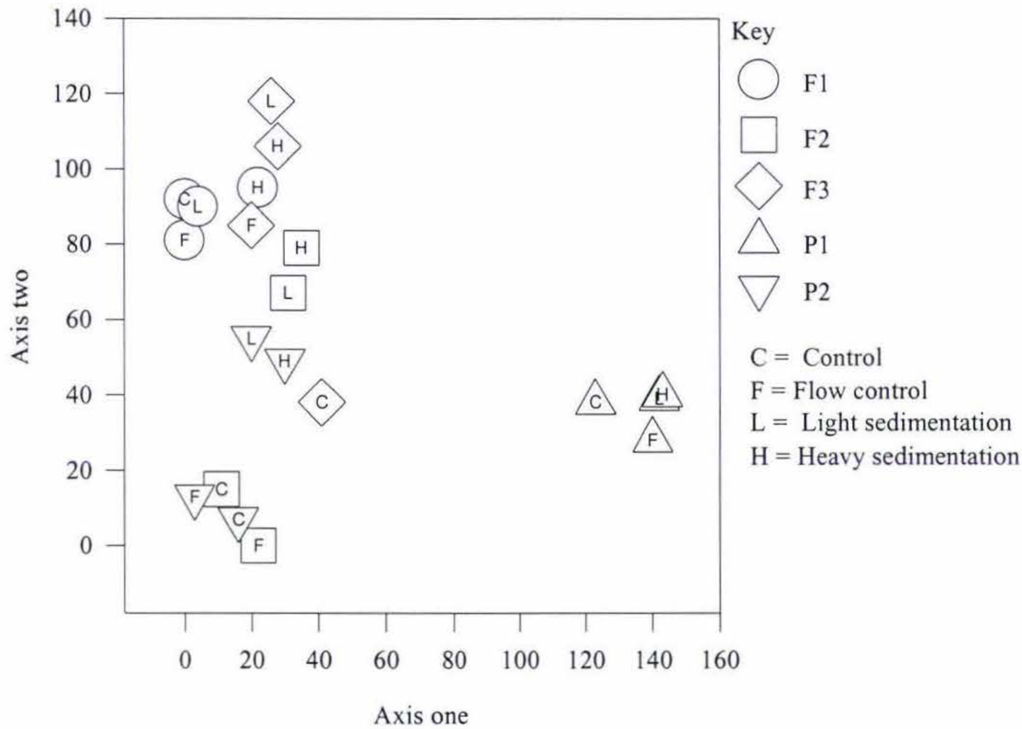

Figure 4.5. Axis one of a Detrended Correspondence Analysis as a function of axis two for macroinvertebrate communities collected in Pirongia forest after 21 days in streams with heavy, light or no sediment addition (with flow control and without) between 22 November and 13 December 1997.

Ceratopogonidae and Diamesinae more common higher on axis two (Table 4.3). Land use may have been more influential on stream communities than sedimentation. However, within each stream, sedimentation did affect the community.

The MRPP revealed that communities were significantly different between treatments overall (i.e., pasture and forest sites combined) (R=0.04, P<0.001). However, when forest and pasture sites were analysed separately, stream communities in exotic forest were different between sediment treatments (R=0.06, P<0.001) but stream communities draining pasture were not (R=-0.008, P=0.55). This indicates that stream communities in mature exotic forest showed greater changes in response to sedimentation whereas streams draining pasture were unaffected.

Table 4.2. Variables correlated (* = *P*<0.05) with Detrended Correspondence Analysis axes for macroinvertebrate communities collected in Pirongia forest after 21 days in streams with heavy, light or no sedimentation (with flow control and without) between 22 November and 13 December 1997.

Parameter	Axis 1	Axis 2	Axis 3
% cover	-0.60 *	0.66 *	-0.16
% EPT	-0.73 *	-0.24	-0.60 *
% exotic	-0.55 *	0.71 *	-0.37
% pasture	0.55 *	-0.71 *	0.37
Altitude	-0.27	0.16	-0.44 *
Conductivity	0.05	0.65 *	-0.08
Depth	0.45 *	-0.35	-0.18
Dissolved oxygen	-0.78 *	-0.06	-0.21
MCI	-0.75 *	0.22	-0.29
Periphyton	0.20	-0.10	-0.35
pH	-0.26	-0.30	0.53 *
QMCI	-0.61 *	-0.13	-0.78 *
Stability	0.88 *	-0.17	0.22
Stream order	-0.26	-0.38	0.49 *
Suspended solids	-0.47 *	0.24	-0.23
Temperature	0.85 *	-0.00	0.22
Velocity	-0.04	-0.80 *	-0.29
Volume of sand added	0.12	0.34	0.53 *
Width	0.65 *	-0.20	0.51 *

Table 4.3. Taxa correlated (* = *P*<0.05) with Detrended Correspondence Analysis axes for macroinvertebrate communities collected in Pirongia forest after 21 days in streams with heavy, light or no sedimentation (with flow control and without) between 22 November and 13 December 1997.

Taxon	Axis 1	Axis 2	Axis 3
Adult Elmidae	0.53 *	-0.46 *	0.13
Ameletopsis perscitus	-0.38	0.04 *	-0.45 *
Aphrophila neozelandica	0.29	-0.43 *	0.12
Atalophlebioides cromwelli	0.46 *	-0.19	0.03
Beraeoptera roria	-0.23	-0.48 *	0.35
Ceratopogonidae	-0.06	0.48 *	0.20
Chironominae	0.67 *	-0.33	0.27
Coloburiscus humeralis	-0.52 *	0.08	-0.18
Deleatidium sp.	-0.15	-0.84 *	-0.48 *
Diamesinae	-0.15	0.48 *	0.01
Elmidae	0.95 *	-0.36	0.13
Empididae	-0.08	-0.14	-0.54 *
Eriopterini sp. 2	0.81 *	-0.25	0.30
Ferrissia dohrnianus	0.46 *	-0.19	0.03
Homalaena sp.	-0.35	-0.18	-0.57 *
Ichthybotus hudsoni	-0.21	-0.32	-0.67 *
Nesameletus sp.	0.83 *	-0.21	0.01
Oecetis unicolour	0.91 *	-0.26	0.17
Olinga feredayi	0.31	-0.50 *	0.38
Oniscigaster wakefieldi	0.56 *	-0.17	0.08
Orchymontia sp.	-0.22	-0.43 *	0.39
Orthocladiinae sp. 1	-0.23	-0.33	0.51 *
Orthocladiinae sp. 2	-0.28	-0.52 *	0.16
Orthopsyche fimbriata	-0.33	-0.44 *	0.13
Potamopyrgus antipodarum	0.04	-0.53 *	0.52 *
Pycnocentrodes sp.	0.72 *	-0.23	0.10
Simuliidae	-0.24	-0.49 *	0.27
Tabanidae	0.61 *	-0.16	0.15
Triplectides obsoleta/dolichos	0.46 *	-0.19	0.03
Zelandobius furcillatus	0.93 *	-0.25	0.15
Zephlebia dentata	-0.55 *	0.22	-0.3

DISCUSSION

The results of this experiment indicate that heavy sediment deposition has marked effects on macroinvertebrate communities primarily by causing significant decreases in total abundance and number of taxa, and changes in community composition. However, invertebrate responses to light sedimentation were more variable. Generally abundances were reduced but not to the same extent.

Decreases in the total number of individuals and taxa as well as the relative abundance of Ephemeroptera due to sedimentation may be a result of suffocation of the stream bed by sediment, the filling of interstitial spaces and reduced oxygen availability to subsurface organisms (Ryan, 1991; Schalchli, 1992; Waters, 1995). Initial input of sediment has been known to result in an immediate increase in the drift of invertebrates, leading to a reduction in density (Rosenberg and Wiens, 1978; Ryder, 1989; Doeg and Milledge, 1991). Once sediment has settled, recolonisation of pre-existing taxa may be limited by the lack of suitable habitat and filling of interstices, reducing potential colonisation from deeper substrates and drifting invertebrates (McClelland and Brusven, 1980; Ryder, 1989). This would keep numbers of invertebrates low in heavy sediment treatments. Recolonisation in light sediment treatments may be greater due to the availability of suitable substrate (i.e., no smothering of the stream bed) although interstices may still become clogged keeping numbers below that of control treatments.

Increased abrasion of sediment on invertebrates, directly or via influence on periphyton (Graham, 1990), may have been a less important factor in this study as velocity in the treatments was low. However, some saltation (i.e., sand moving along the stream bed) must have occurred considering the loss of sediment during the experiment, abrading invertebrates and reducing the ability for periphyton to colonise unstable sand surfaces. Sediment may also have affected periphyton colonisation and production through smothering of the surface layer (Waters, 1995), as colonisation of periphyton tiles was lower in heavy sediment than other treatments (although not significantly). This low periphyton abundance may be important to grazing invertebrates (Ryder, 1989).

Taxa affected by sedimentation in this study were similar to those affected in experiments conducted by Ryder (1989). *Deleatidium* sp. were reduced in abundance in both studies in response to sediment deposition, whereas *Potamopyrgus antipodarum*, Elmidae larvae and Oligochaeta were all unaffected. Elmidae larvae were also more common in the presence of fine silt in a colonisation study by Ryder (1989). The EPT taxa seem most highly affected by sedimentation of gravel substrates (Waters, 1995) and may have attributed to most of the decrease in total abundance of individuals in this study.

Community composition in this study appears primarily determined by land use which influences riparian cover, light penetration and water temperatures (Tait et al., 1994) (Table 4.2), all of which have been identified as important in Chapter Two and other studies (e.g., Hawkins et al., 1982; Noel et al., 1986; Quinn and Hickey, 1990; Harding and Winterbourn, 1995). However, although there was some variation between streams, within streams community composition was affected by sedimentation as indicated by the MRPP and ordination. Sediment influences are likely to have an effect on community structure within streams where the influence of land use does not override the influence of sedimentation (Quinn et al., 1997).

The QMCI and % EPT also differed between control and sediment treatments. However, the MCI showed no differences. This is probably due to the decrease in abundance of invertebrates in light and heavy sediment treatments rather than their extinction; the MCI accounts only for presence or absence of a taxa whereas the QMCI and % EPT are quantitative. This illustrates the inability of the MCI in detecting sediment effects (Stark, 1993), of which the primary effect on invertebrates is decreased abundance.

Changes in community composition have been found by many studies encompassing long term effects of sediment deposition (Wiederholm, 1984). Characteristic taxa that usually increase in numbers with sedimentation are Oligochaeta, Mollusca and Diptera (Waters, 1995). Significant increases in these invertebrate groups did not occur during this experiment, although the relative abundance of Diptera and Mollusca did increase in sediment treatments. The duration of this experiment may not have been long enough to allow for greater domination of such taxa. And, the area of the treatments may also have been too small to simulate the generally catastrophic effects of major sedimentation needed to establish distinctly different communities. Most short term studies have found either decreases in abundance of benthic invertebrates and/or increases in invertebrate drift (e.g., Rabeni and Minshall, 1977; Rosenberg and Wiens, 1978; Culp et al., 1986; Ryder, 1989). These initial reductions in abundance were attributed to saltation effects on the surface dwelling taxa (Culp et al., 1986), and over a longer term (as in this experiment) may be influenced by the retention of organic matter which can be decreased by the presence of fine sediment (Rabeni and Minshall, 1977; Ryder, 1989). This may therefore influence the colonisation rates of sediment tolerant taxa, as food, including both organic detritus and periphyton, has been found to influence numbers (Hawkins et al., 1982; Reed et al., 1994; Friberg et al., 1997; Quinn et al., 1997) and colonisation of invertebrates (Rabeni and Minshall, 1977; Guy, 1997).

Invertebrate responses to sedimentation in streams draining pasture were different from responses of invertebrates in streams draining exotic forest. For example, communities in streams draining pasture showed no significant difference between sediment and control treatments, contrary to invertebrate responses in streams draining exotic forest. Site P1, which had a different community to that of all other streams, was likely to be already affected by agricultural land use (less riparian cover, greater stream bank instability, increased light, water temperature and nutrients), and consequently did not show a significant response to sedimentation i.e., its community had a greater proportion of sediment tolerant taxa already present (e.g., *Potamopyrgus antipodarum* and Elmidae larvae). Site P2 was surrounded by less intensive agriculture (sheep farming) and may not have been as impacted as P1, hence a greater response to sedimentation. Communities in mature exotic forest typically have more taxa sensitive to sedimentation (e.g., Ephemeroptera). This indicates that although other factors, such as removal of riparian vegetation, may be important regulators of community structure, sedimentation will influence the community in a less impacted stream. Therefore, substrate characteristics are important in determining stream communities but may be overridden by other environmental factors.

In summary, the primary effects of sedimentation on stream invertebrate communities in this experiment were significant decreases in abundance and number of taxa and within streams, changes in community composition, with EPT taxa being most affected. This could be attributed to smothering of the stream bed, filling of interstices, abrasion and reduction in food for grazing and other surface dwelling invertebrates. With respect to forest harvesting, this result indicates that sedimentation is an important determinant of community composition. However, effects of land use (pasture versus forest) may reduce the response of invertebrate communities to sediment in streams already impacted by other factors i.e., agriculture. Therefore, other effects of forest harvesting may also be important in determining community structure in affected streams e.g., increased light, water temperature and nutrients.

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CHAPTER FIVE

General discussion

GENERAL DISCUSSION

Using macroinvertebrates as indicators of water quality, it is evident that forestry practices, especially harvesting, do influence water quality. Present findings suggest that macroinvertebrate communities in streams draining mature exotic forest consist of taxa that prefer pristine conditions e.g., Ephemeroptera, Plecoptera and Coleoptera, whereas streams draining harvested exotic forest have communities consisting mainly of pollution tolerant taxa e.g., Crustacea and Mollusca. Although the length of time harvesting effects persist could not be determined exactly, changes in invertebrate community composition were immediate from the commencement of harvesting and tended to indicate a gradual reversion to the pre harvest community within a time frame of six to 10 years after harvesting.

When considered over several regions however, differences in stream communities between mature and harvested exotic forest were not clearly defined. Variation in community responses between regions masked clear differences i.e., stream invertebrate communities in Lismore and Te Wera forest showed stronger responses to forest harvesting and land use type, whereas stream communities in Tawarau and Pirongia forest showed little or no change in community composition. Regional factors (i.e., geology and climate) may not have an overriding influence on invertebrate communities, but such factors seem important in determining community responses to harvesting. At large spatial scales (between regions) the influence of geology and climate may become increasingly important in structuring communities and may therefore mask clear differences in communities between streams in mature exotic forest and those in harvested exotic forest. However, within regions, communities will be similarly influenced by geology and climate, and the effects of forest harvesting and other land use practices may then determine community structure (Harding et al., 1997). This highlights the importance of spatial scale when comparing stream communities between land use types, in contrast to Townsend et al. (1997) who found no great changes in relationships between land use and invertebrate communities at different spatial scales. Rather, findings were more influenced by the statistical robustness of replication.

The most important effects of forest harvesting identified in this study were sedimentation and removal of riparian vegetation. As the extent of sedimentation is largely determined by geological characteristics of the region (Richards et al., 1996; Richards et al., 1997), this may be the most important factor determining community changes in relation to forest harvesting i.e., greater potential for sedimentation results in greater changes in community composition. However, the effects of land use may be overriding the effects of sedimentation i.e., streams draining pasture showed little change in communities in response to sedimentation. This was also the case for Quinn et al. (1997), who determined that sedimentation was not a key driver of change in invertebrate communities or was counteracted by other factors, such as removal of riparian vegetation. However, other studies (Ryder, 1989; Richards et al., 1993; Storey and Cowley, 1997) determined that substrate and fine sediment were the most influential factors determining invertebrate communities. In this study, response of invertebrates to sedimentation was greater in less impacted streams i.e., those draining mature exotic forest. Communities in streams draining pasture in this study characteristically consisted of sediment and pollution tolerant taxa which will not be greatly affected by increased sedimentation. Therefore, the composition of the community may influence its response to sedimentation. When catchments are harvested sedimentation may be more important initially, decreasing the abundance of taxa sensitive to sedimentation (i.e., EPT taxa) and consequently changing the community from one dominated by Ephemeroptera to one dominated by Coleoptera and Diptera. Long term retention of sediment and removal of riparian vegetation may then determine the community that prevails through influences on habitat and food resources (Rabeni and Minshall, 1977; Ryder, 1989).

Temporal variation during the one year monitoring period did not greatly influence macroinvertebrate communities in these streams. And although streams in different land use types are exposed differently to variation in climate (i.e., native and mature exotic forest may buffer streams from seasonal changes (Hildrew and Giller, 1994)), no differences in community responses between land use types occurred. This persistence of communities through minor predictable environmental changes has often been demonstrated in lotic systems (Hildrew and Giller, 1994), and demonstrates the usefulness of macroinvertebrate communities in New Zealand streams as environmental monitoring tools at all times of the year (Richards and Minshall, 1992; Harding and Winterbourn, 1997). However, large disturbances such as forest harvesting may change the community on a long term basis.

It is evident that to effectively manage forestry to reduce or minimise adverse effects on streams, factors at several spatial scales (i.e., within and between regions) need to be considered, highlighting the need to assess regions separately. Regional assessments enable identification of areas of greater potential environmental risk (i.e., through erosion potential) and should be incorporated into management practices to minimise adverse effects of harvesting on stream communities. Harding et al. (1997) suggest that regions (ecoregions) may act as a predictive tool for indicating likely changes in habitat quality and fauna resulting from alterations in management practices, and much of the current research in freshwater ecology is being directed toward the determination of ecoregions for management purposes (e.g., Biggs et al., 1990; Harding and Winterbourn, 1997). This study indicates that such regional classifications are necessary and applicable to environmental management today.

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CHAPTER SIX

Management Report

The effects of exotic forestry practices on stream communities

for Rayonier New Zealand Ltd.

1998

EXECUTIVE SUMMARY

1. A survey of 42 streams draining exotic forest, native forest or pasture in Lismore, Tawarau, Pirongia and Te Wera forests, was conducted in January and February 1997, to assess the effects of exotic forest harvesting on stream macroinvertebrate communities.

2. Monitoring of streams every two months using a subset of the survey sites (12 streams) was conducted from February to December 1997, to assess the immediate effects of harvesting and road construction, and determine seasonal influences such as changes in light, temperature and flood regimes, on macroinvertebrate communities.

3. An experiment was conducted in five streams in Pirongia forest to determine the effects of sedimentation on macroinvertebrates in mature exotic forest and pasture streams.

4. Results indicated that macroinvertebrate communities in harvested streams were different from those in mature exotic forest streams. Harvested stream invertebrate communities were most similar to pasture stream communities, and mature exotic forest stream communities were most similar to native stream communities. Harvested and pasture streams had lower water quality than mature exotic and native forest streams.

5. Responses of communities to harvesting differed between regions, with Lismore and Te Wera showing the greatest differences in invertebrate communities between harvested and mature exotic forest streams.

6. The factors associated with harvesting, and identified as being of potential importance in influencing macroinvertebrate communities, were a combination of the amount of sediment in streams, amount of slash entering streams, and the

percentage of vegetation covering the stream, which in turn dictated stream light and water temperature regimes.

7. Recommendations to minimise effects include: reducing slash input to streams; minimising disturbance to stream bank vegetation and where feasible planting riparian buffer strips; minimising the number of stream crossings and roads running parallel to streams; and further long term monitoring of harvesting effects.

OBJECTIVES

1. To investigate the long term effects of exotic forestry on streams in different regions, using freshwater macroinvertebrate communities as indicators of stream water quality. This study was carried out in four west coast forests in central North Island - Lismore, Tawarau, Pirongia and Te Wera. Streams draining native and pasture catchments were included as a comparison with exotic forestry.

2. To investigate the immediate effects of forest harvesting and road construction (before and after harvest monitoring) on stream macroinvertebrate communities and to assess whether seasonal influences impinge on these effects.

3. To test experimentally the effects of sedimentation on macroinvertebrates. Sedimentation was established as an effect of forest harvesting and was also identified as a potential influence on macroinvertebrate communities.

4. To develop recommendations based on findings from objectives one, two and three, to prevent or minimise harvesting effects in the future.

6.1 INTRODUCTION

The primary emphasis of this report is to attempt to identify the most important effects of forest harvesting on streams as indicated by invertebrate communities, and to make recommendations on the management of forestry operations that may minimise adverse effects on streams. Streams and rivers often reflect the health of the catchments they drain, and influence their invertebrate communities. Stream invertebrate communities are frequently used as indicators of water quality (Rosenberg and Resh, 1992), and this is the approach adopted here.

Regional factors such as geology and climate may be an important influence on macroinvertebrate communities (Friberg et al., 1997) and such influences may also determine the extent of a disturbance such as forest harvesting. Little research on effects of forestry has been conducted around central North Island. Therefore, if regional influences are important, effects identified from studies in other parts of New Zealand may differ from those in central North Island. Monitoring of forests in these regions is therefore important for resource management.

Monitoring was conducted in four exotic forests between January and December 1997, and spanned three regions (Wanganui, Taranaki and Waikato) to establish whether there were different responses of invertebrate communities to forest harvesting between these regions. The effects of forest harvesting were then identified and, drawing from the general literature on stream management and the present results, recommendations made on how to prevent or minimise adverse effects in the future.

6.2 STUDY SITES

Streams were sampled from January to December 1997, in or around four exotic forests, Lismore, Tawarau, Pirongia and Te Wera, on the west coast of central North Island, New Zealand (Fig. 6.1, Appendix 6.1). All streams were of similar size with

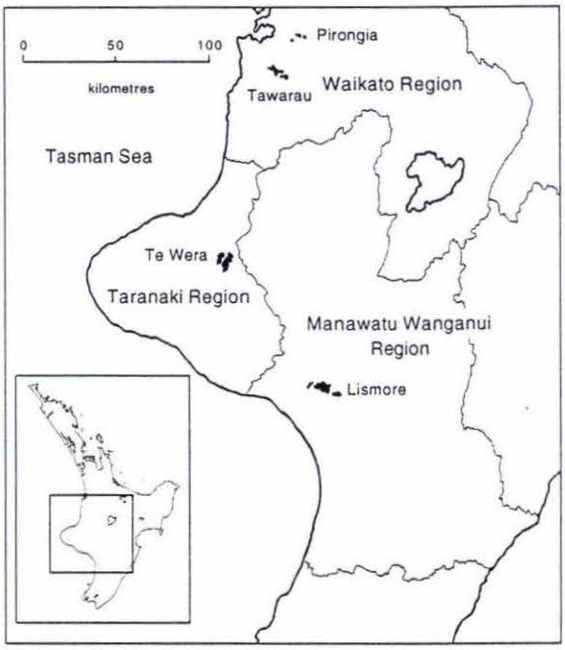


Figure 6.1. Location of the four exotic forests, Lismore, Tawarau, Pirongia and Te Wera, within which streams were sampled from January to December 1997, to assess effects of forest harvesting on stream communities.

catchments which drained at least 90% exotic plantations of *Pinus radiata,* 90% native forest or 90% pasture. Details of sites are given in Appendix 6.1, and number of streams per land use in each region in Table 6.1. Native and pasture streams were included to compare the effects of other land use types. Streams draining native forest represented the "clean water" condition, characteristic of undisturbed land.

The geology of Lismore, Tawarau and Te Wera forests were similar, consisting of siltstone, sandstone and limestone (Department of Scientific and Industrial Research, 1984). Pirongia forest had a distinctively different geology comprising of andesite, agglomerate and breccia, causing its streams to be predominantly cobble compared to the sandy bottomed streams of the other forests. Lismore forest had the lowest rainfall (mean annual rainfall = 1000 mm (Kirkpatrick and Phillips, 1996)), while Pirongia and

Tawarau forests had the highest (2000-2400 mm). The annual rainfall in Te Wera forest was intermediate between these (mean = 1400 mm).

Forests were clear cut and harvesting was conducted using cable logging methods. This method causes less impact to the environment compared to other ground based logging methods (Visser and Smith, 1993). However, forests were harvested right up to stream banks and often large amounts of woody debris were left covering streams and surrounding banks. Harvesting can also continue for long periods of time within a catchment, causing continual disturbance.

6.3 METHODS

6.3.1 Objective 1: Broad scale effects of forestry on invertebrates

Forty two streams were sampled during January and February 1997, in or around four exotic forests, Lismore, Tawarau, Pirongia and Te Wera, on the west coast of central North Island, New Zealand (Fig. 6.1, Appendix 6.1). Exotic forest sites were selected according to forest age to assess whether there were differences between streams in recently harvested and mature exotic forest. Catchments with forests harvested less than six years before sampling are referred to as "harvested" (canopy cover was not highly developed within this time frame), and catchments with forests greater than 10 years old are referred to as "mature".

At each site four macroinvertebrate samples (250 μm-mesh kick net) were collected from a 0.1 m^2 area on the stream bed. Macroinvertebrates (animals > 500 μm) were counted and identified. Four small stones were collected for measurement of algal (periphyton) biomass (important invertebrate food supply). Pigments were extracted in 90% acetone at 5 °C for 24 hours and concentration of chlorophyll determined (Moss 1967a and b). Organic matter from macroinvertebrate samples (also an important food source for invertebrates, originating from surrounding stream bank vegetation) was divided into two size groups, coarse particulate organic matter (CPOM) (> 1 mm) and medium particulate organic matter (MPOM) (250 μm - 1 mm), which were ashed to determine organic matter content (dry weight).

Conductivity (surrogate measure of stream nutrient concentrations), temperature, dissolved oxygen, width (four measurements), depth (five measurements in midstream), and water velocity (five measurements) for each site were measured in the field. Water samples were taken back to the laboratory for pH and suspended solids analysis. Stability of each site was assessed using the Pfankuch Stability index (Pfankuch, 1975). Substrate composition (relative proportions of silt, sand (< 0.2 cm diameter), gravel (0.2-6.0 cm), small cobbles (6.1-12.0 cm), large cobbles (12.1-26.0 cm), boulders (> 26 cm) and bedrock), the percentage of stream channel covered by vegetation and type of stream bank (riparian) vegetation (native forest, exotic woodland, scrub, crop/pasture, other) were assessed visually. All of these environmental measurements may be important in influencing invertebrate communities and may also change with harvesting.

6.3.2 Objective 2: Monitoring immediate effects of harvesting and road construction

The survey in objective one was followed by a monitoring programme using a subset of the survey sites (Appendix 6.1). Twelve streams were sampled once every two months from February to December 1997, in or around three exotic forests - Lismore, Tawarau and Te Wera (Fig. 6.1). Ten streams were sampled to assess whether seasonal influences i.e., decreased light and water temperatures, and increased rainfall, had any effect on macroinvertebrate community responses to harvesting. This aids in the determination of ideal times for monitoring. Within each forest, a mature exotic forest site (greater than 10 years old), a harvested site (harvested within the last year), a native forest site and a pasture site (except at Lismore for the latter two) were sampled.

To assess short term harvesting effects, one stream in Te Wera (harvested between March and November 1997) and one in Tawarau (road construction from July to December 1997) were also sampled every two months before and after disturbance (Plate 6.1).

PLATE 6.1

Plate 6.1. Pre and post harvest sites sampled once every two months from February to December 1997. The Te Wera stream (top) was clear-cut between March and November, and a road constructed beside the Tawarau stream from July to December (bottom).

Sampling methods for the monitoring programme were the same as for the survey (objective one) except that three macroinvertebrate samples were collected instead of four.

6.3.3 Objective 3: Experimental investigation of sedimentation effects on invertebrates

The experiment was conducted in five streams in or near Pirongia forest, in November and December 1997, to assess the effects of deposited sediment on macroinvertebrate communities independently of other environmental variables influenced by harvesting. Two streams drained pasture catchments and three drained mature exotic forest catchments.

Within each stream there were four treatments (each of 1 m^2 area) of no sedimentation (control and flow control), light and heavy sedimentation, each comprising three replicates, placed consecutively in a downstream direction (control, flow control, light and heavy sedimentation) (Plate 6.2). Replicates were placed at 10 m intervals down stream. The flow control, light and heavy sediment areas all had flow regulating nets of nylon mesh (2 mm mesh size), upstream and downstream, that allowed for water and invertebrate movement but slowed water velocity to limit sediment re-suspension.

Five litres of sand was added to the 1 m^2 area of the light sediment treatment and 20 litres to the heavy sediment treatment, the latter covering the substrate to a depth of 5 cm. Sand was alluvial sediment collected from the Pirongia area and sieved to < 4 mm. One terra cotta tile (area 32 mm^2) was placed in each treatment to determine the effects of sedimentation on algae. The experiment was run for 21 days (22 November to 13 December 1997) with sediment being added every three to four days to maintain consistent levels within treatments.

Velocity, depth, width, conductivity and dissolved oxygen were measured at each stream on day 1, 14 and 21. Water samples were also collected for pH and suspended solids analysis. On day 21, one benthic macroinvertebrate sample was taken from the

PLATE 6.2

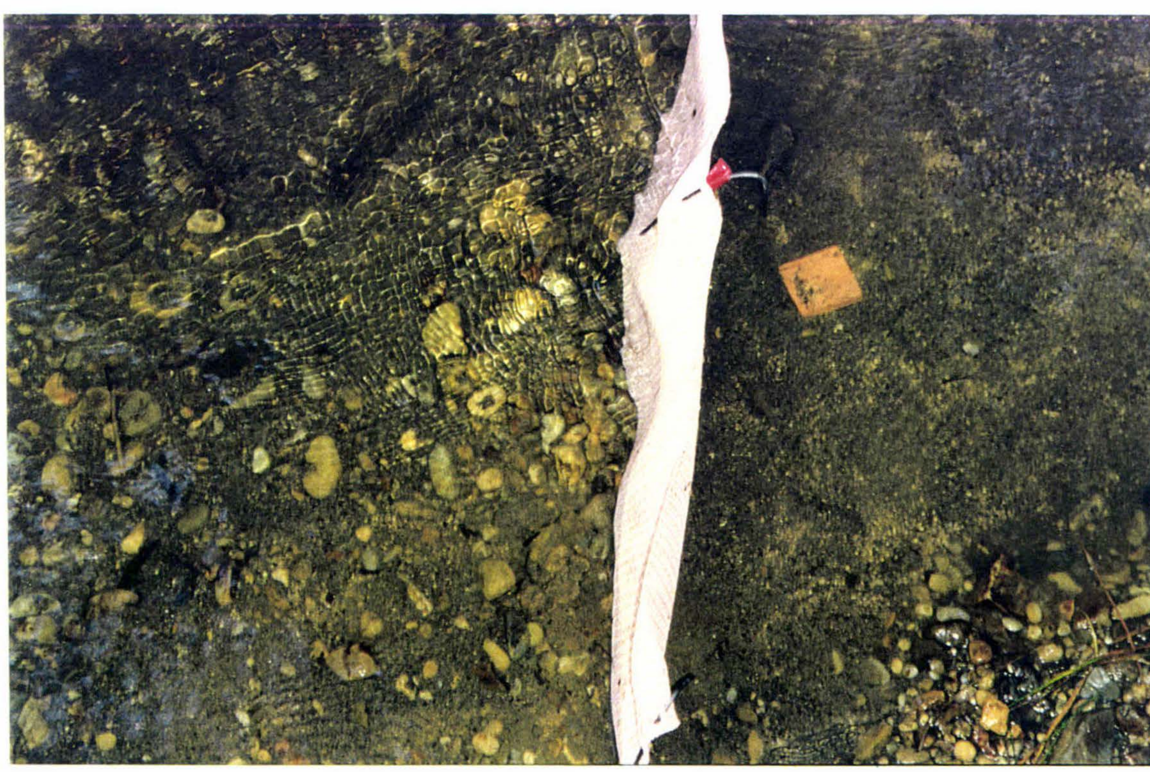


Plate 6.2. Experimental design of a replica from a forest site, showing the four treatments (top); left to right: control, flow control, light and heavy sedimentation; and a comparison of light and heavy sedimentation treatments (bottom).

centre of each treatment (to reduce edge effects) using a Surber sampler (area 0.1 m^2, 250 μm mesh) and the collected macroinvertebrates counted and identified. Algae tiles were removed and biomass measured as per survey (objective one).

6.3.4 Diversity indices

To examine differences in diversity between regions, land use types and season, three diversity indices were calculated for each site using the macroinvertebrate community data. Margelef's index (Clifford and Stephenson, 1975) is a measure of species richness (i.e., the number of species) and a high index value indicates high diversity. The Berger-Parker dominance index (Berger and Parker, 1970) and the Simpson's index (Simpson, 1949) are both measures of evenness (i.e., whether one or two species dominate the community: a significant impact). For both these indices, a high index value indicates that 1 or 2 species dominate the community, signifying low diversity.

6.3.5 Biotic indices

Biotic indices are frequently used to determine water quality using macroinvertebrates as indicators. The Macroinvertebrate Community Index (MCI), Quantitative Macroinvertebrate Community Index (QMCI) and the Percentage of Ephemeroptera (mayflies), Plecoptera (stoneflies) and Tricoptera (caddisflies) (% EPT) are the currently established standards in New Zealand and were calculated to determine whether there were differences in water quality between land use types. The MCI (Stark, 1985) is an index based on the presence of macroinvertebrate species which have previously been allocated a score (1 to 10) based on their pollution tolerances (1 = highly tolerant, 10 = highly sensitive). Streams with MCI scores greater than 120 may be regarded as having clean water and streams with scores less than 80 as severely polluted. The QMCI (Stark, 1993) is a quantitative variant of the MCI. The % EPT (Lenat, 1988) uses the abundance of three orders of insects which characteristically consist of species that prefer pristine conditions, hence a high index score indicates more pristine conditions.

6.3.6 Statistical analyses

Statistical differences between treatments (e.g., region and land use type for the survey; land use type, season and stream for the monitoring; and land use type and sediment and control treatments for the experiment) for environmental variables (i.e., width, depth, conductivity etc.) and invertebrate community characteristics (i.e., diversity and biotic indices), were analysed using Analysis of Variance (ANOVA) with SAS (SAS, 1989).

Site macroinvertebrate community data were also analysed using multivariate analysis. Multivariate analysis is a useful tool which takes complex biological data and presents it in a simpler form, helping to identify trends in communities. Multivariate analysis included ordinations carried out with Detrended Correspondence Analysis (DECORANA) using the PC-ORD statistical package (McCune and Mefford, 1995). This method places communities along a gradient determined by community composition, and also allows the correlation of environmental variables with these gradients. This helps to identify important factors influencing communities. The monitoring data were also analysed using another multivariate technique, classification (Cluster Analysis). This method of analysis places site communities into discrete groups according to their similarity in community composition.

6.4 SUMMARY OF RESULTS

6.4.1 Environmental Variables

Harvested streams tended to have lower water velocity, less vegetation cover over the stream, higher water temperatures, increased sediment deposition and decreased stream bank stability, compared to mature exotic forest streams (Table 6.1). Streams in mature forest were more similar in physical and chemical characteristics to streams in native forest, and harvested streams were more similar to pastures streams, except that pasture streams had higher water velocity.

Water chemistry (i.e., pH, dissolved oxygen) was less affected by land use type. However, conductivity increased in the first six months after harvesting but returned to previous levels within one year. Organic matter was not significantly different between mature and harvested streams but the larger woody debris, which was not measured, was noticeably higher in recently harvested streams.

Regional differences in geology dictated the extent of erosion at disturbed sites and hence the percentage of silt in stream beds differed between regions i.e., Pirongia and Tawarau tended to be less prone to erosion and therefore, were less prone to increased sedimentation in streams following harvesting.

6.4.2 Algae

Algal biomass on stones was lowest in native streams but in other land use types was similar, although mature streams had a slightly higher mean biomass than other groups. Algal biomass changed seasonally, being highest in summer and autumn. This may have obscured harvesting effects during winter when algal growth was limited.

Table 6.1. Environmental characteristics (mean (with range in parentheses)) for land use types within each region, for 42 streams in Lismore, Pirongia, Tawarau and Te Wera forests, sampled in January and February 1997. Cond = conductivity, DO = dissolved oxygen, Temp = temperature and SS = suspended solids.

	No. of Streams	Width (m)	Depth (cm)	Velocity (m/s)	Cond (μS/cm)	DO (mg/l)	Temp (°C)	Stability	pH	SS (mg/l)
Lismore										
Mature	2	1.80 (1.40-2.22)	3.46 (1.14-5.78)	0.29 (0.28-0.31)	578 (491-664)	10.4 (9.3-11.4)	15.1 (11.7-18.4)	118 (105-130)	7.15 (7.10-7.20)	0.37 (0.09-0.64)
Harvested	3	1.08 (0.50-1.95)	11.86 (9.20-16.38)	0.20 (0.09-0.35)	530 (359-661)	6.9 (2.9-9.1)	16.2 (15.2-16.8)	102 (83-133)	7.38 (7.09-7.83)	0.14 (0.07-0.25)
Pasture	3	1.56 (0.51-2.18)	17.57 (5.32-31.80)	0.47 (0.18-0.75)	398 (322-479)	9.5 (9.2-9.5)	16.6 (15.8-17.4)	98 (76-120)	7.51 (7.20-7.78)	0.08 (0.07-0.09)
Native	3	0.95 (0.34-1.76)	12.04 (5.10-20.44)	0.25 (0.20-0.37)	263 (247-272)	10.4 (10.1-11.1)	14.0 (13.2-14.5)	70 (64-82)	7.78 (7.62-7.94)	0.13 (0.02-0.30)
Tawarau										
Mature	5	1.05 (0.02-1.62)	20.82 (6.16-52.72)	0.31 (0.25-0.36)	139 (101-191)	9.9 (9.4-10.6)	12.0 (11.4-12.5)	92 (62-114)	7.07 (6.84-7.23)	0.06 (0.03-0.09)
Harvested	1	1.21	15.46	0.18	107	9.3	13.0	141	7.21	0.02
Pasture	2	1.60 (1.46-1.74)	33.65 (32.20-35.10)	0.32 (0.32-0.32)	106 (90-122)	9.2 (8.7-9.6)	14.6 (14.3-14.8)	84 (77-91)	6.74 (6.66-6.81)	0.07 (0.04-0.10)
Native	2	1.54 (1.52-1.57)	17.29 (16.10-18.40)	0.35 (0.33-0.37)	158 (156-159)	9.9 (9.4-10.4)	12.3	100 (84-115)	7.22 (7.19-7.24)	0.05 (0.04-0.05)
Pirongia										
Mature	3	1.41 (1.29-1.61)	13.09 (10.40-16.62)	0.53 (0.45-0.62)	92 (68-128)	9.5 (9.2-9.7)	13.6 (12.4-15.0)	68 (63-74)	7.00 (6.90-7.10)	0.03 (0.02-0.05)
Pasture	1	2.83	14.92	0.65	401	9.7	19.9	95	7.21	0.02
Native	1	0.90	6.62	0.20	188	10.1	12.3	72	7.21	0.06
Te Wera										
Mature	5	0.95 (0.35-1.92)	17.02 (6.90-33.60)	0.40 (0.17-0.65)	112 (93-147)	7.9 (8.1-9.8)	14.4 (13.3-15.2)	93 (76-123)	7.01 (6.82-7.24)	0.21 (0.03-0.39)
Harvested	5	1.29 (0.35-2.02)	17.57 (3.05-29.34)	0.21 (0.16-0.29)	111 (66-134)	7.0 (5.0-9.7)	16.2 (14.2-17.7)	103 (87-114)	7.36 (6.70-8.72)	0.61 (0.007-2.83)
Pasture	3	1.12 (0.83-1.53)	31.33 (18.64-44.46)	0.57 (0.27-0.99)	92 (88-97)	8.1 (7.5-8.7)	16.2 (15.5-17.2)	108 (101-118)	6.77 (6.61-7.02)	0.03 (0.01-0.05)
Native	3	0.65 (0.44-0.94)	15.22 (7.96-24.90)	0.30 (0.13-0.58)	72 (53-88)	8.6 (7.3-10.3)	13.5 (13.2-13.7)	73 (61-85)	7.02 (6.85-7.22)	0.10 (0.04-0.17)

6.4.3 Macroinvertebrates

Macroinvertebrates showed a marked response to harvesting. Harvesting was associated with greater numbers of animals (Fig. 6.2), and changes in community composition i.e., harvesting was associated with greater numbers of pollution tolerant species (e.g., amphipods and ostracods). There were no differences in diversity (i.e., number of species) between mature and harvested streams, however, pasture streams had a greater number of species than other land use types.

Mature streams had greater numbers of mayflies, stoneflies and beetles, and harvested streams had greater numbers of amphipods, ostracods, worms and fly larvae. The DECORANA graded sites on axis one from those in harvested and pasture catchments to those in mature exotic and native forest catchments, although there was no distinct division between these groups (i.e., the circles overlap) (Fig. 6.3). This overlap may have been influenced by forest age i.e., as pine trees re-establish, communities gradually revert to pre harvest condition. The DECORANA indicated that invertebrate communities in mature exotic and native forest streams were most similar and communities in harvested and pasture streams were most similar.

Communities were not clearly separated into regions either (Fig. 6.3), suggesting regional effects did not have a dominating influence on the macroinvertebrate communities. However, regional influences may have had more effect on the response of communities to different land use types i.e., Lismore and Te Wera stream communities showed distinct differences between land use types but Tawarau and Pirongia stream communities were similar in all streams, with Pirongia showing a weaker separation between land use types on axis two (Fig. 6.3). Regional influences were also important for other aspects of invertebrate communities e.g., number of species and number of animals, suggesting that the extent to which a community responds to harvesting and other land use types, depends on which region it is in.

Several environmental variables correlated with the DECORANA axes (Fig. 6.3) indicating that communities in pasture and harvested streams were associated with high

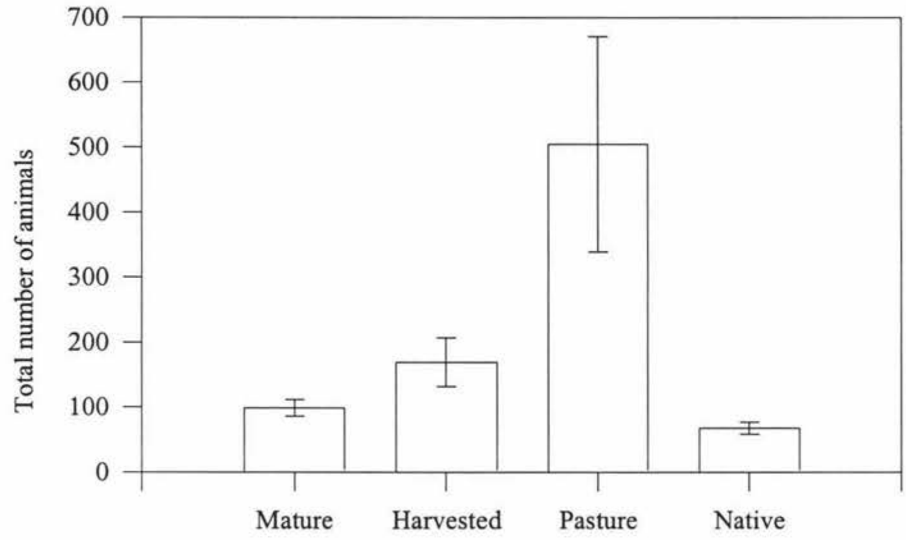


Figure 6.2. Mean total number of animals (± 1 SE) for each land use type, for macroinvertebrates collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, sampled during January and February 1997.

amounts of silt, less riparian cover, higher water temperatures and decreased stability compared to native and mature exotic forest streams. Most Lismore and Te Wera sites also occurred lower on axis one associating streams in these forests with higher amounts of silt and lower stability (independent of land use).

The biotic indices (MCI, QMCI, % EPT) all indicated that native and mature exotic forest streams had species characteristic of good water quality, whereas pasture and harvested streams had more pollution tolerant species (e.g., snails, oligochaetes and amphipods) (Fig. 6.4). However, both pasture and harvested sites were still only moderately polluted (MCI scores were greater than 80). There were no significant differences in index scores between pasture and harvested sites or native and mature forest sites, which suggests a similar situation to the DECORANA, that there are two general community types, one for open streams and one for shaded streams.

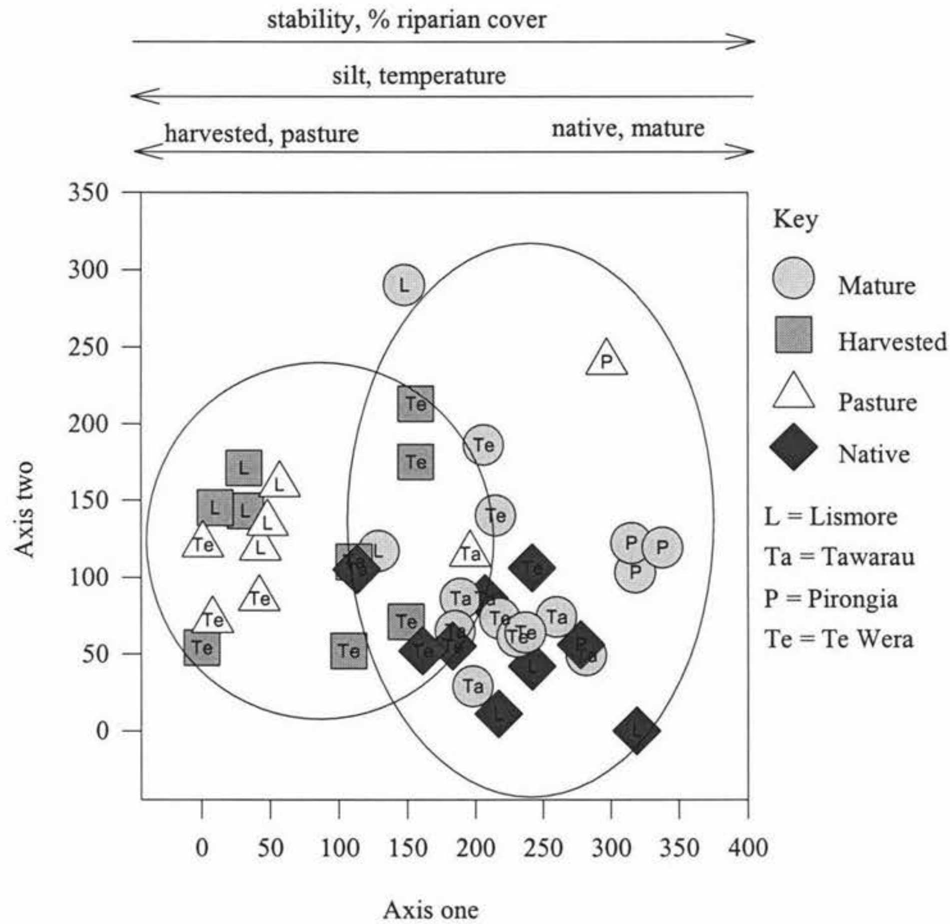


Figure 6.3. Axis one of a Detrended Correspondence Analysis as a function of axis two for macroinvertebrate communities collected from 42 streams in Lismore, Pirongia, Tawarau and Te Wera forests, sampled during January and February 1997.

The two streams sampled before and after harvesting showed differences in invertebrate responses. The Te Wera stream community showed characteristics of harvested stream communities in general i.e., increases in the number of animals (Fig. 6.5) and changes in community composition to greater numbers of pollution tolerant species. However, the Tawarau stream community was scarcely affected by the construction of a road beside it, with only the number of animals increasing after construction began. Community composition remained similar throughout the monitoring period.

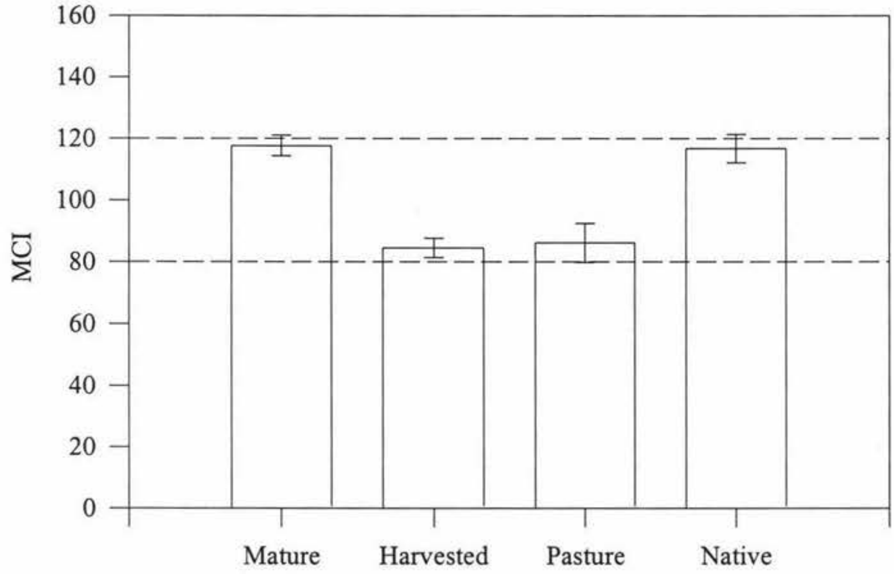


Figure 6.4. Mean Macroinvertebrate Community Index (MCI) scores (± 1 SE) for each land use type, for macroinvertebrate communities collected from 42 streams in Lismore, Tawarau, Pirongia and Te Wera forests, during January and February 1997. Scores greater than 120 indicate pristine conditions; scores between 80 and 120 indicate moderate pollution and scores less than 80 severe pollution.

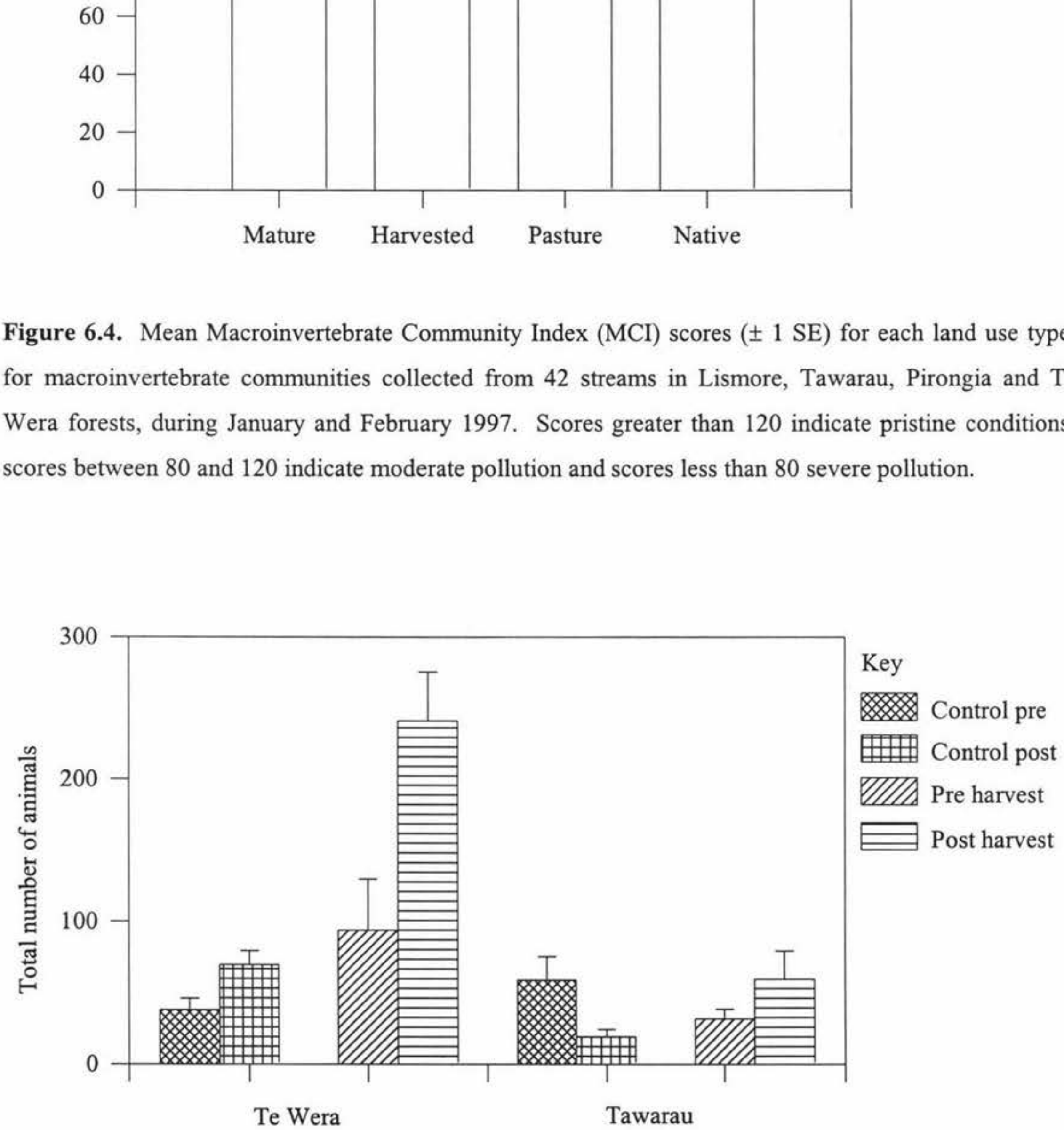


Figure 6.5. Mean total number of animals (± 1 SE) for pre and post harvest streams and respective controls, for macroinvertebrates collected from four streams in Te Wera and Tawarau forests, sampled every two months from February to December 1997.

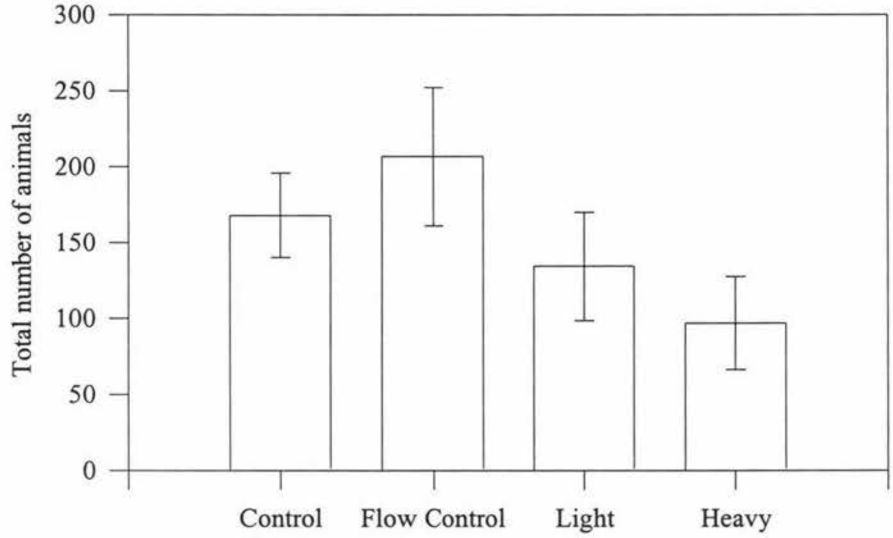


Figure 6.6. Mean total number of animals (± 1 SE) for samples collected from experiment after 21 days in streams with heavy, light or no sediment addition (with flow control and without) between 22 November and 13 December 1997.

The results of the experiment, assessing the effects of sedimentation on macroinvertebrates, suggest that heavy sedimentation by itself was not the primary determinant of macroinvertebrate community structure in forest harvesting. The number of individuals decreased substantially with heavy sedimentation and to a lesser extent with light sedimentation (Fig. 6.6), but diversity and overall community composition changed little between sediment and control treatments. QMCI and % EPT scores indicated a decrease in the number of pollution sensitive species (mayflies and stoneflies), but not an increase in the numbers of pollution tolerant species as often happens after harvesting. The length of the experiment (21 days) may not have been long enough to bring about a greater change in community composition.

6.5 DISCUSSION AND RECOMMENDATIONS

6.5.1 Synthesis from Results

The results indicate that forest harvesting does adversely affect stream macroinvertebrate communities, and therefore, water quality. The characteristic effects were increases in the number of animals and changes in community composition from species characteristic of pristine streams (e.g., mayflies and stoneflies) to species characteristic of low water quality (e.g., snails, amphipods and ostracods). Diversity of invertebrate communities was not affected by forest harvesting but was higher in pasture streams compared to other land use types. Although harvesting had no effect on diversity, this does not indicate that the community was unaffected by harvesting. The community composition has changed even if the number of species remained similar. In addition, the aim when conserving a community, is not necessarily to increase diversity of a community but to minimise disturbance to the existing one.

Environmental variables influenced by harvesting included the amount of deposited sediment on the stream bed, stability of the stream banks, light and water temperature regimes, and the percentage of stream covered by vegetation. These factors can therefore be related to trends in invertebrate communities i.e., pasture and harvested stream communities are associated with increased silt on the stream bed, increased water temperatures, and decreased stability and percentage of stream covered by vegetation. It must be noted however, that these associations between environmental variables and invertebrate communities are not absolute. They are only links (through correlations). As this is only a one-off study that included only one season, few definitive conclusions can be established.

The effect of forest harvesting on invertebrate communities differed between regions. This highlights the importance of regional factors, such as climate and geology, in determining levels of impact. Such large scale factors have been identified as important influences on both the physical stream environment and the invertebrate communities within it (Biggs and Gerbeaux, 1993; Friberg et al., 1997). Lismore and Te Wera forests have been identified as more likely to be influenced by effects of forest harvesting due to the high erodibility of their soils. In Pirongia and Tawarau, it seems effects of harvesting on communities is less.

The similarity of stream invertebrate communities in mature exotic forest to stream communities in native forest indicates that reforestation allows the re-establishment of communities comparable to those in native forest streams. However, this reversion does not take place for at least six years after harvesting (i.e., communities in catchments harvested within six years of sampling were similar in community composition). During this time invertebrate communities in harvested streams are similar to those in pasture streams and indicates that water quality is low in these streams. Therefore, harvesting can cause negative environmental effects for at least six years. All mature exotic forest streams had similar communities and these forests were greater than 10 years old. Therefore, the time frame for complete recovery to pre harvest community composition is between six and 10 years.

Invertebrate communities sampled immediately after harvesting in Te Wera were different from those before harvesting commenced and remained that way for greater than one year after harvesting. However, the construction of a road beside the Tawarau stream had little impact on the stream community. This could be attributed to several factors i.e., a small strip of vegetation was left relatively undisturbed between the stream and road construction. Furthermore, the length of stream exposed to impact was only 50 m on one side and this may have influenced the extent of community responses (Murphy and Hall, 1981; Davies and Nelson, 1994; Tait et al., 1994). Geology may have also been an important factor i.e., the stream was mainly bedrock and therefore less prone to erosion. This again emphasises the importance of large scale regional influences.

Although sedimentation was identified as a potentially important influence on invertebrate communities in the survey and monitoring, the experiment indicated that sedimentation did not influence the community in a similar way to harvesting i.e., sedimentation resulted in a decrease in animal numbers, mainly those sensitive to pollution, whereas forest harvesting also resulted in an increase in the numbers of pollution tolerant species above that of numbers before harvesting. The removal of stream bank vegetation may also be an important effect of harvesting (Quinn et al., 1994; Collier, 1995; Quinn et al., 1997), and subsequent effects of stream bank vegetation removal - increased light and water temperature - tend to increase the growth of algae, which is an important food source for many stream invertebrates. The invertebrate community may have initially decreased in numbers due to sedimentation (as the experiment indicated). However, the stream may then be recolonised with species that are determined by type and abundance of food available (i.e., algae or terrestrial organic matter) which is influenced by riparian vegetation removal. Although algae did not increase significantly after harvesting in this study, the greater number of animals at harvested sites may have kept algae levels down (Steinman, 1992; Botts, 1993; Allan, 1995).

Therefore, the physical effects of harvesting most likely to influence invertebrate communities, that should be prevented or minimised during harvest operations are sedimentation and stream bank vegetation removal. Care should be taken especially in regions identified as most susceptible to disturbance (i.e., higher erosion potential) e.g., Lismore and Te Wera.

Under the Resource Management Act 1991 it is required to promote the sustainable management of natural and physical resources and in particular, "to avoid, remedy or mitigate any adverse effects of activities on the environment". The survey and monitoring indicate that forest harvesting does adversely affect the stream environment and therefore, we are obliged to avoid, remedy or mitigate such adverse effects.

6.5.2 Development of recommendations

Having established the potentially most important adverse effects of forest harvesting on stream communities i.e., sedimentation and removal of stream bank vegetation, recommendations, drawn from the present results and other literature, have been made to reduce these effects.

6.5.2.1 Sedimentation

Most sediment in streams draining exotic forest catchments is found to be derived from forestry roads and skid tracks that cross or pass near streams (Mosley, 1980; Roberts, 1994; Waters, 1995). In the survey and monitoring, many of the streams sampled ran parallel to roads, and therefore, roads are likely to be a major source of sediment to these streams. The best way to protect streams from sedimentation is to remove the source (Collier et al., 1995; Waters, 1995). Therefore, effects can be minimised by reducing the number of stream crossings and roads running parallel to streams, and locating these roads on ridges where feasible (Waters, 1995). In Tawarau forest, there was little effect of road construction on stream communities primarily because of the short area of stream subjected to disturbance, as well as stream bank vegetation being left undisturbed. Therefore, if care is taken during construction, any effects should be short term and invertebrate communities will then make a rapid recovery (Ryder, 1989). Better construction of roads e.g., culverts and road surfacing, will also reduce the extent of adverse effects over time.

However, the deposition of sediment is controlled by stream water velocity i.e., low water velocity encourages greater sediment deposition (Ryder, 1989). In the survey, the water velocity of most of the harvested streams were significantly slower than those of streams in mature exotic forest. Low velocity coupled with the fact that most sites sampled were beside forestry roads, maximises sediment deposition.

Decreases in velocity was associated with the amount of slash deposited in streams during harvesting. This was prevalent in several of the streams sampled with some streams being totally covered (e.g., TWF6, LF2). The woody debris acted as a dam, slowing velocity and thereby facilitating major deposition of sediment (Murphy and Hall, 1981; Quinn et al., 1997). In this case, effects may be more long term, as even during periods of high flow, flushing of sediment will be minimal.

Therefore, to prevent long term sedimentation and high levels of short term sedimentation, care should be taken to minimise addition of woody debris to streams.

This may involve increased harvesting costs due to more careful harvesting techniques or having to harvest from either side of the stream. Road construction near streams should be minimised where possible.

6.5.2.2 Light and water temperature changes

One common solution to prevent major changes in the light and water temperature regimes is to leave a riparian buffer strip on either side of the stream. A riparian buffer strip is defined as a vegetated zone located between a stream or river and adjacent areas subject to human alteration (Castelle et al., 1994) and is an area where little alteration occurs. Not only do riparian buffers offer continuous shading when harvesting is conducted, but they also filter nutrients and sediment from runoff originating from harvested, road and landing areas. Riparian buffers can therefore reduce sedimentation, light, water temperature and nutrient increases in streams, as well as reduce the input of large amounts of woody debris, as harvesting activity would be minimal in these zones. As harvesting is the most damaging process in the forestry industry, the role of a riparian buffer would be most important during this phase and the initial replanting (Gilliam et al., 1992).

There is little debate about how effective riparian buffers can be, but there is controversy over how wide a riparian buffer should be in order to protect the stream and its communities (Castelle et al., 1994). Most literature on riparian buffer widths has shown that anything less than 5-10 m will provide little protection under most conditions (Castelle et al., 1994). The desired width of a buffer zone depends on several aspects e.g., intensity of adjacent land use, soil type, slope steepness, riparian vegetation and specific buffer functions required. For temperature moderation, buffers of 15-30 m are adequate and for sediment control 10-60 m is sufficient, while 30 m seems to be the minimum width required to maintain the biological components of a stream (Castelle et al., 1994).

The size of the stream must also be taken into consideration. Most streams sampled in this study were very small (< 2 m wide) and although most were permanently flowing, it may not be feasible to place a 30 m buffer zone on either side of a stream that small. However, smaller streams will require buffers to moderate temperatures more than larger streams and rivers which are naturally more open (Gilliam et al., 1992).

Costs of riparian management and loss of production are also important issues. It is not practical to leave mature pine trees near streams as, apart from loss of production, these may cause greater problems than no riparian margin i.e., decreasing bank stability (Collier et al., 1995). Dense planting of *Pinus* species has also been demonstrated to prevent growth of ground cover species, due to the acidity generated by fallen pine needles (Taranaki Regional Council, 1992), and mature trees are susceptible to wind throw (Gilliam et al., 1992). Therefore, existing pine trees should be removed from the riparian area and a riparian buffer established at the replanting stage to protect streams at the next rotation.

With careful harvesting around streams minimal damage may occur to existing vegetation, reducing the effort required to establish a healthy riparian zone. Native species are relatively slow growing and if a riparian zone is needed to function efficiently in a short period of time, exotic species may be better (Collier et al., 1995). In forestry however, a riparian zone can be allowed longer to establish as it is 30 years until the next rotation.

6.5.3 Monitoring

The process of gathering information and monitoring forest operations is a statutory requirement of Regional Councils (Marden, 1994). It is also important to conduct independent impact assessments that will facilitate consent procedures as well as assist in planning in forest management. At Mangatu Forest in 1991, three factors were identified as being most important for compliance with the Resource Management Act 1991 when monitoring during harvesting (Marden, 1994):

1. Early identification of environmental values and specific hazards.
2. Monitoring of harvesting impacts throughout the term of harvesting.

3. Maintaining open lines of communication with the public and in particular, with local authorities.

In the monitoring conducted in this study several important factors were apparent:

1. Forest harvesting effects on macroinvertebrate communities differed in the different regions e.g., in Lismore and Te Wera, community composition was significantly different between harvested and mature streams but in Tawarau, changes were much less marked. This helps to identify areas that will be more susceptible to harvesting effects.
2. Potential effects of forest harvesting on macroinvertebrate communities were identified i.e., sedimentation and removal of stream bank vegetation.
3. Changes in water quality were detected through monitoring of macroinvertebrate communities over time i.e., more pollution tolerant species were present in harvested and pasture streams, indicating that these land uses were associated with higher productivity (i.e., greater algal biomass and numbers of animals) and sedimentation.

This study demonstrates that succession of stream communities back to pre harvest structure (similar to communities in native streams) does occur. However, exactly when such a reversion happens needs to be determined. This study was unable to quantify the exact time that effects due to harvesting were mitigated (i.e., some time between six and 10 years). Therefore, more long term monitoring is required at specific sites to assess more accurately when the system changes back to pre harvest conditions.

6.6 SUMMARY OF RECOMMENDATIONS

1. Where possible, locate roads away from streams and keep stream crossings to a minimum. When constructing roads and landing areas near streams, keep disturbance to a minimum and re-seed areas of bare soil on completion. Well constructed roads with good drainage systems incorporated and culverts or bridges included, will minimise long term effects, decrease maintenance costs and reduce the amount of road construction required at the end of the next rotation.

2. Excessive slash input into streams should be minimised by careful harvesting and riparian management. This will prevent long term sedimentation and allow streams to flush out short term excess sediment.

3. Identify areas that will be more likely to have a high risk of environmental effects. Regional differences have been highlighted in this study so all regions should be assessed individually.

4. Where feasible, plant riparian buffer strips to protect streams. A width of 10 m may be adequate if of good quality and implemented in conjunction with other recommendations. If areas are of high risk i.e., high erosion potential or steeper stream banks, a wider buffer strip may be required. If a buffer strip is not feasible, care should still be taken during harvesting to leave existing riparian vegetation as undisturbed as possible. Healthy riparian buffers should have abundant undergrowth and a few larger trees. *Pinus* species should not be left in riparian zone and self-seeded *Pinus* species should be removed.

5. Continue monitoring before, during and after harvesting, incorporating a longer term monitoring strategy to determine when effects of harvesting are mitigated.

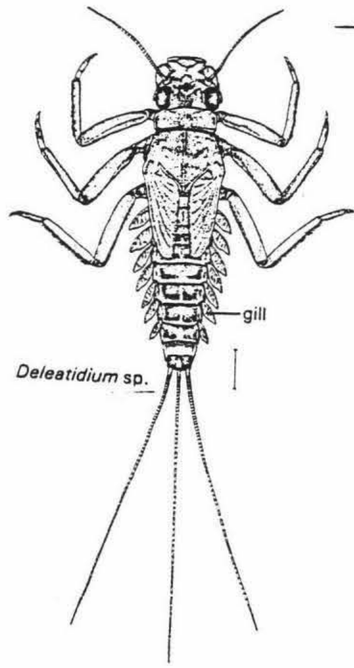
6.7 CONCLUSIONS

In broad terms, the forestry industry has had only a relatively small part to play in the overall decline in water quality standards in New Zealand compared to other land uses e.g., agriculture (O'Loughlin, 1995). Furthermore, effects of forest harvesting on water quality may be short lived if impact is minimised, as throughout most of the rotation water quality is protected. However, adverse effects during harvesting are significant, and if such effects are still prevalent six years after harvesting they can not be referred to as short term. Furthermore, as some regions are likely to be more susceptible to environmental disturbance than others, such effects may be even more long term. Therefore, action must be taken to minimise these effects.

Of the recommendations, several are currently being practised. Most harvesting has been done in only small areas of the catchment, and roads are well constructed, although many follow and frequently cross streams. To attain the aim of minimising effects of forest harvesting, effort must be made to use a combination of the recommendations so that sources of water quality degradation are eliminated or reduced (e.g., roading locations), and protection measures function at maximum efficiency. Also, in areas of higher risk such as Lismore and Te Wera forests, more care should be taken during the harvesting period.

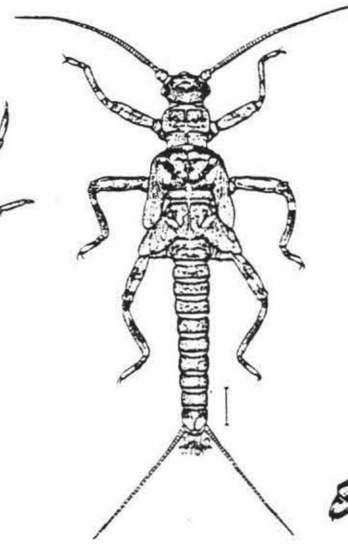
6.8 GLOSSARY OF COMMON MACROINVERTEBRATES

Ephemeroptera (mayfly)



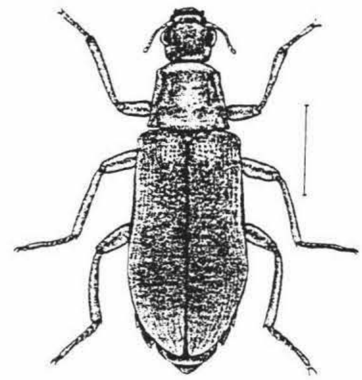
Deleatidium sp.

Plecoptera (stonefly)



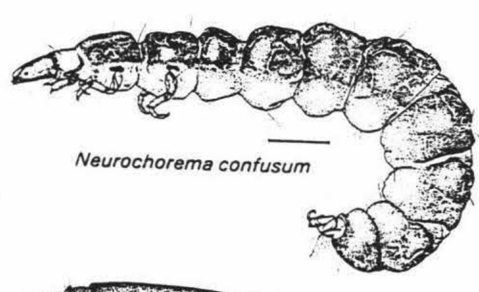
Acroperla trivacuata

Coleoptera (beetle)



Elmidae

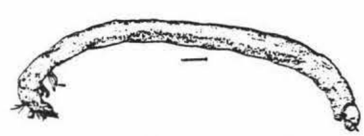
Tricoptera (caddisfly)



Neurochorema confusum

Olinga feredayi

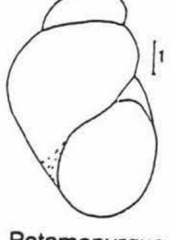
Diptera (fly)



Chironomus zealandicus

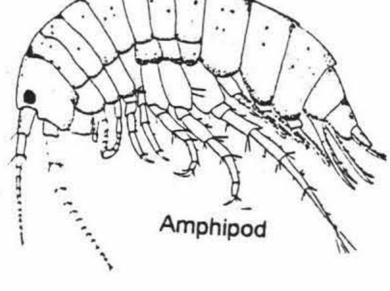
Paralimnophila skusei

Mollusca (snail)



Potamopyrgus

Crustacea



Amphipod

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6.10 APPENDIX

Appendix 6.1. Site name abbreviations and details for 42 streams in Lismore, Pirongia, Tawarau and Te Wera forests, sampled in January and February 1997. * = sites sampled every two months from February to December 1997.

Site Abbreviation	Site Name	Land use type	NZMS 260 Grid Reference
LF1 *	Lismore exotic 1	Mature	S22: 975524
LF2 *	Lismore exotic 2	Harvested	S22: 967511
LF3	Lismore exotic 3	Harvested	S22: 979494
LF5	Lismore exotic 5	Mature	S22: 049505
LF6	Lismore exotic 6	Harvested	S22: 994485
LP1	Lismore pasture 1	Pasture	S22: 980486
LP2	Lismore pasture 2	Pasture	S22: 053504
LP3	Lismore pasture 3	Pasture	S22: 954471
LN1	Lismore native 1	Native	S21: 948661
LN2	Lismore native 2	Native	S21: 944654
LN3	Lismore native 3	Native	S21: 104602
TaF1 *	Tawarau exotic 1	Harvested	R16: 825225
TaF2	Tawarau exotic 2	Mature	R16: 809227
TaF5 *	Tawarau exotic 5	Mature	R16: 789228
TaF6	Tawarau exotic 6	Mature	R16: 773205
TaF7	Tawarau exotic 7	Mature	R16: 788228
TaF11 *	Tawarau exotic 11	Mature/road constructed during monitoring	R16: 796231
TaP1	Tawarau pasture 1	Pasture	R16: 825222
TaP2 *	Tawarau pasture 2	Pasture	R16: 822226
TaN1 *	Tawarau native 1	Native	R16: 776237
TaN2	Tawarau native 2	Native	R16: 778236
PF8	Pirongia exotic 8	Mature	S15: 931412
PF9	Pirongia exotic 9	Mature	Q15, R15:898417
PF10	Pirongia exotic 10	Mature	Q15, R15:901415
PP3	Pirongia pasture 3	Pasture	S15: 913423
PN3	Pirongia native 3	Native	S15: 921431
TWF1	Te Wera exotic 1	Harvested	R19: 502207
TWF2	Te Wera exotic 2	Harvested	Q19: 495211
TWF3	Te Wera exotic 3	Mature	Q20: 498178
TWF4	Te Wera exotic 4	Harvested	Q20: 486178
TWF5 *	Te Wera exotic 5	Harvested	R19: 522231
TWF6 *	Te Wera exotic 6	Mature	R19: 524215
TWF7	Te Wera exotic 7	Mature	R19: 521201
TWF8	Te Wera exotic 8	Mature	R19: 509174
TWF10	Te Wera exotic 10	Harvested	R20: 514182
TWF11 *	Te Wera exotic 11	Mature/harvested during monitoring	Q20: 497183
TWP1	Te Wera pasture 1	Pasture	Q20: 494203
TWP2 *	Te Wera pasture 2	Pasture	Q20: 491199
TWP3	Te Wera pasture 3	Pasture	Q20: 483188
TWN1	Te Wera native 1	Native	R19: 520224
TWN2	Te Wera native 2	Native	R20: 515177
TWN3 *	Te Wera native 3	Native	Q20: 498183

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