THE SEASONAL AND SPATIAL DYNAMICS IN THE PHYTOMACROFAUNAL COMMUNITIES OF LAKE HENLEY, MASTERTON.

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By

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ABSTRACT

Spatial and seasonal dynamics of macrophyte beds in Lake Henley, a shallow artificial lake in Masterton, were examined between June 1993 and June 1994. Observed fluctuations in macrophyte biomass were linked with changes in filamentous algae associated with the weed beds. Invertebrate communities associated with submerged macrophytes were also examined at multiple sites between June 1993 and June 1994. Overall community composition was related more to seasonal influences than differences between sites within the lake. However, species richness and abundance did differ spatially. Recommendations for the ongoing management of Lake Henley, including management of the macrophyte beds and the maintenance of water quality and quantity, are made with respect to the ecological characteristics of the lake. The influence of trophic status on macrophyte invertebrate communities was also explored with a survey conducted in May 1994 of 13 other lakes in the North Island. Nutrient enriched lakes were characterised by phytomacrofaunal communities with high abundance and higher numerical dominance, whereas nutrient poor lake phytomacrofaunal communities were characterised by lower abundances but higher diversity of some taxonomic groups, particularly insects.

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INTRODUCTION

Lake Henley was built to provide the Masterton community with a recreational water body. Within a few years of completion of the lake excessive growth of introduced aquatic weeds began to place limitations on many recreational activities that the lake had been intended for, particularly boating and fishing. Concern regarding the most suitable method of weed control, together with a recognition of the need to understand the overall lake ecology in order to establish appropriate management strategies, led to the commissioning of this project.

Introduced aquatic weeds have been invading New Zealand water ways for the past century (Howard-Williams *et al* 1987) and the warm shallow waters and silty substrate of Lake Henley offer ideal conditions for the establishment of tall growing submerged plants such as *Elodea canadensis* (Michx) and *Potamogeton crispus*. These submerged aquatic plants contribute to the overall ecological functioning of such a shallow water body, but nevertheless are classified as nuisance due to their tall growth form (Johnstone 1986). Weed management in New Zealand relies on integrating mechanical, chemical and biological forms of weed control (Clayton & Wells 1989; Hughes 1976). Macrophytes are an integral part of the aquatic environment, and therefore this adds to the importance of careful assessment of the effects on the environment of macrophyte removal or control (Johnstone 1986).

To determine the most environmentally sound management approach for Lake Henley it was necessary to examine the spatial and seasonal dynamics of the macrophyte beds in relation to the physicochemical and biological characteristics of the lake. Chapter one describes the changes that occurred in the macrophyte beds between June 1993 and June 1994. It is suggested that the macrophyte beds undergo biomass fluctuations in response to seasonal changes in the filamentous algae associated with them, which are in turn influenced by the changes in nutrient concentration in the lake.

The use of invertebrate communities for biomonitoring is on the increase in New Zealand, particularly the use of invertebrate communities that inhabit stream benthos (Stark 1994). The invertebrate communities associated with the submerged aquatic plants in Lake Henley (phytomacrofaunal communities) were considered potentially useful for similar monitoring of changes in lake water quality. In Chapter Two the seasonal and spatial dynamics of the invertebrate communities associated with the submerged macrophytes in Lake Henley are examined in this context. Comparisons are also made between *Elodea canadensis* and *Potamogeton crispus* in the lake and a neighbouring stream to examine the influence of macrophyte species on these communities.

The paucity of quantitative data on New Zealand phytomacrofaunal communities, however, and a lack of basic understanding of their response to environmental changes makes it difficult to use them in biomonitoring. In an attempt to address this a survey of the composition of invertebrate communities associated with the macrophytes in the shallow littoral zone of 14 North Island lakes was conducted. Chapter Three explores the relationship between the community of invertebrates associated with submerged aquatic plants, and the nutrient enrichment status of the lake in which they occurred.

In the final chapter of this thesis the findings on the ecology of Lake Henley are collated within a management plan designed to maximise the lake's natural biotic characteristics whilst reducing those detrimental to recreational uses of the lake. The guide-lines for lake management include managing the macrophytes through the introduction of herbivorous grass carp, *Ctenopharyngodon idella*, and monitoring water quality through assessment of filamentous algae and invertebrates, and by maintaining sufficient lake water levels in the lake.

Chapter1: Physico-chemical and biological determinants of macrophyte dynamics in Lake Henley, Masterton

ABSTRACT Seasonal and spatial dynamics of dense macrophyte beds of *Elodea canadensis* in Lake Henley, a shallow artificial lake, were examined in relation to physico-chemical and biological characteristics of the lake between June 1993 and June 1994. Macrophyte biomass was influenced by differences in substrate characteristics between sites within the lake, and by seasonal growths of thick mats of filamentous algae. Despite lake water having high nutrient concentrations this did not yield high phytoplankton levels possibly because of the rapid flushing of water through the lake, and/or the filtering of nutrients from the water by the filamentous algae.

Keywords: artificial lake; aquatic weed ; *Elodea canadensis*; filamentous algae; nutrients.

INTRODUCTION

Lake Henley is a small, artificial lake situated on the edge of Masterton in the Southern Wairarapa. It was built 8 years ago by excavation of a quarry site and the establishment of a channel to divert water into the lake from the nearby Ruamahunga river. Despite the lake's young age there are extensive beds of *Elodea canadensis* which interfere with recreational uses of the lake such as angling and boating. It is likely that fragments of *Elodea* colonised the lake by drifting in from the Ruamahunga river, while one of the two inflowing streams is the most likely source for the other less common weed species, *Potamogeton crispus*, that occurs in seasonally abundant clumps throughout the lake.

The invasion of tall growing aquatic weeds into New Zealand water ways has resulted in much research into different methods of control (reviewed in Clayton & Wells 1989; Howard-Williams *et al* 1987; Hughes

1976). Exotic aquatic plants have achieved weed status primarily because of their tendency to form dense monospecific stands of surface reaching vegetation. Macrophytes are however an integral part of the aquatic environment (Johnstone 1986), playing both a beneficial as well as a nuisance role. The method of weed control employed thus impacts on the entire system.

The management focus for Lake Henley is a programme of weed control through the introduction of a moderate stocking rate of Grass Carp (*Ctenopharyngodon idella*) (Buchanan 1991; Miller 1994). Other Grass Carp trials in New Zealand have aimed to achieve complete elimination of all submerged vegetation (Clayton *et al* 1992; Mitchell *et al* 1984; Rowe & Schipper 1985; Tanner *et al* 1990). The introduction of Grass Carp into Lake Henley for the purpose of managing weed beds at a desirable level therefore constitutes an experiment in aquatic weed management. This is a long term experiment that requires an understanding of the physical and chemical conditions in the lake prior to the introduction of the fish, and an appreciation of the seasonal and spatial dynamics of the macrophyte beds in relation to these physical and chemical characteristics. The aim of this study therefore is to describe the physical, chemical and biological factors of the lake in relation to their likely impact on macrophyte growth.

STUDY SITE

Lake Henley was constructed as an artificial recreational lake and consequently has a shallow uniform depth (mean depth 2 m). The lake is situated at 114 m.a.s.l, and has a surface area of 11.14 ha, a shoreline of 1.6 km and a volume of 223 000 m^3. It has a rapid flow through of water diverted from the nearby Ruamahunga river which results in a short residence time estimated to range between 3 and 5 days. The bulk of the water enters the lake via the diversion channel from the river (through the main inflow), but two small streams, Hiona stream and Te Ore Ore stream (Fig. 1.1), also contribute to the total inflow into the lake. The main outflow from the lake returns much of the water back to the Ruamahunga river, and two smaller outflows maintain water levels in a series of artificial wetlands adjoining the lake.

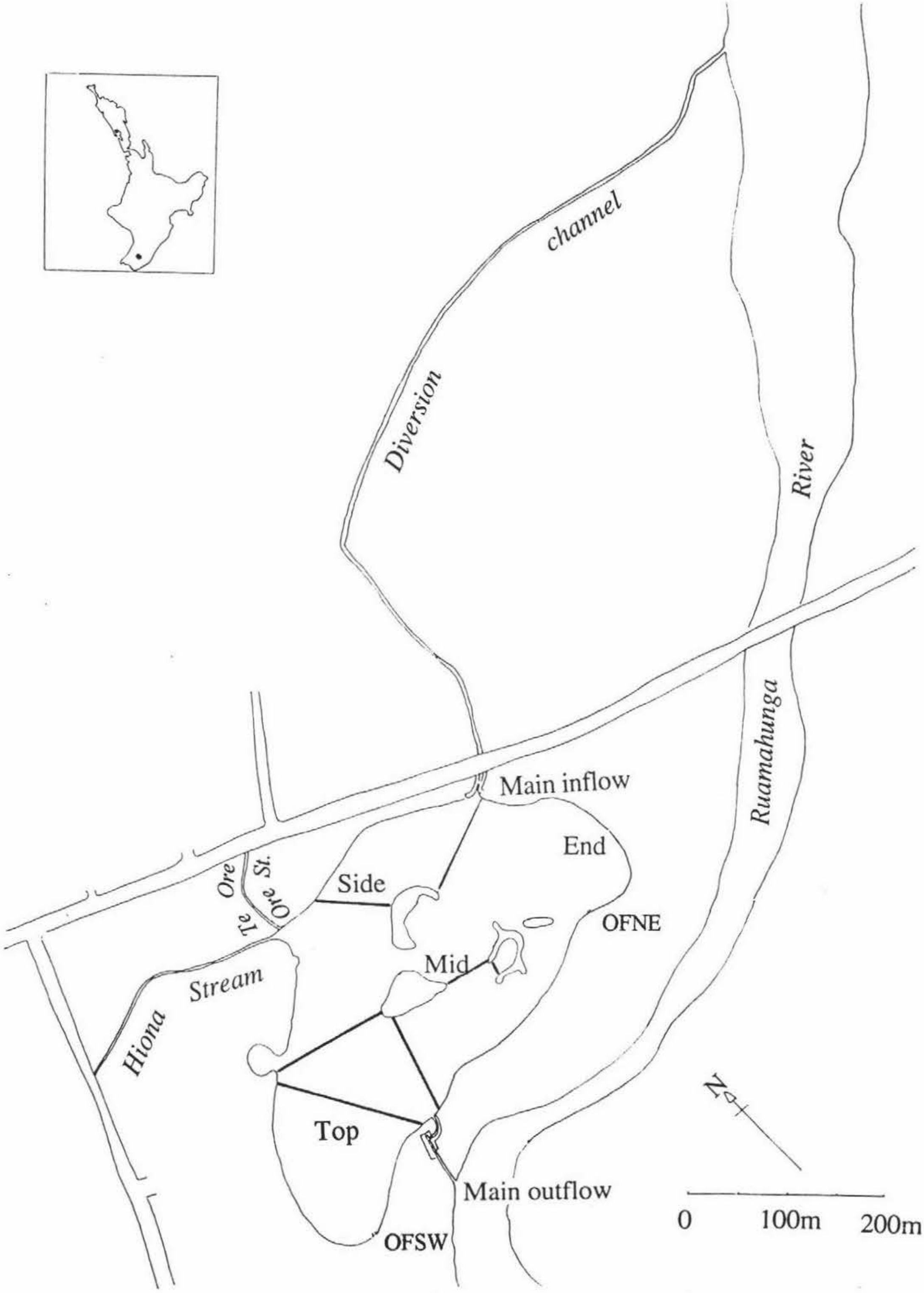


Fig. 1.1. Location map of Lake Henley, and the Ruamahunga river showing sampling sites and transect lines (———).

METHODS

Hydrology

Discharge measurements were made at monthly intervals between May 1993 and August 1994 at the three inflows to the lake: the main inflow, Hiona stream, and Te Ore Ore stream, and at the three outflows: the main outflow, the outflow to the Northeast wetlands (OFNE), and the outflow to the Southwest wetlands (OFSW) (Fig.1.1). Discharge rates were determined by measuring depth and current velocity with a velocity head rod at 50 cm intervals across the main inflow and outflow, and at 10 cm intervals across Hiona stream. Discharge from the Te Ore Ore stream was determined by measuring the time for a fixed volume of water to accumulate in a graduated bucket as it flowed out of the culvert. The cross sectional surface area and velocity head rod measurements taken at the centre of each outflow pipe were used to estimate discharge rates from the outflows into the wetlands (Brater & King 1976).

Chemical characteristics

Chemical characteristics were measured at monthly intervals between May 1993 and August 1994 at the four lake sites (Top, Mid, End and Side), the three inflows and the three ouflows (Fig.1.1). Dissolved oxygen was measured with a YSI model 58 dissolved oxygen meter, conductivity with an Orion model 122 conductivity meter, pH with a Solstat pH/MV meter, and temperature with an Orion model 122 meter. Temperature range was also recorded with maximum/minimum thermometers at Hiona stream (near to its entrance to the lake), and at the End and Mid lake sites, however vandalism prevented continuous readings.

Water samples were collected monthly between May 1993 and August 1994 from the same study sites in acid washed polythene bottles. They were kept on ice and filtered through Watman GFC filters (1.2 μm) in the laboratory within 48 hours of collection, and frozen prior to chemical analysis. Concentration of nitrate nitrogen (NO_3-N) (the nitrogen form most easily available to plants) was determined for samples collected between June 1993 and October 1993 by the colourmetric autoanalyses method (Techicon), and dissolved inorganic phosphorus was determined on filtered (0.45 μm) samples by the method of Murphy and Riley (1962). The remainder of the samples (November 1993 to August 1994) had dissolved reactive phosphorus (DRP) determined using molybdenum blue and ascorbic acid reduction followed by an automated colorimetric finish (Downs 1978a), and nitrate nitrogen (NO_3-N) was determined by an automated hydrazine reduction, diazotization with sulphanilamide, and an automated colorimetric finish. (Downs 1978b).

Phytoplankton

Chlorophyll-*a* concentration of lake water samples collected monthly from the four lake sites between May 1993 and August 1994 was used as an indication of the phytoplankton density (Pridmore & Hewitt 1984). This was determined by filtration of water samples through Watman GFC filters (1.2 μm pore size), followed by acetone extraction of the crushed filters. Spectrophotometer measurements at 630 μm, 645 μm and 750 μm wavelengths were made of the acetone after centrifuging, and again following acidification (Pridmore *et al* 1983). Chlorophyll-*a* concentrations were below the detection limits of the equipment available for the analysis of water samples collected between May 1993 and February 1994. Water samples from March 1994 to August 1994 were analysed with the use of more sensitive equipment (G. Paine *pers comm*) and chlorophyll-*a* concentration was determined using calculations in Pridmore *et al* (1983) with corrections made for phaeopigment concentrations.

Macrophyte distribution

Macrophyte distribution was surveyed between June and August 1993. Seven transects were established at representative sections of the lake (Fig.1.1) and grab samples taken at 2 m intervals by hooking up all plant material present at each point to determine macrophyte composition (Clayton 1983).

Macrophytes were sampled monthly between June 1993 and June 1994 to assess seasonal and spatial changes in biomass and composition from three sites within the lake (Mid lake, End lake, and Side lake) (Fig.

1.1), and from the Hiona stream between June 1993 to March 1994, until macrophytes were cleared from the stream by council workers. Sites differed slightly in their substrate characteristic with the Side site having a greater depth of silt than the other two sites, possibly due to the close proximity of the inflowing Hiona stream which during storm periods was observed to be heavily laden with silt. Three replicate macrophyte samples were collected from each site using a 0.02 m^2 cylinder-sampler (300 μm mesh, 0.15 m diameter). The lake samples were all collected from a depth of 1.5 m, where the height of the weed beds ranged from 0.7 - 1.2 m. Plant material was weighed after removal of the associated invertebrates (Chapter Two) and dried for 4-5 days at 80 °C to constant weight.

Presence or absence of filamentous algae associated with the macrophyte beds was noted on each sampling occasion. In November 1993 filamentous algae associated with the weed samples was manually separated from the macrophyte material and weighed after drying both at 80 °C for 4-5 days to constant weight.

Data analysis

A two level mixed model analysis of variance (ANOVA) on $\log_{10}(x+1)$ transformed data was conducted using SAS (1988) to assess whether macrophyte biomass, or physiochemical characteristics differed between sites and seasons. Site was treated as a fixed criteria, and month as a random one, with SITE x MONTH as the appropriate error term for testing site effects (Sokal & Rohlf 1981). Spearman rank (r_s) correlations were performed using the CORR procedure of SAS (1988) to examine links between the biological and physical variables measured.

RESULTS

Hydrology

On average 77 %, (range 30 % to 80 %), of the total volume of water entering Lake Henley via the three inflows left again via the three outflows (Fig. 1.2). However, on two sampling occasions more water was recorded exiting the lake than entering it (July 1993 and May 1994). These sampling dates co-incided with days when the intake gate was closed to reduce silt laden water from entering the lake. Unaccounted for losses and gains of water from the lake are most likely attributed to water loss through ground water seepage and evaporation, and gains from rainfall.

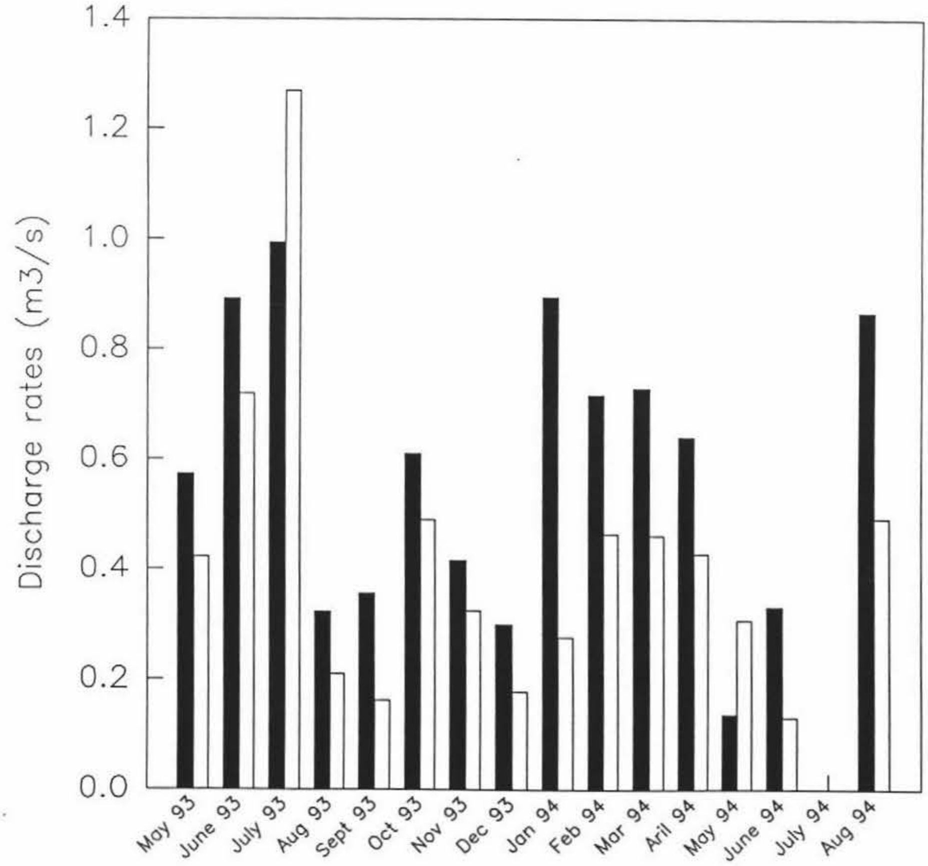


Fig. 1.2. Total discharge volumes into ( ) and out of ( ) Lake Henley recorded between May 1993 and August 1994.

Chemical characteristics

Not surprisingly, spot temperature measurements and temperature range recorded at study sites increased in the summer months, and decreased during winter (Fig. 1.3). However, water temperature in the lake and in the inflowing streams were significantly different ($F_{6,84} = 7.54$ $P < 0.001$) with water flowing into the lake from the main inflow and Hiona stream consistently lower in temperature than any of the lake sites (Table 1). Dissolved oxygen measurements also reflect seasonal changes in water temperature (Fig. 1.4) but there were no differences between the inflows and lake ($F_{6,79} = 1.54$ $P = 0.18$). There were no seasonal trends in conductivity or pH, although there were differences in conductivity measurements at the different sites ($F_{6,54} = 5.00$ $P < 0.001$) with conductivity highest in Te Ore Ore stream and the main inflow (Table 1).

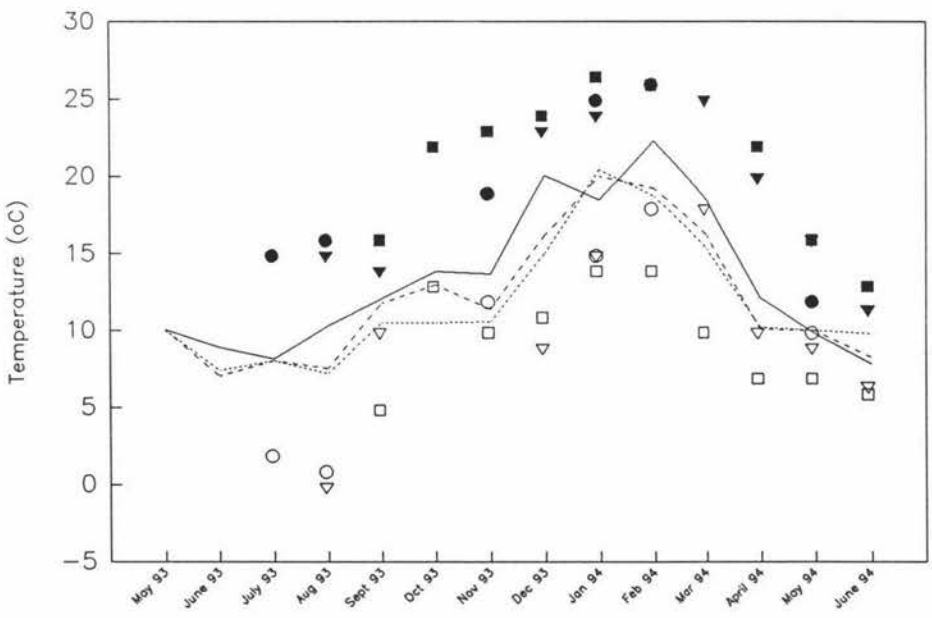


Fig. 1.3. Mean temperature of Lake Henley (——), Hiona stream (-----), and the main inflow (········) recorded between May 1993 and June 1994.
Maximum and minimum temperatures are also given where recordings were made.
(○) Mid min (●) Mid max (▽) End min (▼) End max
(□) Hiona stream min (■) Hiona stream max.

Table 1. Mean values for physicochemical and biological variables measured at ten sampling sites in and around Lake Henley between May 1993 and August 1994.

	Hiona	Te Ore Ore	Inflow	Outflow	OF SW	OF NE	Top	Mid	End	Side	F	P
Temperature (^oC)	12.2 (7-19.2)	13.5 (6.8-19.8)	11.9 (7.2-18.7)	-	-	-	13.5 (7.5-22.2)	13.4 (7.7-22)	13.7 (8-22.2)	13.4 (7.6-22.6)	$F_{6,84}$=7.54	P<0.001
Dissolved Oxygen (mg/m^3)	10.9 (8.6-14.2)	7.9 (6.4-12.4)	11.7 (9.8-12.7)	-	-	-	9.8 (15.2-5.0)	10.4 (15.0-6.4)	11.1 (15.0-8.0)	10.81 (15.4-7.6)	$F_{6,79}$=1.54	P=0.18
Conductivity (ucm/s)	114.2 (74-178)	160.3 (186-140)	118.8 (95-166)	-	-	-	121.9 (97-160)	107.6 (94-153)	118.5 (95-142)	117.4 (94-144)	$F_{6,54}$=5.00	P<0.001
DRP (mg/m^3)	192.6 (10-1606)	61.4 (7-226)	29.4 (5-148)	27.8 (5-135)	45.6 (10-181)	69.6 (3-512)	38.3 (8-192)	26.4 (4-152)	24.3 (7-148)	29.6 (9-175)	$F_{9,116}$=6.77	P<0.001
NO_3-N (mg/m^3)	673 (22-2750)	2600 (2100-3900)	374 (93-750)	161 (7-575)	135 (1-495)	115 (3-453)	236.81 (0-1750)	161.5 (0-625)	134.1 (0-575)	218.7 (0-1150)	$F_{9,116}$=20.70	P<0.001
Chlorophyll-*a* (mg/m^3)	-	-	-	-	-	-	1.5	0.8	1.0	1.6	$F_{3,12}$=3.07	P=0.07
Macrophyte biomass (g/m^2)	203 (49-416)	-	-	-	-	-	-	433 (192-991)	520 (106-1052)	459 (98-1250)	$F_{3,33}$=10.91	P<0.001

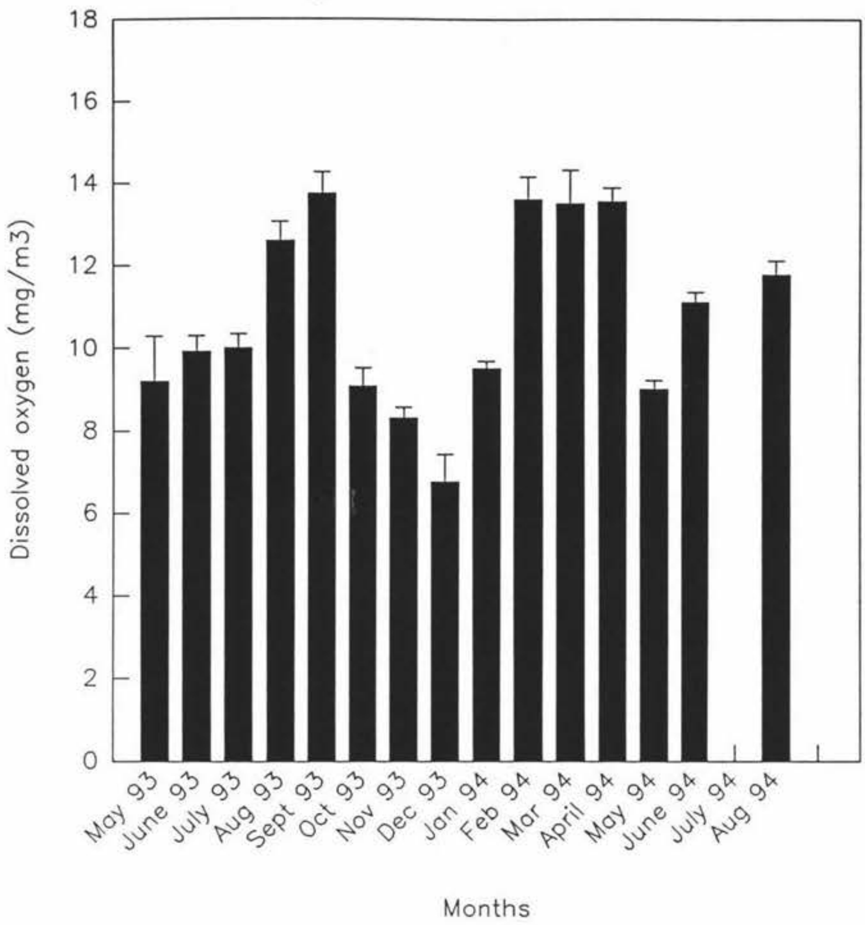


Fig. 1.4. Concentration of dissolved oxygen (mg/m^3) averaged over the four lake sites in Lake Henley between May 1993 and August 1994.

Mean concentrations of dissolved reactive phosphorus and nitrate nitrogen collected monthly from the lake are presented in Figure 1.5. Differences between the concentrations of DRP and NO_3-N from the sites within the lake were not significant ($F_{3,116} = 1.93$ $P = 0.14$, $F_{3,116} = 1.98$ $P = 0.14$ for DRP and NO_3-N lake concentrations, respectively), however, the concentration of DRP in Hiona stream was on average higher than at any other sampling location, and the concentration of nitrate nitrogen was consistently higher in Te Ore Ore stream (Table 1). Concentrations of both DRP and NO_3-N were lowest between November 1993 and May 1994 with peaks in DRP and NO_3-N in August 1994 and May 1993 respectively (Fig. 1.5). The concentration of plant available phosphorus (DRP) was negatively correlated with conductivity ($r_s = -0.54$ $P < 0.001$), and positively correlated with nitrate nitrogen ($r_s = 0.38$ $P = 0.005$). Nitrate nitrogen was only correlated with DRP concentration.

Nutrient loadings for Lake Henley were estimated as a product of the mean DRP concentrations and average discharge, although the monthly sampling strategy gave only approximate nutrient loading estimates, particularly given the short residence time of water in the lake. The inflows contributed an average of 15, 0.2 and 42 kg/month (for Hiona stream, Te Ore Ore stream and the main inflow respectively) of plant available phosphorus to the lake meaning an estimated 693 kg/year entered Lake Henley, while an estimated 663 kg/year was recorded leaving the lake. Using monthly averages could result in an inflated estimate of a yearly load given the large phosphorus loading in the outflows in July 1993 and October 1993 (Fig. 1.6A). The median phosphorus loading entering the lake was only 352 kg/year and leaving the lake was 110 kg/year which again also suggests that less phosphorus left the lake than entered it. The amount of plant available nitrogen entering the lake via the inflows was estimated to be 96, 9 and 514 kg/month (for Hiona stream, Te Ore Ore stream and the main inflow respectively). Thus a total of 6865 kg/year was estimated to enter the lake during the sampling period, with an estimated total of 3098 kg/year leaving the lake (Fig. 1.6B).

Phytoplankton

Chlorophyll-*a* concentrations in the lake were extremely low throughout the entire sampling period, and were not significantly different at the four sites within the lake (Fig.1.7). During January 1994 the lake experienced a brief bloom of *Pandorina* sp. (Volvocales) (Bold 1977).

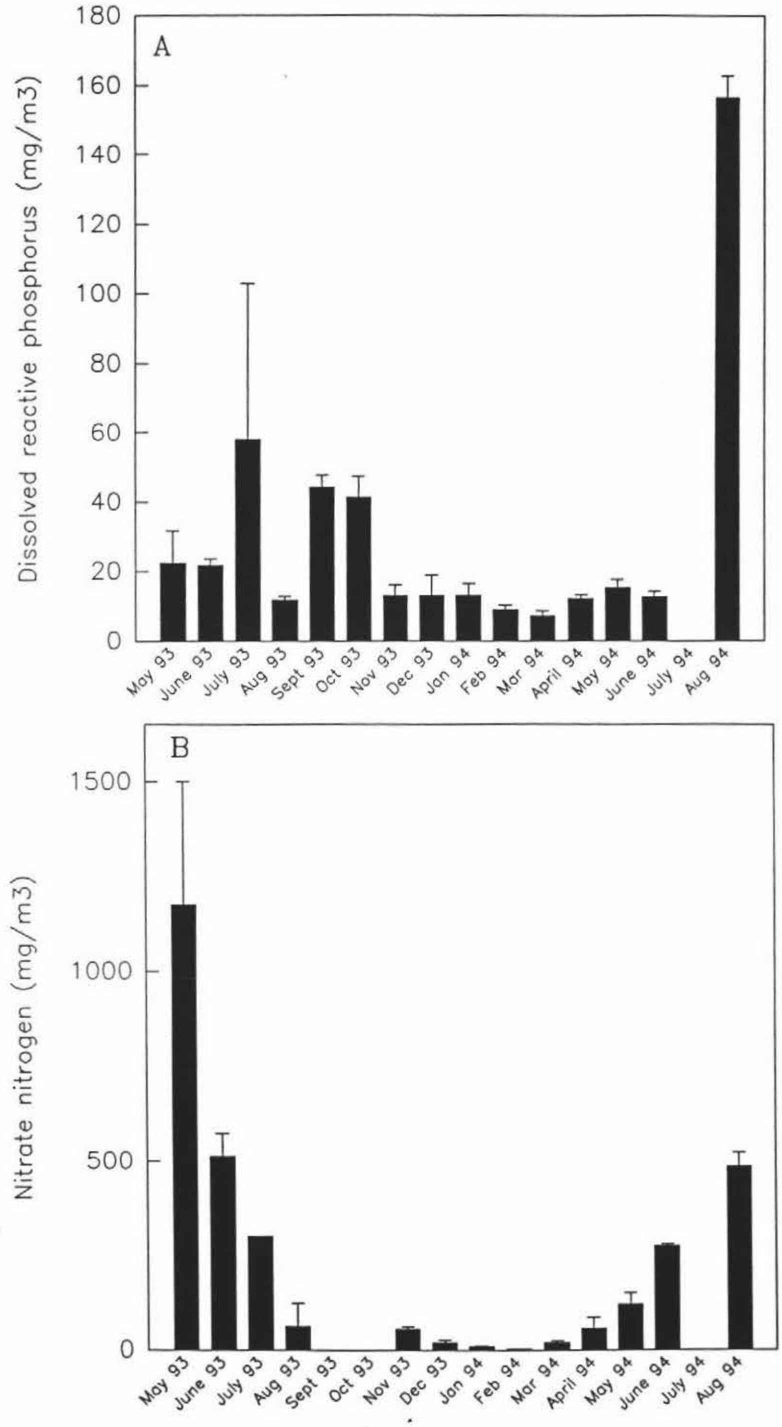


Fig. 1.5 Mean concentrations of **A**. Dissolved reactive phosphorus and **B**. Nitrate nitrogen (mg/m3) measured in Lake Henley between May 1993 and August 1994.

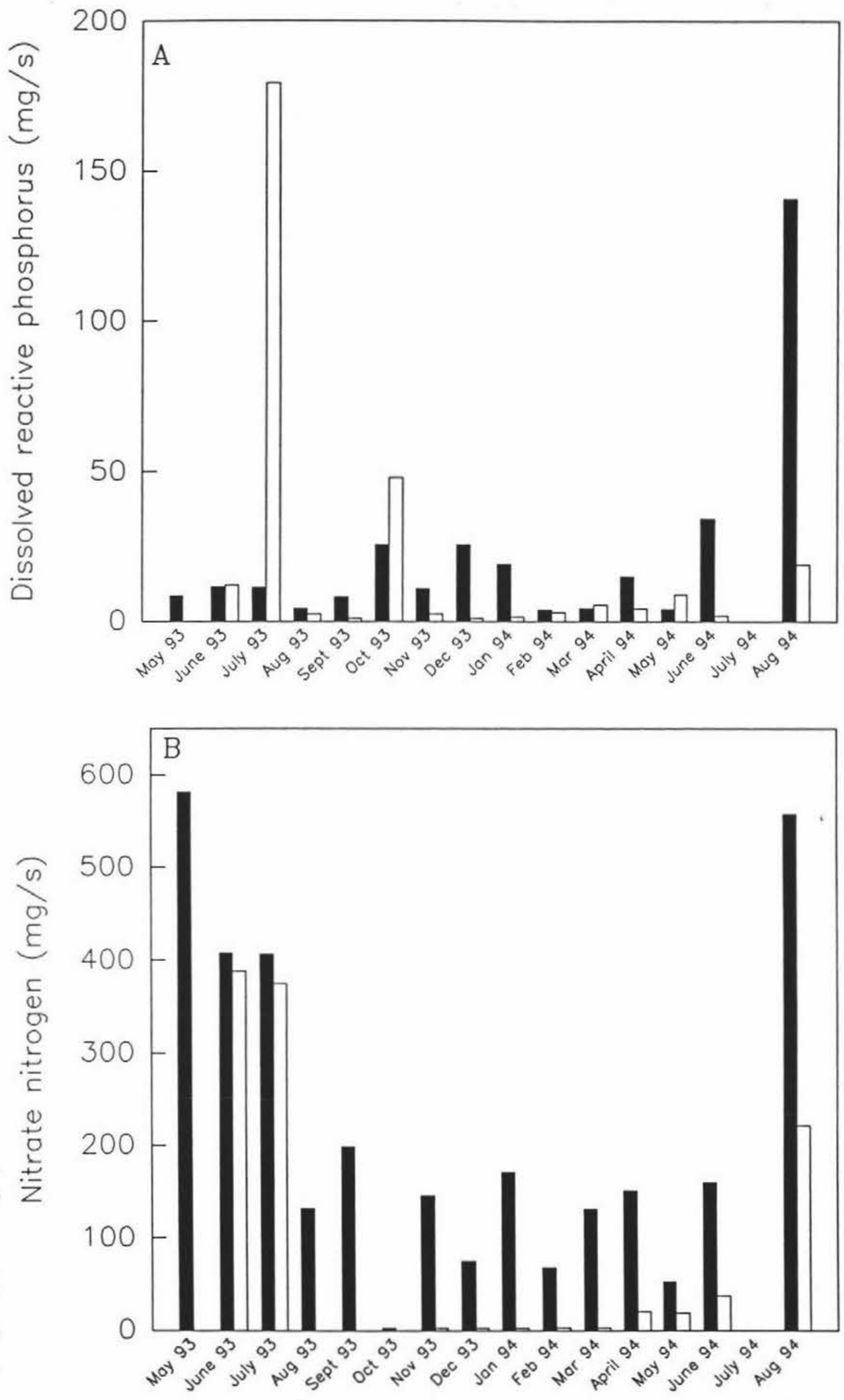


Fig. 1.6. Nutrient loadings for Lake Henley recorded between May 1993 and August 1994. **A.** Dissolved reactive phosphorus loadings recorded entering the lake from all three inflows (■) and exiting from all three outflows (□). **B.** Nitrate nitrogen (mg/m^3) loadings recorded entering the lake from all three inflows (■), and exiting the lake from all three outflows (□).

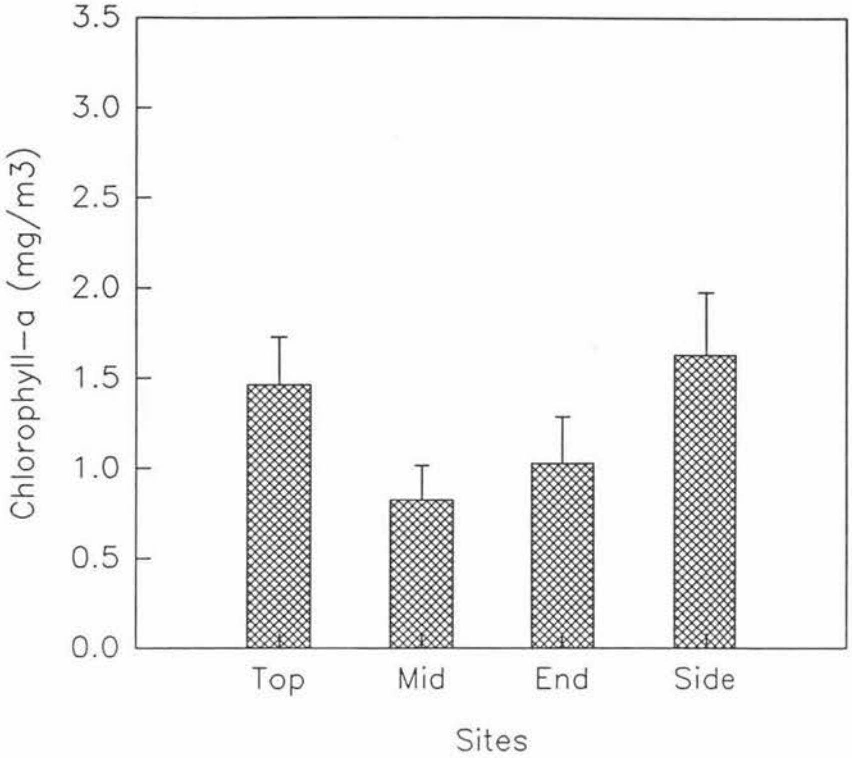


Fig. 1.7 Mean chlorophyll-a concentrations at the four sampling sites within Lake Henley from samples taken between March and August 1994.

Filamentous algae

Filamentous algal mats, composed predominantly *Spirogyra* sp. and *Oedogonium* sp. were observed over the macrophyte beds at the Side sampling site between June and December 1993, and throughout the lake between September and December 1993. In December 1993 the thick algal mats died and began to decay with an associated decline in dissolved oxygen levels, and a noticeable hydrogen sulphide odour. All filamentous algae had completely disappeared from the macrophytes by the end of January 1994. The algal mats were observed again in May and June 1994 associated with the submerged plants, although they were not as dense as previously. By August 1994 they had increased in density, and thick floating mats were again observed in the lake, this time near the main inflow. Filamentous algal biomass measured in November 1993 ranged from 100 to 500 g/m^2. The presence of filamentous algal mats did not lead to any change in pH although their presence was negatively correlated with dissolved oxygen levels (r_s = -0.33 P = 0.01).

Macrophyte distribution

Three macrophyte species (*Elodea canadensis*, *Potamogeton crispus*, and *Nitella* sp.) were present in Lake Henley for some or all of the sampling period. All of the points sampled in the distribution survey contained submerged vegetation. Of the 305 sample points, 88% contained only *E. canadensis*, 10% contained *E. canadensis* and *P. crispus* and 2% contained both *E. canadensis* and *Nitella* sp.. *E. canadensis* formed a dense bed across the entire lake bed, *P. crispus* formed scattered clumps on the wharf side of the islands, and isolated clumps of *Nitella* sp. were found towards the end of the lake nearest the outflow to the Northeast wetlands. Although the survey of the entire lake bed was only conducted once, observations during regular monthly sampling trips indicated relatively little change in this pattern. The only significant change noted was the establishment of beds of *P. crispus* at the Mid and Side lake sites in February 1994 and at the End lake site in April and May 1994. These *P. crispus* beds differed from the scattered clumps observed during the survey in that they were restricted to the shallow (50 cm depth) water along side the concrete edge of the lake.

The maximum biomass of *E. canadensis* was obtained from the Side lake site in August 1993 when the three replicates averaged 1178 g/m^2. There were marked differences in the macrophyte biomass between the four sampling sites (Table 1), although this difference depended on season ($F_{33,97}$ = 3.25, P < 0.001). Overall, the stream site had the lowest macrophyte biomass (average 203 g/m^2), with the End lake site having the greatest average (520 g/m2). The Side and Mid had averages of 459 and 433 g/m^2 respectively.

Changes in macrophyte biomass were not linked directly with obvious seasonal influences such as temperature (r_s = -0.07 P = 0.60), but seemed more closely linked to the presence of filamentous algae. Macrophytes under the thick algal mats in November and December were severely etiolated with a marked decline in biomass (Fig.1.8), increasing again between February to May 1994 when the algal mats had dissipated.

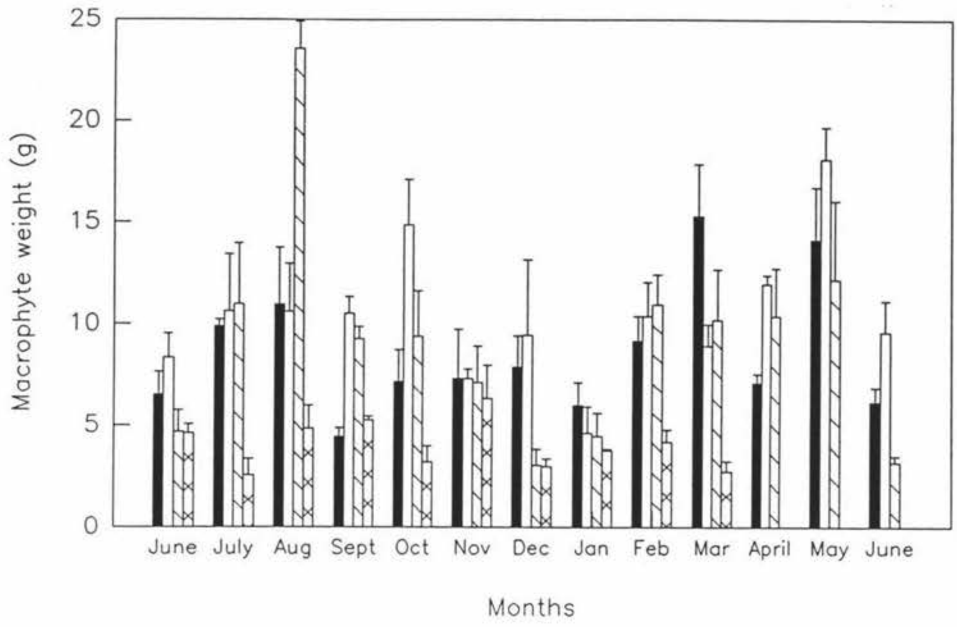


Fig. 1.8. Average dry weed weight from triplicate samples collected from Hiona Stream from June 1993 to March 1994, and from three sites within Lake Henley between June 1993 to June 1994.

( ) Mid Lake ( ) End Lake ( ) Side Lake

( ) Hiona Stream

DISCUSSION

Lake Henley is dominated by submerged exotic macrophytes, has a low residence time, high dissolved reactive phosphorus and nitrate nitrogen levels and low planktonic algal abundance. The range of macrophyte biomass recorded in Lake Henley (98-1250 g/m^2) is comparable to those recorded for *Elodea* in other New Zealand lakes. It is similar to that recorded for Lake Alexandrina (100-1100 g/m^2, Ward *et al* 1984), and for Lake Rotokakahi (1496 g/m^2, Brown 1975). However it is considerably higher than recorded in Lake Okataina (120g/m^2, Brown 1975) or range from the Nelson lakes (178-749 g/m^2, Brown 1975). The depth of silt associated with *E. canadensis* beds has been correlated with plant height in other studies (e.g., Brown 1975), therefore it is not surprising that the Side site, with the greatest silt depth supported the densest growth of macrophyte. Lake Henley had a lower mean biomass though than recorded from the 'very dense' stands in Lake Alexandrina (Ward *et al* 1987). This may relate to the shallow depth of lake (2 m) which would limit weed height, or to the marked influence of seasonally abundant filamentous algae.

The macrophyte stands have a large influence on the physico-chemical characteristics of the lake, acting as mechanical sieves for suspended material, increasing the magnitude of daily oxygen changes, consolidating the sediments and acting as the substrate for much of the lakes biotic component (Carpenter & Lodge 1986). Furthermore, the dense growths of submerged plants, and their associated epiphilic algae, are likely to restrict phytoplankton growth which might otherwise have occurred in such nutrient enriched waters (Pokrovskaya 1984). During periods of active growth, macrophytes draw phosphorus mainly from the sediments, and attached filamentous algae draw nutrients from the water. However, during periods of senescence of the macrophyte beds brought about by the overgrowth of filamentous algal mats, and during senescence of the algal mats themselves, dissolved phosphorus is released back into the water (Carpenter & Lodge 1986). The effect of any pulsed release of nutrients on the growth of microscopic green algae is likely to be lessened by the rapid flushing rate of water through the lake (Pridmore 1987). Also associated with these periods of senescence, oxygen levels decline as macrophytes and algae decay. Interestingly, the low phytoplankton abundance in Lake Henley, despite high nutrient levels makes it difficult to establish the trophic status of the lake - the high nutrient concentrations suggest a eutrophic status while the low phytoplankton levels suggest an oligotrophic status (Pokrovskaya 1984; Pridmore 1987).

Macrophytes in New Zealand waters do not exhibit the same climate induced annual cycles of growth and decay that occur in Northern Hemisphere lakes but rather exhibit year round growth with periodic, non seasonal die-back (Howard-Williams *et al* 1987). In Lake Henley the *E. canadensis* did exhibit marked changes in biomass, but this seems to be more closely linked with the occurrence of filamentous algae than any direct seasonal influences. Pokrovskaya (1984) found that increases in large influxes of nutrients and Howard-Williams (1981) has shown experimentally that nutrient enrichment in the presence of macrophytes increases the growth of filamentous algae rather than phytoplankton. Filamentous algae is believed to reduce macrophyte growth, although it can not compete when macrophytes are rapidly growing (Pokrovskaya 1984). High temperatures that occur in floating filamentous mats can be lethal to the algae (Hillebrand 1983). It is possible that a combination of lower nutrient levels, higher water temperatures and vigorous macrophyte growth in December induced decay of the algal mats. Their reappearance at some sites in May, June and August in turn probably relates to increases in nutrient concentrations, lower temperatures and reduced macrophyte vigour.

Lake Henley is currently dominated by macrophytes, the biomass of which lake management wish to reduce by grazing with Grass Carp (Buchanan 1991; Miller 1994). The likely impact of this on the water quality of the lake is difficult to predict (Blindow *et al* 1993; Rowe & Schipper 1985). However, it would appear that the low residence time for water in the lake is likely to prevent phytoplankton blooms even in the case of severe weed reduction. The dense macrophyte beds are probably more important in stabilising the fine sediments. Shallow lakes can quickly attain a state where sediments are constantly resuspended by wave or wind action yielding turbid water, which in turn prevents the establishment, or reestablishment of aquatic plants (Hoare & Spigel 1987). The generally high water clarity of Lake Henley, despite high nutrient levels therefore seems to be a result of the high flushing rate and stabilising influence of the macrophyte beds.

In conclusion, Lake Henley is a shallow artificial lake dominated by a continuous bed of submerged exotic macrophytes, and a rapid flushing rate. Although nutrient levels are high, the macrophyte beds, associated filamentous algae and rapid flow through reduce phytoplankton abundance to yield an essentially oligotrophic lake. Seasonal fluctuations of filamentous algae associated with the macrophyte beds induced by changes in water temperature and nutrient levels appear to control the growth and decay cycle of the macrophytes by excluding light.

Chapter 2: Seasonal and spatial dynamics in the phytomacrofaunal community of a shallow New Zealand lake.

Abstract The invertebrate communities associated with dense beds of exotic macrophyte were sampled monthly between June 1993 and June 1994 from three sites in Lake Henley, a small artificial lake in the Southern Wairarapa, and from a small stream flowing into the lake. All communities were dominated by either the gastropod *Potamopyrgus antipodarum* or by oligochaetes. Community composition differed more between seasons than spatially, despite the fact some sites had lower species diversity. Seasonal fluctuations in the presence of filamentous algae on the macrophyte beds seems the most likely explanation for the observed changes in community structure possibly because it altered the structural complexity, provided additional food resources, and encouraged macrophyte decay thereby increasing palatability of macrophyte material. Communities sampled from the stream site, although sharing some taxa with the lake communities, were quite distinctive. Although the diversity of the lake phytomacrofaunal community was comparable to similar studies in other New Zealand lakes, the community had a lower proportion of insect taxa perhaps because of the low floral diversity in the lake, the lack of depth variation, the absence of a nearby colonising source and the young age of the lake.

Keywords: artificial lake; aquatic invertebrate communities; *Elodea canadensis*; macrophytes; phytomacrofauna.

INTRODUCTION

Tall growing macrophytes are a dominant feature of the littoral zone in a wide variety of lakes, and increasingly so in many New Zealand lakes where exotics have invaded (Howard-Williams *et al* 1987). Despite this, the ecological importance of submerged macrophytes in freshwater systems has only recently been recognised (Carpenter & Lodge 1986), particularly with respect to their role as substrates for invertebrate communities (Beckett *et al* 1992a; 1992b; Botts & Cowell 1993; Cyr & Downing 1988a; 1988b; Lodge *et al* 1988).

New Zealand invertebrate communities associated with submerged macrophytes in particular have been very poorly studied, at least when compared to research on other lake benthic communities. This has been variously attributed to sampling difficulties and the problem of comparing studies with varying methodology (Talbot & Ward 1987; Winterbourn & Lewis 1975). Studies in New Zealand have been concentrated in the South Island, in particular, Lake Grasmere (Stark 1981), Lake Alexandrina (Talbot & Ward 1987), Lakes Aviemore and Waitaki (Greig 1973), and surveys of seventeen South Island lakes (Stark 1993), and of the Upper Clutha Valley (Biggs & Malthus 1982). Research into the macrophyte associated communities in North Island lakes has been restricted to a qualitative survey of dune lakes (Cunningham *et al* 1953), and an evaluation of the effects of changing water levels in Lake Waikaremoana (Mylechreest 1978). Furthermore, despite an abundance of shallow lakes in New Zealand with large areas of submerged vegetation, the majority of the above studies have being conducted in large South Island lakes where the zone of submerged vegetation forms a small fringe around a body of deeper water.

In this chapter I attempt to redress this deficit by describing spatial and seasonal patterns in the invertebrate communities associated with submerged aquatic plants in Lake Henley. Lake Henley differs markedly from other lakes in which the phytomacrofauna have been investigated because it is a young, shallow lake heavily dominated by a single macrophyte taxa, *Elodea canadensis*. Invertebrate communities associated with macrophytes in a small inflowing stream were also examined to evaluate the possible effects of plant species on the invertebrate community - the stream macrophyte flora being dominated by *Potamogeton crispus*.

STUDY AREA

Lake Henley is a shallow artificial lake on the outskirts of Masterton : altitude (114 m a.s.l), maximum length (600 m), maximum depth (3 m), mean depth (1.5 m), volume (223 000 m^3), surface area (11.14 ha) and age (8 years) (Fig. 1.1). It was constructed by the Lake Henley Trust to provide Masterton with a recreational water body, a project that was initiated in 1966 and completed by 1986. A diversion of a section of the Ruamahunga River provides a managed inflow of water which is subsequently returned to the river via an outflow on the opposite side of the lake. Residency time is consequently low and estimated to be about five days. The rapid flow through results in high optical water quality, and this coupled with the shallow nature of the lake allows a dense canopy of *Elodea canadensis*, with scattered clumps of *Potamogeton crispus,* and *Nitella* sp., to grow across the entire lake bed.

Three sites within the lake were sampled (Fig. 1.1): a site situated in the channel between the inflow and the outflow just off a central island (designated as Mid lake), a site at the end of the lake which appeared to have less flow interference (End lake), and a site on the side of the lake immediately below Hiona Stream (Side lake). The Side lake site was characterised by a greater depth of silt beneath the weed beds, and initially, greater levels of filamentous algae present on the macrophytes. Environmental characteristics of each of the sampling sites are listed in Table 1. Weed samples collected from the lake were almost exclusively *Elodea canadensis* except at the Mid and Side lake sites when sufficient *Potamogeton crispus* was found to allow its sampling in February 1994, and in April and May 1994 at the End lake site.

Hiona stream, a small creek (width 2.5 m, average discharge 39.0 l/s) which flows into the lake and was sparsely covered with *Potamogeton crispus*, was also sampled for the first ten months of the study period before the weed beds were removed by council staff.

Table 1. Mean environmental measurements (± 1 SE) recorded in Lake Henley and Hiona Stream at the sampling sites between June 1993 and June 1994.

	Mid lake	End lake	Side lake	Hiona Stream
Temperature (°C)	7.7-22.0	8-22.2	7.6-22.6	7-19.2
Dissolved oxygen (mg/l)	10.48 ±2.1	11.11 ±2.5	10.81 ±2.7	10.93 ±1.3
N-NO_3 (mg/m^3)	161.5 ±150	134.1 ±190	218.7 ±147	673 ± 558
DRP (mg/m^3)	26.4 ±15	24.3 ±11	29.6 ±15	192.6 ±470
Conductivity (µs/cm)	107.6 ±61	118.5 ±64	117.4 ±64	114.2 ±58
pH	8.41 ±3.7	8.59 ±4.3	8.45 ±4.3	6.98 ±3.2
Chlorophyll-*a* (mg/m^3)	0.82 ±0.38	1.02 ±0.52	1.63 ±0.70	-

METHODS

Sampling protocol

Three replicate samples were collected from each site monthly from June 1993 to June 1994. All plants within a 0.02 m^2 area were removed at substrate level after placing a cylinder-sampler (300 µm mesh, 0.15 m diameter) over the macrophyte bed. The lake samples were all collected from a depth of 1.5 m, a depth representative of the rest of the lake, where the height of the weed beds ranged from 0.7 - 1.2 m. Samples were stored at 5°C until processing, always within two days, and invertebrates were fixed with 5% formalin. Plant material was separated from the invertebrates by washing over an 8 mm Endecott sieve, and dried for 4-5 days at 80°C to a constant weight. Invertebrates and detritus were sieved into 2 mm, 1 mm and 500 µm fractions and stored in 70% alcohol until identified and counted. Invertebrates in the 2 mm and most of the 1 mm fractions of each sample were sorted by eye in a white tray, while the 500 µm and remaining 1mm fractions were subsampled using a cylindrical sample splitter and sorted under a dissection microscope (40x) in a Bogorov counting tray (Gannon 1971). Invertebrates were identified to species wherever possible using the keys of Winterbourn and Gregson (1989), Stout (1976), Chapman and Lewis (1976) and Winterbourn (1973) and to morphospecies when more accurate identification would have proved too difficult.

At the time of collecting macrophyte samples, dissolved oxygen (YSI model 58 dissolved oxygen meter), conductivity (Orion model 122 conductivity meter), water temperature (Orion model 122 meter), and pH (Solstat pH/MV meter) were also measured at each site. Water samples were collected in acid washed polythene bottles, filtered through Whatman GFC filters (1.2 μm), and frozen prior to analysis. The nitrate nitrogen concentration of water samples collected from June 1993 to October 1993 was determined by the colourmetric autoanalyses method (Techicon), and inorganic phosphorus was determined on filtered (0.45μm) samples by the method of Murphy and Riley (1962). Dissolved reactive phosphorus (DRP) and nitrate nitrogen (NO_3-N) concentrations for the remainder of the sampling period were determined using an automated colorimetric technique (Downs 1978a; 1978b). Chlorophyll-*a* of surface water samples collected at each lake site was determined by acetone extraction of crushed GFC filters, direct spectrophotometer measurements, and calculations (Pridmore *et al*, 1983).

Data analysis

Species richness was assessed using the number of taxa, Margalef's index and rarefaction. Species number is the simplest measure of species richness but is dependant on the number of individuals in each sample. Margalef's index (Clifford & Stephenson 1975), calculated as

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

(where S is the number of taxa, and N is the total abundance), attempts to account for differing sample sizes but presupposes a functional relationship between S and N (Ludwig & Reynolds 1988). Rarefaction does not rely on such a relationship but still accounts for differences in total number of individuals (Hurlbert 1971). Thus the species richness of samples with differing abundances, from samples containing different biomasses of macrophyte, were able to be compared more accurately. The rarefacted species number in a sample of n individuals, was determined with Hurlbert's (1971) rarefaction equation -

$$E(S_n)=\sum_{i=1}^{s}\{1-[\binom{N-n_i}{n}/\binom{N}{n}]\}$$

(where n_i is the number of individuals in the ith species). This was performed using a modified version of the RAREFAC.BAS computer program of Ludwig and Reynolds (1988).

The other component of diversity examined was equatibility or evenness. Again two indices were used to assess slightly different components of this measure, the Berger-Parker index (Berger & Parker 1970) which measures the numerical dominance of the commonest animal, and Simpson's index (Simpson 1949) which weights the entire community more evenly. The Berger-Parker index is given by:

$$D_E=\frac{n_{\max}}{N}$$

(where n_{max} is the number of individuals of the most common species, and N = total number of individuals),
and Simpson's index by:

$$\hat{\lambda}=\sum_{i=1}^{s}\frac{n_i(n_i-1)}{N(N-1)}$$

(where n_i = the number of individuals in the ith species, and N = total number of individuals). The two evenness indices were expressed as reciprocals so that increasing index values indicate increasing diversity.

To assess whether sites and seasons differed with respect to univariate community characteristics all variables were analyzed with a two level mixed model analysis of variance (ANOVA) using SAS (SAS 1988). SITE was treated as a fixed factor, and MONTH as a random factor, with SITE*MONTH as the appropriate error term for testing site effects (Sokal & Rohlf 1981). Spearman rank (r_s) correlations were performed on $\log_{10}(x+1)$ transformed data using the CORR procedure of SAS (SAS

1988) to examine links between the biological and physical variables associated with the samples. Species and/or environmental variables were correlated with the three main DECORANA axes using the SAS (SAS 1988) CORR procedure.

Patterns in overall community structure were examined using the PATN multivariate statistics package (Belbin 1993). Invertebrate abundances were standardised using a $\log_{10}$ (x+1) transformation. Distance matrices were calculated using the Bray and Curtis association measure (Beales 1984), and clustered using Ward's clustering algorithm (Milligan 1979). Ordination analysis was performed with detrended correspondence analysis (DECORANA) using the DCR procedure of PATN (Belbin 1993).

RESULTS

Diversity

A total of 48 macroinvertebrate taxa were collected from the study sites over the 13 month period, 44 from the lake sites, 42 from the stream site with 39 common to both habitats. Margalef's index and the rarefacted species number both indicated significantly lower species richness at the Side site than at the other three sites ($F_{3,33} = 5.2$ $P = 0.005$, and $F_{3,33} = 5.2$ $P = 0.005$ for Margalef's index, and the rarefacted species number, respectively), although there was no significant difference in species number between sites ($F_{3,33} = 1.7$ $P = 0.19$) (Fig. 2.1). However, significant interactions between SITE and MONTH for both Margalef's index and for the rarefacted species number suggest site differences vary with season ($F_{33,98} = 3.49$ $P < 0.001$, and $F_{33,98} = 3.24$ $P < 0.001$, respectively). Overall there appears to be a trend for species richness to increase in the summer months (Fig. 2.2) and all three measures of species richness were correlated with water temperature (Table 2).

Table 2. Correlation coefficients between biological indices and physical measurements made at the lake and stream sites between June 1993 and June 1994. * = $P < 0.05$

	Temp	DO	Algae	$N\text{-}NO_3$	DRP	pH
Abundance	0.18	-0.47*	0.50*	0.09	0.12	-0.16
Number per gram	0.23	-0.52*	0.22	-0.04	0.35	-0.3
Species no.	0.48*	-0.3*	0.24	-0.48*	0.12	-0.18
Margalef's	0.39*	-0.14	0.07	-0.48*	0.09	-0.13
Rarefacted species no.	0.38*	-0.17	0.16	-0.46*	0.08	-0.07
Simpson's	0.54*	-0.18	0.21	-0.49*	0.00	0.07
Berger-Parker	0.48*	-0.13	0.17	-0.40*	-0.03	0.07
Insect abundance	0.43*	-0.45*	0.26	-0.31*	0.19	-0.40*
Crustacean abundance	0.18	-0.42*	0.60*	-0.01	0.25	-0.04
Acarina abundance	0.35*	-0.26	0.49*	-0.30*	-0.3	0.35*
Annelida abundance	0.19	-0.29*	0.38*	0.04	0.13	-0.04
Mollusca abundance	0.01	-0.37*	0.17	0.04	-0.21	0.00
Coelenterate abundance	0.06	0.21	0.05	-0.08	0.30	0.10

There was a significant difference in evenness, as measured by both Berger-Parker and Simpson's indices between the four sites, with the Side site most heavily dominated by oligochaetes ($F_{3,33} = 5.7$ $P = 0.003$, and $F_{3,33} = 8.12$ $P = 0.003$ for the Berger-Parker and Simpson's indices, respectively) (Fig. 2.3). Once again, significant SITE*MONTH interactions suggested that differences in site evenness vary with season ($F_{33,98} = 2.3$ $P<0.001$, and $F_{33,98} = 2.66$ $P < 0.001$, respectively). As with species richness, community evenness peaked in summer and decreased in the cooler months towards a community type more heavily dominated by fewer taxon (Fig. 2.4). Both the Berger-Parker and Simpson's indices were correlated with temperature (Table 2).

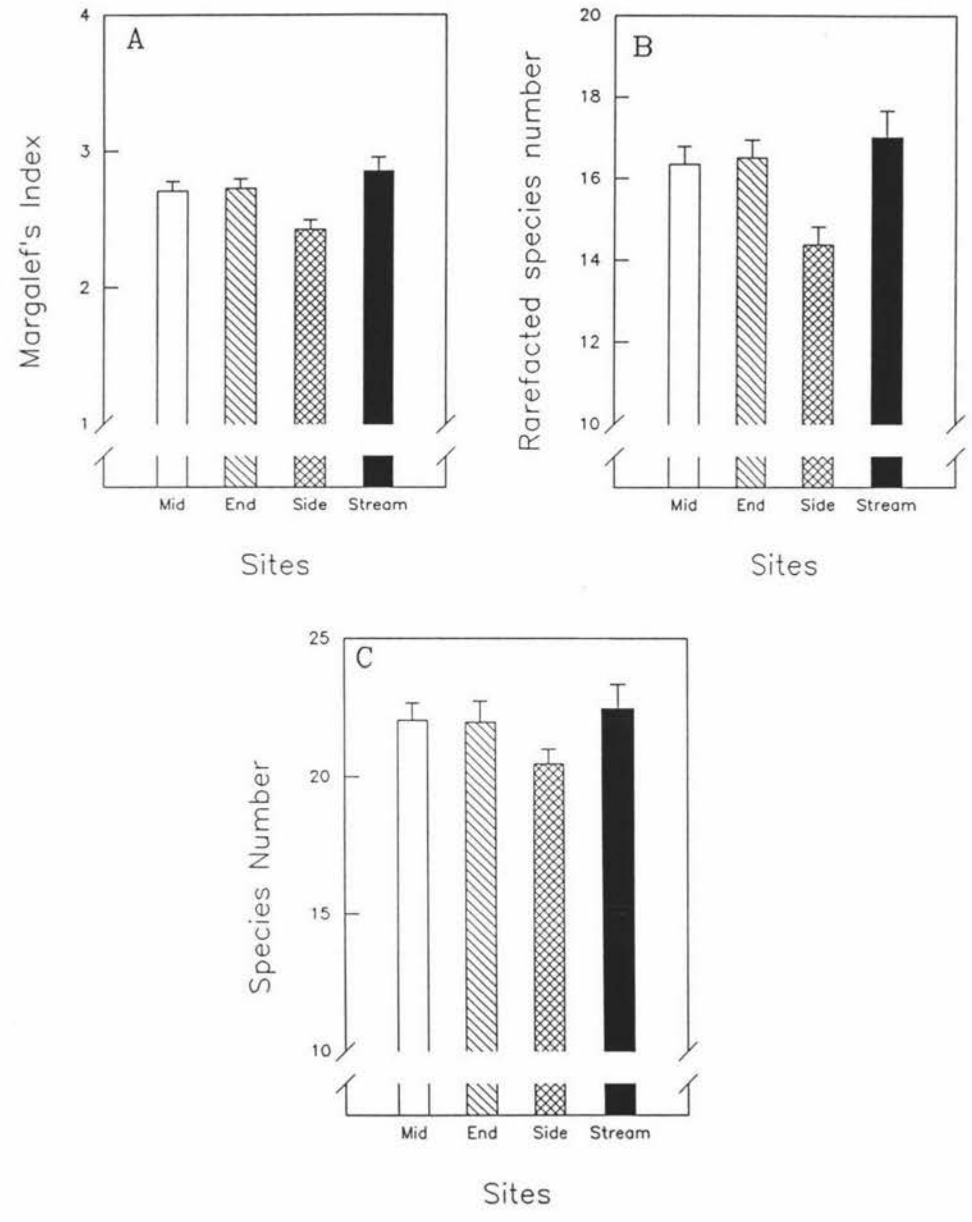


Fig. 2.1. Mean species richness measures (± 1 SE) from 3 replicate weed samples collected at the 3 lake study sites between June 1993, and June 1994, and the stream site between June 1993 and March 1994. **A**. Margalef's index, **B**. Rarefacted species number and **C**. Species number.

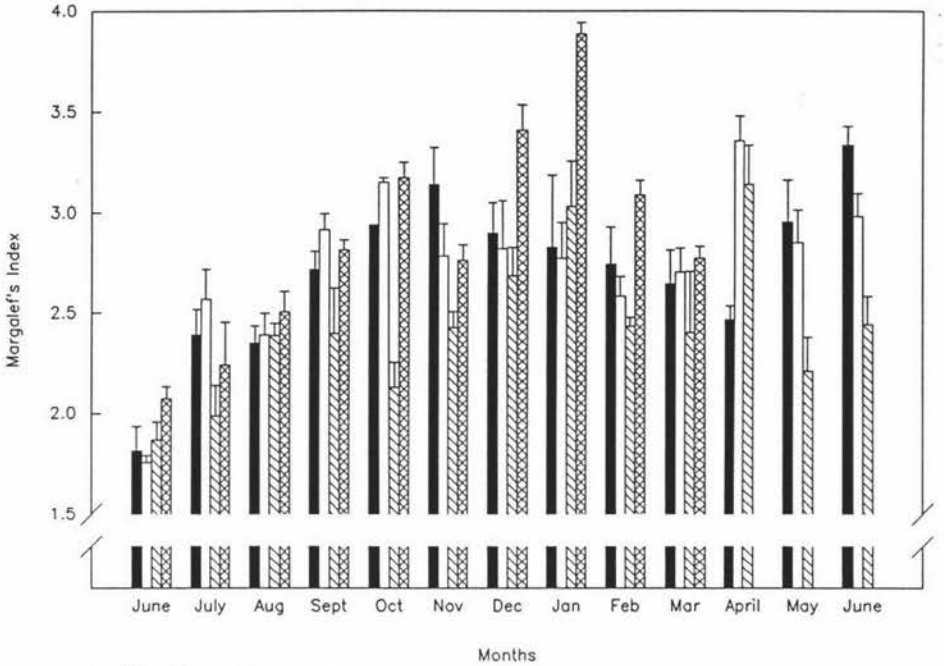


Fig. 2.2. Mean Margalef's index measures (± 1SE) from three replicates collected at each lake site for each month between June 1993 and June 1994, and at the stream site for the months of June 1993 to March 1994.

( ) Mid Lake ( ) End Lake ( ) Side Lake ( ) Stream.

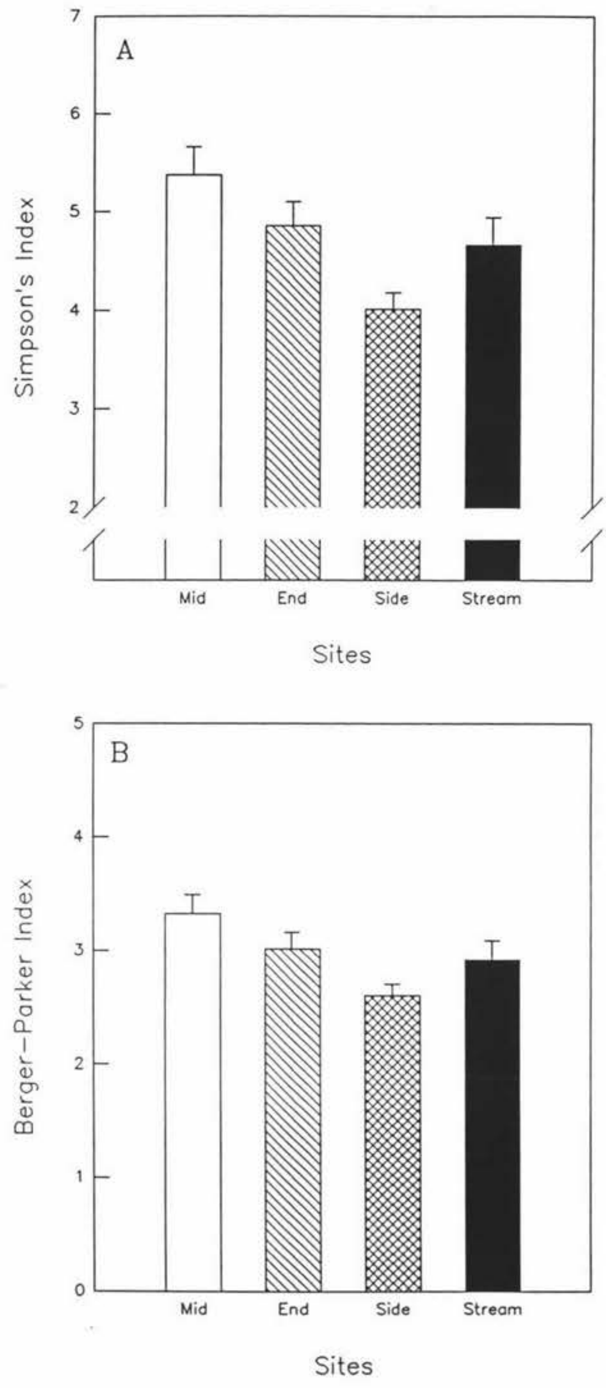


Fig. 2.3. Mean evenness measures (± 1 SE) from 3 replicate weed samples collected at the 3 lake sites between June 1993, and June 1994, and the stream site between June 1993 and March 1994. **A**. Berger-Parker index and **B**. Simpson's index.

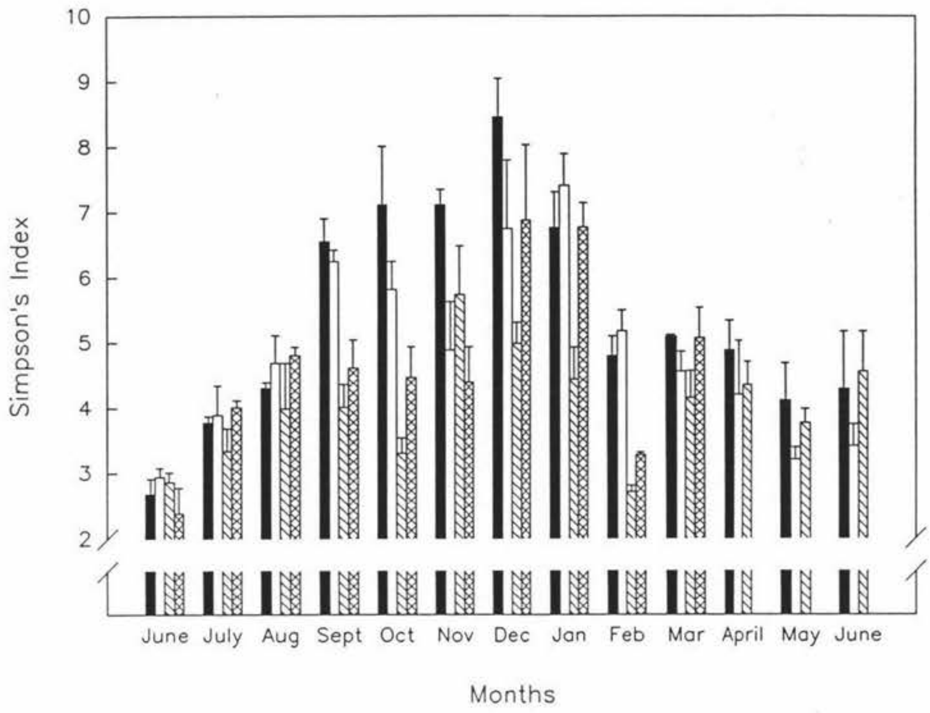


Fig. 2.4. Mean Simpson's index (± 1SE) from three replicates collected at each lake study site for each month between June 1993 and June 1994, and at the stream site for the months of June 1993 to March 1994.

( ) Mid Lake ( ) End Lake ( ) Side Lake ( ) Stream.

The total number of invertebrates collected per sample ranged from 317 to 10 578. The Stream site had a significantly lower total invertebrate abundance than the other three sites ($F_{3,33} = 6.95$ $P < 0.001$), due to the significantly lower biomass of weed collected from this site ($F_{3,33} = 10.91$ $P < 0.001$). When abundance was expressed as the number of individuals per gram of dried macrophyte, thus attempting to correct for this difference in sample size, there was no significant difference between sites ($F_{3,33} = 0.6$ $P = 0.62$). Interestingly, the number of invertebrates was negatively correlated with dissolved oxygen and positively related to the presence of filamentous algae (Table 2).

Community composition

Invertebrate communities from all sites were in general dominated by either the gastropod *Potamopyrgus antipodarum*, or oligochaetes. Molluscs comprised between 7 and 87%, and annelids between 2 and 84% of the total abundance. Abundances of each of the main taxonomic groups differed between the four sites (Fig. 2.5A). The Stream site had a greater number of insects than either the Side or Mid lake sites ($F_{3,33}$ = 6.37 P = 0.002) and a significantly lower number of molluscs, Acarina and crustaceans than all the lake sites ($F_{3,33}$ = 4.33 P = 0.01, $F_{3,33}$ = 69.63 P < 0.001 and $F_{3,33}$ = 3.48 P = 0.027, for molluscs, Acarina and crustaceans, respectively). The Mid and End lake sites did not differ from each other in their taxonomic make up, however, the Side lake site had a greater number of annelids ($F_{3,33}$ = 4.33 P = 0.01) and a lower number of insects ($F_{3,33}$ = 6.37 P = 0.002) than the other three sites. Insect and Acarina abundance were positively correlated with temperature, crustacean and insect abundance were negatively correlated with dissolved oxygen, and numbers of crustaceans and Acarina were related significantly to the presence of filamentous algae (Table 2).

The relative abundance of each major taxonomic group is presented in Fig. 2.5. The relative abundance of insects decreased from highest to lowest in the order Stream, End, Mid, and Side sites ($F_{3,33}$ = 18.84 P < 0.001), although significant SITE*MONTH interactions suggested that this difference depended on season ($F_{33,98}$ = 4.32 P < 0.001). Crustacean relative abundance was significantly correlated with the presence of filamentous algae (Table 2) but was not significantly different between sites ($F_{3,33}$ = 0.72 P = 0.55). The Mid and End sites had a higher relative abundance of molluscs than the other two sites ($F_{3,33}$ = 3.04 P = 0.04) and the Side site had a significantly lower proportion of Aracina than the other three sites ($F_{3,33}$ = 32.58 P = 0.001).

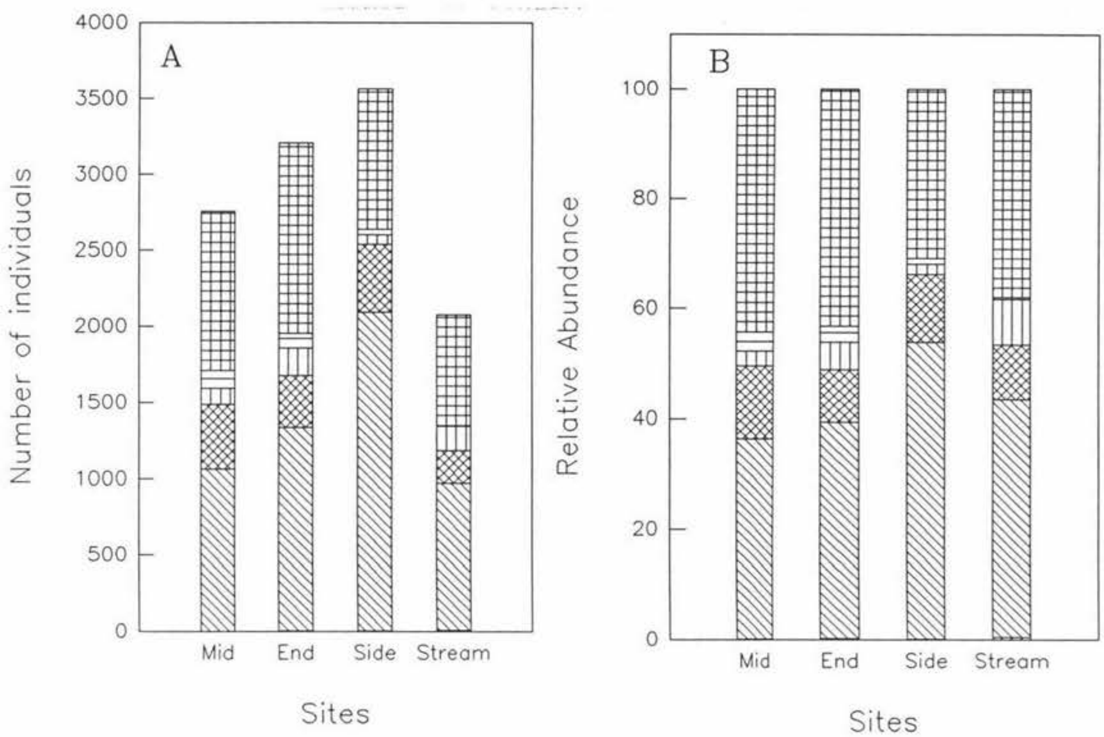


Fig. 2.5. Abundance of individuals in the major taxa groups represented as **A**. Mean numbers of individuals, and **B**. Mean relative abundance. Mean values obtained from 13 months at each lake site (June 1993 to June 1994), and 10 months at the stream site (June 1993 to March 1994).

( ) Coelenterata ( ) Annelids ( ) Crustacea

( ) Insecta ( ) Acarina ( ) Mollusca

Overall community structure

Examination of overall community structure using DECORANA split the samples into two broad groups: the stream communities, and all the lake communities which exhibited no tendency to group according to site (Fig. 2.6). Axis 1 therefore appeared to separate the communities primarily according to habitat type, although four environmental variables did correlate significantly, albeit weakly, with Axis 1. These were biomass of macrophyte ($r_s = -0.33$ $P = 0.015$), the presence of filamentous algae ($r_s = -0.38$ $P < 0.01$), both of which reflect lower macrophyte biomass and an absence of filamentous algae in the stream, dissolved oxygen ($r_s = 0.29$ $P = 0.035$) and DRP ($r_s = 0.37$ $P < 0.01$).

Not surprisingly, species correlated positively with Axis 1 (Table 3) were those more commonly found in the stream communities, whereas those species negatively correlated were those associated with the lake communities. These included cladocerans, copepods and mites - taxa commonly found in pelagic zooplankton communities.

The lake communities also appeared to grade from communities collected between June and November 1993 to those collected between December 1993 and June 1994, separating the lake communities weakly into two large seasonal groups (Fig. 2.6). Axis 2 correlated with variables associated with this seasonal change, i.e., temperature ($r_s = -0.29$ $P = 0.03$), the presence of filamentous algae ($r_s = 0.47$ $P < 0.01$), and NO_3-N concentrations ($r_s = 0.49$ $P < 0.01$). Species positively associated with Axis 2 (Table 3) were those species more common in the first six months of the sampling period (June 1993 to November 1993) while those with a negative association were those species associated with the latter half of the sampling period (December 1993 to June 1994) such as *Paroxyethira* sp. Interestingly, the three collections of *Potamogeton crispus* from the lake sites clustered closely with the other lake sites and not with the stream samples from the same macrophyte species. Axis 3 also appeared to be associated with seasonal changes in community composition, and was again correlated with temperature ($r_s = -0.44$ $P < 0.001$), and NO_3-N concentrations ($r_s = -0.43$ $P = 0.002$).

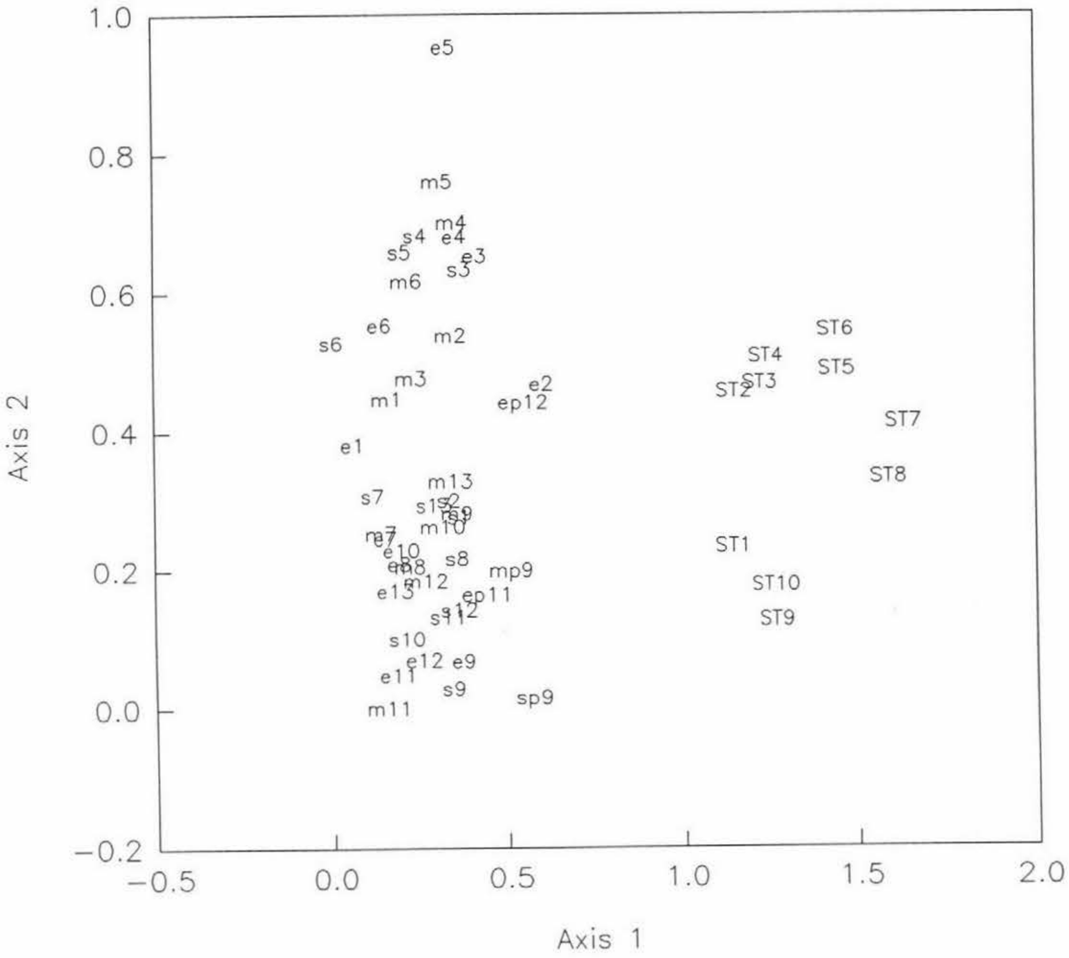


Fig. 2. 6. Axis 1 of a detrended correspondence analysis as a function of Axis 2 for the lake communities collected from the three sites in Lake Henley for 13 months, and Hiona Stream for 10 months. Each symbol represents the average of the 3 replicate macrophyte samples collected at each site.

M1-13, E1-13, S1-13 = Mid, End and Side lake communities collected from *Elodea canadensis* during June 1993 to June 1994. ST1-10 = Stream communities collected from *Potamogeton crispus* during June 1993 to March 1994. MP9 and SP9 = Lake communities from *Potamogeton crispus* from the Mid and Side lake sites, collected in February 1994.

EP11 and EP12 = Lake communities from *Potamogeton crispus* from the End lake sites, collected in May and June 1994.

Table 3. Taxa associated with the three main axes from a DECORANA analysis of invertebrate communities collected at the lake and stream sites between June 1993 and June 1994.

AXIS I	AXIS II	AXIS III
Positive:		
Sphaerium novaezelandiae	*Lumbriculus variegatus*	
Ostracoda sp.	*Xanthocnemis zealandica*	
Simuliidae	Ceratopogonidae	
	Bosminidae	
Negative:		
Gyraulus corinna	Platyhelminthe sp. A	*Chironomus zealandicus*
Oribatidae	*Glossiphonia*	Orthocladinae
Acarina sp.C	*Paroxyethira* sp.	*Chaetogaster* sp.
Chydoridae		Tanypodinae
Cyclopoidae		*Paroxyethira* sp.
Antiporus sp.		

Cluster analysis also divided the stream and lake communities into two distinct groups (Fig. 2.7). However, within the lake samples five further groupings were evident again based on season. The groups comprised communities from all three lake sites from the following month groups: June and July 1993, August to October 1993, November and December 1993, January to March 1994, and April to June 1994. Communities collected from the *Potamogeton crispus* formed a separate lake group. Seasonal differentiation of communities was also evident in the stream samples which clustered into three similar seasonal groups: June to July 1993, August 1993 to January 1994, and February to March 1994.

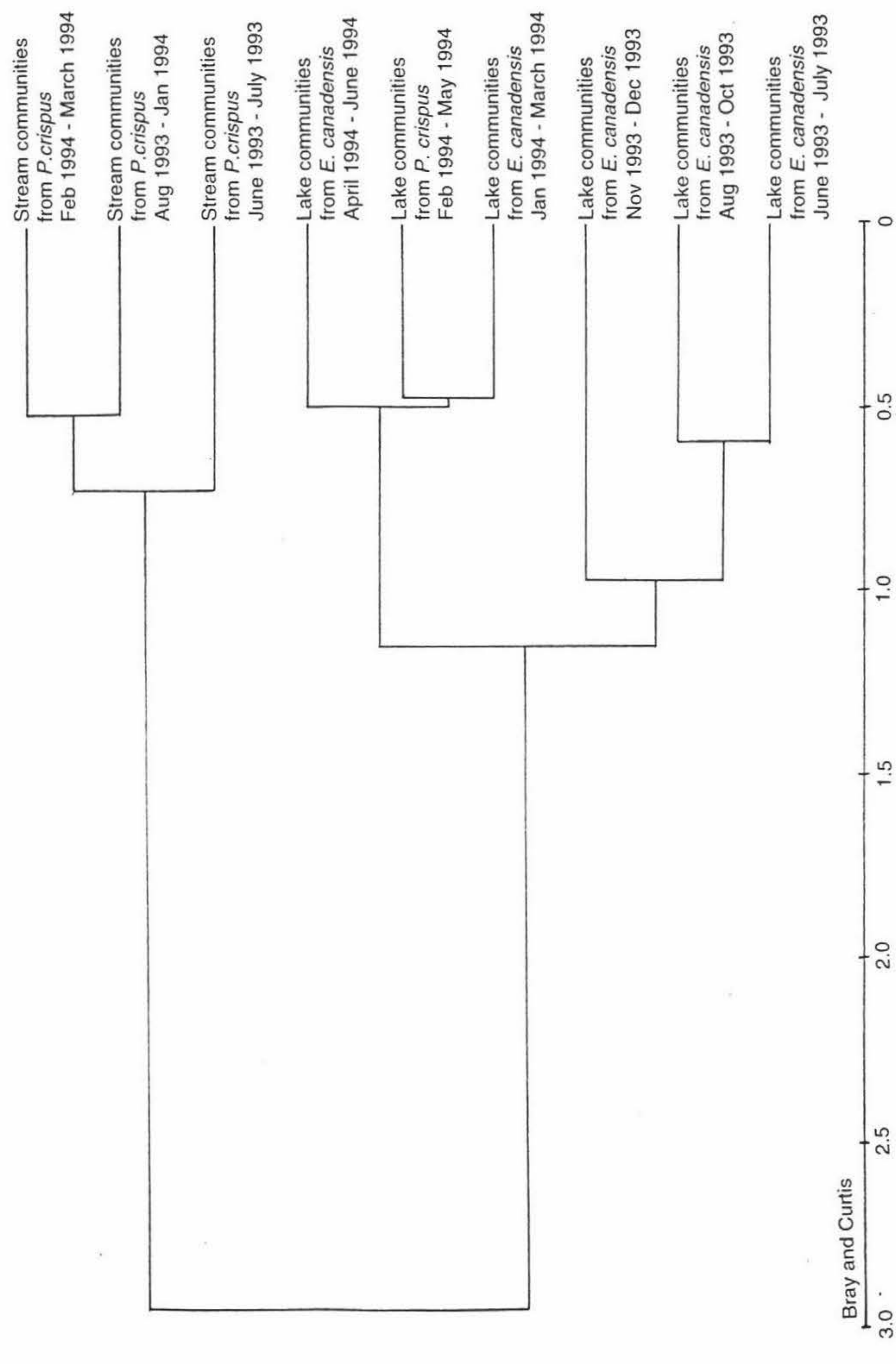


Fig. 2.7. Cluster analysis of invertebrate communities collected from macrophytes at the three lake sites in Lake Henley between June 1993 and June 1994, and from Hiona Stream between June 1993 and June 1994.

DISCUSSION

The phytomacrofauna of Lake Henley

The invertebrate communities associated with the submerged macrophytes in Lake Henley were characterised by high densities of invertebrates but low diversity. The relatively low species richness of the communities supports the generally held view that macrophyte invertebrate communities are relatively species poor (Talbot & Ward 1987; Winterbourn & Lewis 1975) and the high numerical dominance of either *Potamopyrgus antipodarum*, or oligochaetes is consistent with other studies of New Zealand phytomacrofaunal communities (Biggs & Malthus 1982; Stark 1981; 1993; Talbot & Ward 1987). Communities did exhibit some spatial variation between the lake sites particularly with respect to species richness and the level of dominance by certain taxa, however, seasonal influences seemed much more important in dictating differences in overall community structure than spatial ones.

The most notable intra-lake site differences were found at the Side site where greater abundance of filamentous algae and a deeper silt layer were associated with lower species richness and evenness, a greater number of annelids, and a lower number of insects. Minshall (1984) concluded that heavy silting generally resulted in lower insect species diversity and productivity. Oligochaetes and chironomids have been observed to increase with siltation in both regulated rivers (Ward 1984) and lakes with fluctuating water levels (Hunt & Jones 1972). Silt retention of macrophyte beds is also increased by filamentous algae (Paterson 1993; Petr 1968), further increasing this sites suitability as habitat for oligochaetes.

The small size of the lake, and the monoculture of *Elodea* probably contributed to the general homogeneity of communities found at the three lake sites. The low species richness of these communities perhaps precluded the likelihood of much spatial variation occurring in a lake of this size. Temporal influences, rather than spatial differences, had a greater impact on overall community composition. Talbot and Ward (1987) also found this to be the case with overall invertebrate abundances peaking in late summer, and oligochaete abundance in particular was found to be linked to temperature increases.

The seasonal dynamics of filamentous algae associated with the macrophytes is also likely to have influenced the seasonal dynamics of the invertebrate communities. Algal mats spread across the lake from the Side site, and covered all areas of the lake between September and December 1993 (Chapter One). The observed increases in invertebrate abundance may be attributed to the presence of filamentous algal mats providing an increased surface area for both invertebrates and attached algae to colonise (Botts & Colwell 1993; Campbell & Clark 1983). Although it is generally not considered a food resource for aquatic invertebrates, Hydroptilid caddis flies are an exception (Winterbourn & Gregson 1989) and were present in high numbers in December when algae mats were particularly thick. The filamentous algae may also affect the invertebrate community indirectly by increasing macrophyte palatibility through enhancing its rate of decay. Macrophytes covered with the algal mats were observed to became etiolated, and regained vigorous growth only after the algal mats dissipated. Several studies have found increased abundance of invertebrates with the greater palatibility of macrophytes produced by decomposition (Smock & Stoneburner 1980; Beckett *et al* 1992a; Botts & Cowell 1993).

Lake Henley compared to other New Zealand lakes

It is difficult to make comparisons with other studies on lake phytomacrofauna communities because of the different sampling techniques, taxonomic levels, and sieve sizes used. However, despite the young age of Lake Henley, the lack of a nearby colonising source, the monoculture of exotic weed that dominates the lake bed, and the constant shallow nature of the lake, the total number of invertebrate taxa found in its phytomacrofaunal communities is comparable to those found in the littoral zone of other lakes in New Zealand. Table 4 presents the number of invertebrate taxa found in lakes with flora dominated by *Elodea canadensis*. The markedly lower species richness recorded in Lake Alexandrina and the Upper Clutha studies probably reflects the clumping of all Chironomidae and the larger sieve size used (1 mm) which would have excluded any Crustacea.

However, the lower diversity of insect taxa evident in Lake Henley compared to the other lakes is possibly because of the narrow range of depths in Lake Henley and/or to the homogenous nature of the monoculture of *Elodea*. Stark (1993) observed that species richness of invertebrates associated with macrophytes increased with depth down to at least 4 or 5 m - much deeper than the average 2 m depth of Lake Henley, a characteristic which was particularly marked in natural verses hydro lakes. Brown *et al* (1988) found that the greatest factor influencing the diversity and abundance of phytomacrofaunal communities in Lake St. Clair was the taxonomic diversity within the macrophyte stands. They attributed the low species diversity and abundance found to the vertical homogeneity of tall vascular macrophyte stands, similar to those found in the dense beds of *Elodea* in Lake Henley. Studies from other aquatic and terrestrial systems also support the view that species richness decreases as spatial heterogeneity is reduced (Begon *et al* 1990), and in particular insect diversity decreases as plant structural complexity declines (Lawton 1983).

The lower diversity of insect taxa recorded for Lake Henley, particularly the Trichoptera and Chironomidae, could also relate to the colonisation process of a newly created lake. Invertebrates associated with lotic environments could easily colonise the lake via the inflowing streams or river, while others may have been carried there as eggs on the feet of aquatic birds. Larvae of insects that rely on colonisation by adults are however likely to encounter greater difficulties in reaching a new habitat such as Lake Henley if their adults do not disperse widely (Sheldon 1984).

The dominance of the hydrobiid gastropod, *Potamopyrgus antipodarum* in Lake Henley has also been recorded in a number of other studies on macrophyte invertebrate communities in New Zealand (Biggs & Malthus 1982; Mylechreest 1978; Talbot & Ward 1987), although Stark (1993) found oligochaetes to be most dominant in all 17 South Island lakes he surveyed. Stark (1981), Mylechreest (1978) and Biggs and Malthus (1982) found that chironomids were often co-dominant with *Potamopyrgus antipodarum*, and in Lake Alexandrina, either *Gyraulus corinna*, chironomids, or oligochaetes were the co-dominant, or sub-dominant species (Talbot & Ward 1987). Not only does diversity in general appear to be low in New Zealand lake phytomacrofaunal communities but only a narrow range of taxa are found to be dominant in any of these communities.

Table 4: Diversity of phytomacrofaunal communities associated with *Elodea canadensis* in New Zealand lakes as recorded in a number of studies in the literature. Data from Biggs and Malthus (1982)[1], Talbot and Ward (1987)[2], Stark (1981)[3] and Stark (1993)[4]. Annelid and Crustacean taxa from Lake Henley have been clumped to allow comparison with other studies.

Taxa	Henley	Upper Clutha (1982)[1]	Alexandrina (1987)[2]	Grasmere (1981)[3]	Rotoiti (1993)[4]	Rotoroa (1993)[4]	Pearson (1993)[4]	Ohau (1993)[4]	Benmore (1993)[4]	Hawea (1993)[4]	TeAnau (1993)[4]	Manapouri (1993)[4]	Monowai (1993)[4]
Hirudinea	1	0	1	1	1	0	1	0	1	0	0	0	0
Annelidea etc	5	1	2	4	4	4	3	5	3	4	4	2	2
Arachnida	4	0	1	1	1	3	2	1	2	1	4	3	2
Insecta	18	11	10	22+	19	29	26	27	18	25	22	20	15
Mollusca	6	5	5	4	4	7	6	6	7	6	4	6	5
Coelenterata	1	0	0	2	0	0	1	0	1	0	0	1	0
Crustaceans	3	0	0	2	2	6	7	4	7	7	3	2	1
TOTAL TAXA	36	17	19	36	31	49	47	41	41	42	36	36	25

Lake macrophyte communities compared to stream communities

The communities associated with macrophytes in the stream differed, not unsurprisingly, from the lake communities. However, this difference cannot be attributed purely to differences in macrophyte species because communities sampled from lake *Potamogeton crispus* appeared more similar to the other lake communities than to those sampled from this macrophyte species in the stream. The stream communities were characterised by having a lower abundance of invertebrates than the lake sites, a lower number of molluscs, Acarina and crustaceans and a greater proportion of insects. A species of Sphaeriidae, a filter feeding clam, was strongly associated with the stream communities. Its rarity in some lake macrophyte communities has been linked to the lack of water movement at the base of tall dense plants (Talbot & Ward 1987). This could also explain the abundance of other filter feeding taxa such as ostracods and simuliids in the stream phytomacrofaunal community but not in the lake communities. Knott *et al* (1978) found the benthic invertebrate fauna of a Tasmanian stream richer in terms of both numbers of species, numbers of insects and density of organisms than similar communities in an associated lake. However, differences in the composition and structure of invertebrate communities in lentic and lotic habitats appear to be largely unexplored, especially in New Zealand.

In summary, the phytomacrofaunal community of this newly created artificial lake reflects some of the characteristics inherent in the invertebrate communities associated with macrophytes in other New Zealand lakes. These include a generally low species diversity, low spatial variation and numerical dominance by one or two taxa, in this case *Potamopyrgus antipodarum*. Differences in its species richness with other surveyed lakes seem largely attributable to its lack of floral diversity and depth variation although the temporal dynamics of the colonisation process may also have contributed to the low insect diversity. Seasonal, rather than spatial influences seem more important in determining patterns in community structure, again probably because of the generally low diversity of invertebrates in lake phytomacrofaunal communities and the relative homogeneity of the floral and spatial characteristics of the lake.

Chapter 3: Phytomacrofaunal communities of 14 North Island lakes, New Zealand: the influence of trophic status.

Abstract The invertebrate communities associated with macrophytes of the shallow littoral zone of 14 North Island lakes were sampled in May 1994. There were three broad groups of phytomacrofaunal communities which corresponded approximately to lake trophic status. The communities from the lakes with the most enriched trophic status had the greatest abundance but lowest species richness, while communities from less enriched lakes had communities that were more even. The low species richness found generally precluded the use of specific indicator species, although there were associations of species linked with each of the community groups of the communities. In particular, communities from nutrient poor lakes were characterised by a greater diversity of insects.

Keywords: littoral zone; macroinvertebrates; macrophytes; North Island lakes; phytomacrofauna; trophic status.

INTRODUCTION

Submerged macrophytes fulfil a variety of roles in aquatic ecosystems, acting both as a substrate and/or a food resource for colonising invertebrates (phytomacrofauna) (Carpenter & Lodge 1986). Many factors have been linked to the composition of phytomacrofaunal communities including the species of macrophyte (Brown *et al* 1988; Chilton 1990), plant surface area and architecture (Cyr & Downing 1988a; Jeffries 1993; Petr 1968), seasonal changes in the quality of macrophyte (Schramm *et al* 1987), organic matter in sediments, water phosphorus concentration, depth, and trophic status (Cyr & Downing 1988b; Lalonde & Downing 1992). However, no single factor appears to be more important than any other in determining the abundance, or composition, of macrophyte invertebrate communities in lake littoral zones (Cyr & Downing 1988b).

The trophic status of a lake, as an index of nutrient enrichment, is a key indicator of overall productivity (O.E.C.D. 1982). The influence of increasing productivity on the structure of aquatic invertebrate communities has been examined in the benthic and profundal zones (e.g., Dougherty & Morgan 1991; Forsyth 1978; Forsyth & McCallum 1981; Johnson 1974; Saether 1980), the pelagic zone (Ravera 1980), and the littoral zone (Cyr & Downing 1988b; Lalonde & Downing 1992; Verdonschot 1992) in a variety of lakes. These studies conclude that increasing productivity leads to an increase in the abundance of invertebrates with a concurrent shift in community composition. Highly productive, or eutrophic lakes have in general been found to support a greater abundance (Dougherty & Morgan 1991; Stark 1993), and biomass of invertebrates (Forsyth 1978; Lalonde & Downing 1992; Reavell & Frenzel 1981). Changes in the dominant species and an increase in the degree of dominance in communities have also been associated with increasing nutrient enrichment (Ravera 1990; Saether 1990). Lalonde and Downing (1992) found that the biomass of chironomids, trichopterans, and oligochaetes were significantly higher in the phytomacrofauna of more eutrophic lakes, and Stark (1993) found the most productive lakes of his South Island survey were numerically dominated by molluscs (particularly *Potamopyrgus antipodarum*) and oligochaetes. While an increase in abundance or biomass with increasing productivity is not surprising, the response of species richness is less predictable and has been found to both increase (Dougherty & Morgan 1991; Stark 1993) and decrease (eg. Growns *et al* 1992; Reynolds 1984) in the face of nutrient enrichment.

An increasing demand for biological monitoring, and the recognition of the usefulness of aquatic macroinvertebrates for this task, has necessitated a greater understanding of the way invertebrate communities respond to changes in their environmental conditions (Stark 1994) and in particular, how they are effected by nutrient enrichment. Research on phytomacrofaunal communities in New Zealand has focused more directly on the influence of fluctuating water levels (Greig 1973; Mylechreest 1978; Stark 1993), macrophyte species (Stark 1981; Talbot & Ward 1987), seasonal dynamics (Stark 1981; Talbot & Ward 1987) and the impact of exotic macrophytes (Biggs & Malthus 1982; Stark 1981) on these communities. In addition, Talbot and Ward (1987) found that the invertebrate community from the mesotrophic Lake Alexandrina had a higher biomass than comparable communities from oligotrophic lakes (Biggs & Malthus 1982), and Stark (1993), in a survey of South Island lakes, also observed that the invertebrate communities from the most productive lakes had the greatest abundance and highest species richness. In this chapter I examine the phytomacrofauna of submerged aquatic plants in the shallow littoral zone of 14 North Island lakes. In particular, I explore the relationship between nutrient status and invertebrate communities associated with these submerged aquatic plants.

STUDY SITES

Fourteen lakes, encompassing a range of trophic status, from oligotrophic to eutrophic, and supporting growths of tall vascular macrophytes within the shallow littoral zone were selected for the survey. The lakes consisted of three dune lakes from the Wanganui region: Dudding's Lake, Lake Pauri and Lake Wiritoa, two landslide lakes: Lake Namunamu (Rangitikei region) and Tutira (Hawke's Bay), two hydro-lakes: Karapiro (Waikato region) and Otomanakau (Central Plateau region), seven volcanic lakes: Lake Pupuke (Auckland), Lakes Tikitapu, Tarawera, Okataina and Rotoiti (Rotorua region),and Lake Taupo (Central Plateau region), and Lake Henley, a small artificially created lake in the Southern Wairarapa (Fig. 3.1). Two sites were sampled from Lake Rotoiti: the Rotoiti site was situated off the North Western arm of the lake, and Ruato Bay site was off the bay of the same name. Three sites were sampled from Lake Henley (Mid, End and Side sites) (described in Chapter One), yielding a total of 17 lake sites. Details of the physical and environmental characteristics of each of these lakes recorded from the literature are listed in Table 1.

METHODS

Sampling protocol

The lakes were sampled between April 25 and May 6 1994. All plants within a 0.02 m^2 circular area were enclosed within a 300 μm mesh cylinder-sampler and removed at substrate level. Samples were collected

Table 1. Physical and Environmental characteristics of the lakes included in the survey. Data supplied from (1) Chapter One. (2) Livingston *et al* 1986 and references therein and (3) S. Pilkington unpublished data

	Mean depth	Maximum depth	Area	Altitude	Mean secchi	DRP	NO_3-N	Mean Ch-a	%Pasture	%Forest	%Urban	Type	Region
Henley[1]	2.5	4	0.1	114	4	17.94	203.45	1.19	33	0	33	Artifical	Wairarapa
Pupuke[2]	34	57	1.63	0.3	289.1	2.25	14.75	5.25	0	0	100	Volcanic	Auckland
Tikitapu[2]	1.8	27.5	1.4	417.8	7.5	2.4	1.3	1.99	5.9	30.5	0	Volcanic	Rotorua
Otomanakau[2]	m	9	1.8	665	4	30.5	4.2	1.37	31	57.7	0	Hydrolake	Taupo
Taupo[2]	97	162.8	622.63	357	16	1.75	3.43	1.4	21	59.1	0	Volcanic	Taupo
Karapiro[2]	m	30.5	5.37	55	2	42.8	21.8	102	57.8	39.6	0.7	Hydrolake	Waikato
Pauri[2]	6.3	14.9	0.23	46	m	13.2	89.4	5.25	100	0	0	Dune	Wanganui
Tutira[2]	21	42	1.47	150	4.02	10	12.32	3.9	90.5	0	0	Landslide	Hawkes Bay
Namunamu[3]	7.9	18.6	0.13	190	6.59	37.7	48.35	1.57	80	20	0	Landslide	Rangitikei
Duddings[2]	5	11.9	0.13	86	1.94	6.65	128.65	7.13	95.9	4.1	0	Dune	Wanganui
Tarawera[2]	50	87.5	41.02	299	7	2	1	1.64	12.1	60.2	0	Volcanic	Rotorua
Okatina[2]	39.4	78.5	10.8	311	9.15	1.7	3	1.99	8.9	74.4	0	Volcanic	Rotorua
Wiritoa[2]	9.1	19.5	0.26	44	2.4	9.4	67.6	4.3	100	0	0	Dune	Wanganui
Rotoiti[2]	31.53	93.5	33.48	279	5.57	7	4.14	6.91	33	38.2	0	Volcanic	Rotorua

from a depth of between 0.5 m and 1.5 m and fixed in the field with 5% formalin. As macrophyte species can influence their associated invertebrate communities, attempts were made to sample *Elodea canadensis*, or when this was not possible, plants of a similar structural form. The dominant macrophyte species sampled at each of the 17 lake sites is listed in Table 2. Plant material was separated from the invertebrates by washing over an 8 mm Endecott sieve, and then dried for 4-5 days at 80°C to constant weight. Invertebrates and detritus were sieved into 2 mm, 1 mm, and 500 µm fractions and stored in 70% alcohol until the invertebrates were identified and counted. Invertebrates in the 2 mm fraction of each sample were sorted by eye in a white tray, while the 1 mm and 500 µm fractions were subsampled using a cylindrical sample splitter and sorted under a (40x) dissection microscope with the aid of a Bogorov sorting tray (Gannon 1971). Invertebrates were identified to the lowest possible taxonomic level using the keys of Winterbourn and Gregson (1989), Stout (1976), Chapman and Lewis (1976), and Winterbourn (1973), or sorted to morphospecies in the case of oligochaetes.

Table 2. Macrophyte species sampled from each lake.

Lake	Dominant macrophyte	Other species sampled
Tarawera	*Lagarosiphon major*	
Tikitapu	Characean	
Taupo	*Lagarosiphon major*	*Ceratophyllum demersum*
Okataina	*Lagarosiphon major*	
Ruato Bay	*Lagarosiphon major*	
Rotoiti	*Ceratophyllum demersum*	*Egeria densa*
Pupuke	*Vallisneria gigantea*	*Myriophyllum* sp.
Tutira	*Hydrilla* sp.	
Karapiro	*Ceratophyllum demersum*	*Egeria densa*
Dudding's	*Elodea canadensis*	*Potamogeton crispus*
Wiritoa	*Myriophyllum sp.*	*Potamogeton crispus*
Pauri	*Elodea canadensis*	*Potamogeton crispus*
Namunamu	*Elodea canadensis*	*Nitella* sp.
Otomanakau	*Lagarosiphon major*	
Henley	*Elodea canadensis*	

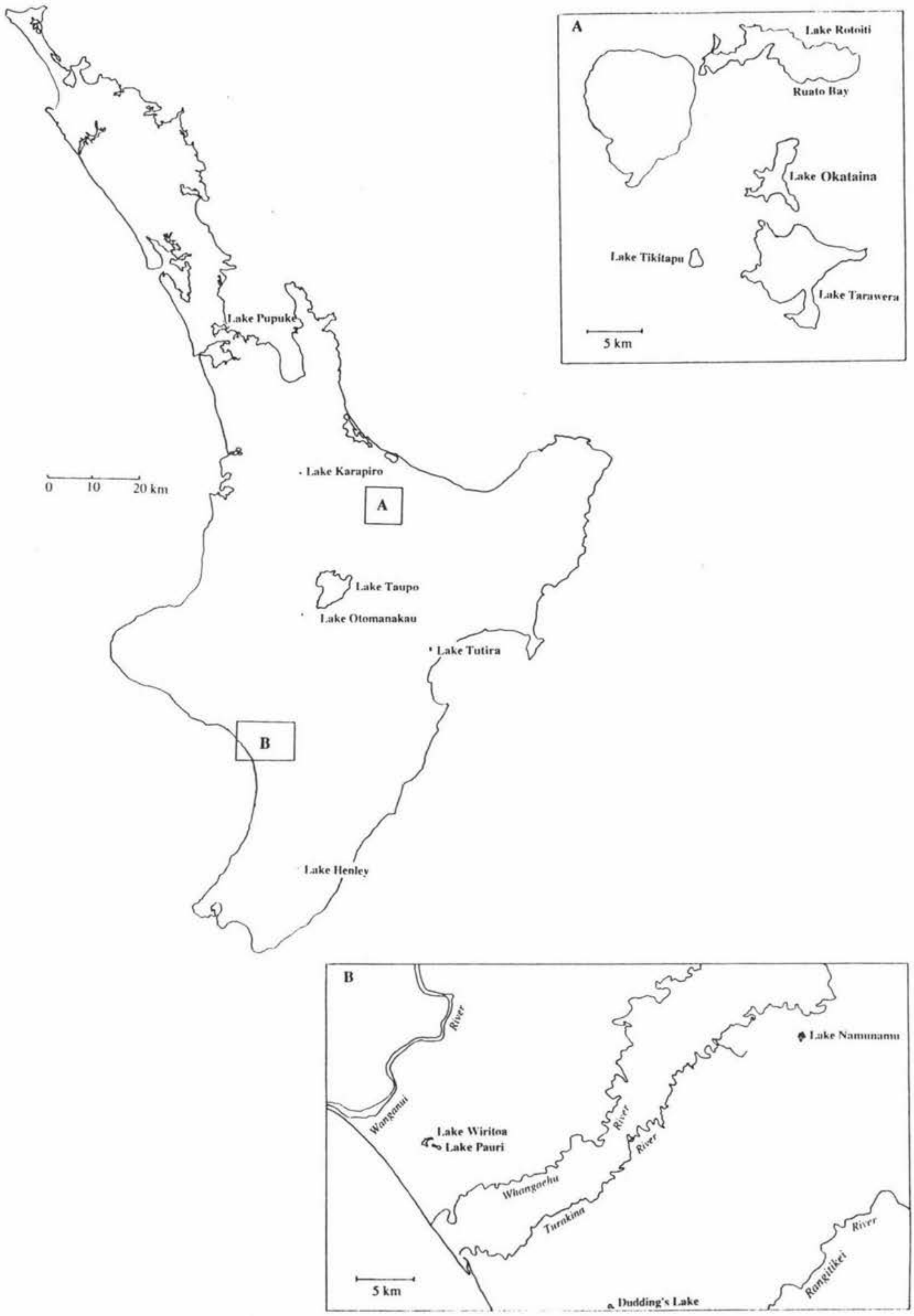


Fig. 3.1. Map of the North Island showing lakes included in the survey of phytomacrofaunal communities.

Physical measurements

At the time of sample collection measurements of conductivity, temperature (Orion model 122 conductivity meter), pH (Solstat pH/Mv meter) and dissolved oxygen (YSI model 58 dissolved oxygen meter), were made at each site. Water samples were collected in acid washed polythene bottles, filtered through Watman 47 mm GFC filters (1.2 µm), and frozen prior to analysis. Dissolved reactive phosphorus (DRP) was determined using molybdenum blue and ascorbic acid reduction followed by an automated colorimetric finish (Downs 1978a), and nitrate nitrogen (NO_3-N) was determined by an automated hydrazine reduction, diazotization with sulphanilamide, and an automated colorimetric finish. (Downs 1978b). Chlorophyll-*a* was determined using the method of Pridmore *et al* 1983 by acetone extraction of crushed GFC filters and spectrophotometer measurements.

Determination of Lake Trophic Status

Determining the trophic status of a lake is complicated because it cannot be readily assessed by reference to a single criterion (White 1983). Furthermore, different authors use different criteria, not all of which are necessarily correlated, so lakes can be classified as oligotrophic by one method and eutrophic by another (Henderson-Sellers & Markland 1987). For example, the Wanganui lakes selected for this survey (Dudding's, Pauri, and Wiritoa) have been classed as oligotrophic by measures of nutrient and chlorophyll-*a* concentrations (Vant 1982), but eutrophic through use of macrophytes (Kelly 1978). Criteria suggested for use in determining trophic status of a water body by the O.E.C.D. (1982) include nutrient concentrations, hypolimnetic oxygen depletion rate (in stratified lakes), secchi disc depth, and primary productivity (assessed by measuring carbon assimilation, chlorophyll-*a* concentration, or algal densities). While the relationship between the various measures has been examined (e.g., Dillon & Rigler 1974; O.E.C.D. 1982; Pridmore & Hewitt 1984), the use of a single trophic status index to encompass all such measures has been limited (Carlson 1977).

To assess trophic status of the lakes in this survey a multivariate approach was adopted because the above methods did not yield a consistent trophic ranking for all of the lakes. Dissolved reactive phosphorus, nitrate nitrogen, secchi disk depth, and chlorophyll-*a* measurements were combined in a principal components analysis (PCA). Mean values for each of these measures were obtained, where possible, from long term studies recorded in the literature (Livingston *et al* 1986 and references therein), and/or from unpublished data (N. Burns unpublished data; S. Pilkington *pers comm*; and data from Chapter One) (Table 1) . Data for the 17 lake sites were normalised prior to performing the PCA procedure using the multivariate statistics package PATN (Belbin 1993).

Invertebrate Community Structure

Species richness at the study sites was assessed using the number of taxa, Margalef's index and rarefaction. Species number is the simplest measure of species richness but is dependant on the number of individuals in each sample. Margalef's index (Cliford & Stephenson 1975), calculated as

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

(where S is the number of taxa, and N is the total abundance), attempts to account for differing numbers of individuals in samples but presupposes a functional relationship between species number and abundance (Ludwig & Reynolds 1988). Rarefaction does not rely on such a relationship but still accounts for differences in species richness associated with the total number of individuals (Hurlbert 1971). The expected number of species in a sample of *n* individuals, was determined with Hurlbert's (1971) rarefaction equation

$$E(S_n) = \sum_{i=1}^{s} \{1 - [\binom{N-n_i}{n} / \binom{N}{n}]\}$$

(where n_i is the number of individuals in the ith species) This was performed using a modified version of the RAREFAC.BAS computer programme of Ludwig and Reynolds (1988).

The other component of diversity examined was equability or evenness which assesses how individuals are distributed amongst species. Again two indices were used to assess slightly different components of this measure, the Berger-Parker index (Berger & Parker 1970) which measures dominance of the commonest animal, and Simpson's index (Simpson 1949) which is weighted more evenly. The Berger-Parker index is given by:

$$D_E = \frac{n_{\max}}{N}$$

(where n_{max} is the number of individuals of the most common species, and N = total number of individuals), and Simpson's index by:

$$\hat{\lambda} = \sum_{i=1}^{S} \frac{n_i(n_i - 1)}{N(N-1)}$$

(where n_i = the number of individuals in the ith species, and N = total number of individuals). The two evenness indices were expressed as reciprocals so that increasing index values indicate increasing diversity.

Cluster analysis was performed on log transformed species abundance data and presence/absence data using the Bray Curtis distance measure (Beales 1984) and the group average clustering algorithm of the PATN multivariate statistics package (Belbin 1993). Detrended correspondence analysis (DECORANA) was also conducted, again on log transformed data, using the DCR procedure of PATN (Belbin 1993).

Nested analysis of variance were performed using SAS (1988) to examine differences in community characteristics between the lake trophic groups, and lake groups produced from the cluster analysis. Pearson (r) correlations were performed on $\log_{10}(x+1)$ transformed data using the CORR procedure of SAS (1988) to examine links between the physical and biological variables associated with the samples. Spearman rank correlations (r_s) were performed to determine which species and/or environmental variables were associated with the three main DECORANA axes.

RESULTS

Determination of trophic status

Mean secchi disk measurements, DRP, and NO_3-N were all strongly correlated with Axis 1 of the PCA ($r_s = -0.75$ $P < 0.01$, $r_s = 0.86$ $P < 0.01$, and $r_s = 0.60$ $P = 0.01$, respectively). Chlorophyll-*a*, the other important indicator of trophic status was correlated with Axis 3 ($r_s = 0.82$ $P < 0.01$). A plot of Axis 1 against Axis 3 therefore incorporated all measures of trophic status and yielded four "trophic" status groups (Fig. 3.2). These corresponded approximately to traditional trophic status labels of oligotrophic, meso-oligotrophic, meso-eutrophic and eutrophic. Although strictly speaking the term "eutrophic" is used to infer phytoplankton dominated lakes which do not support submerged macrophytes, the term will be used in this paper to indicate the most nutrient enriched of the lakes included in the survey. The oligotrophic group was characterised by low nutrient levels, high water clarity, and low average chlorophyll-*a* (Lakes Tarawera, Tikitapu, Okataina, and Taupo), the meso-oligotrophic group had higher levels of chlorophyll-*a*, low nutrients and variable water clarity (Rotoiti (both sites), Pupuke, and Tutira), the meso-eutrophic group was characterised by low chlorophyll-*a* values yet high nutrient concentrations, (Otomanakau, Namunamu, and Lake Henley), and the eutrophic group had high levels of chlorophyll-*a*, high nutrient levels, and low water clarity, (Lakes Karapiro, Wiritoa, Dudding's, and Pauri).

Diversity

A total of 70 invertebrate taxa were identified from the 10 species of macrophytes collected at the 17 lake sites. Each lake had between 20 (Dudding's Lake) and 35 (Lake Taupo) species, with an overall average of 26 species per lake (Fig. 3.3). Taxa that occurred at all 17 lake sites were: platyhelminthes spp., *Potamopyrgus antipodarum*, oligochaetes, nematodes, and cycloid copepods, while 16 taxa were recorded from only one of the lakes sampled.

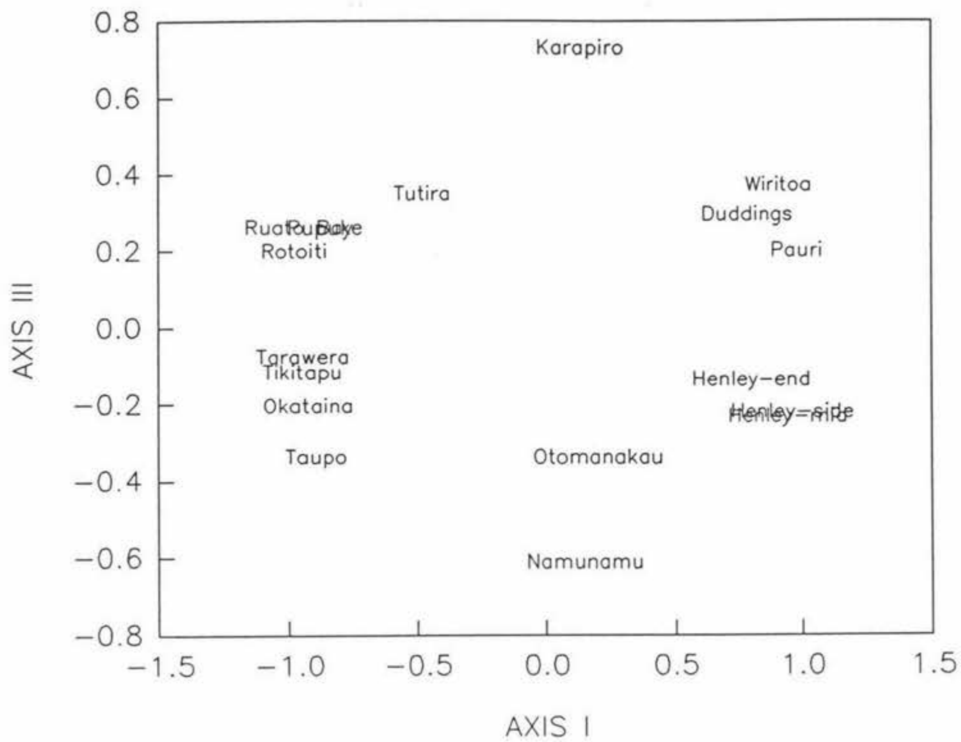


Fig. 3.2. Axis I of a Principal Components Analysis as a function of Axis III for mean physical variables of the 17 lake sites.

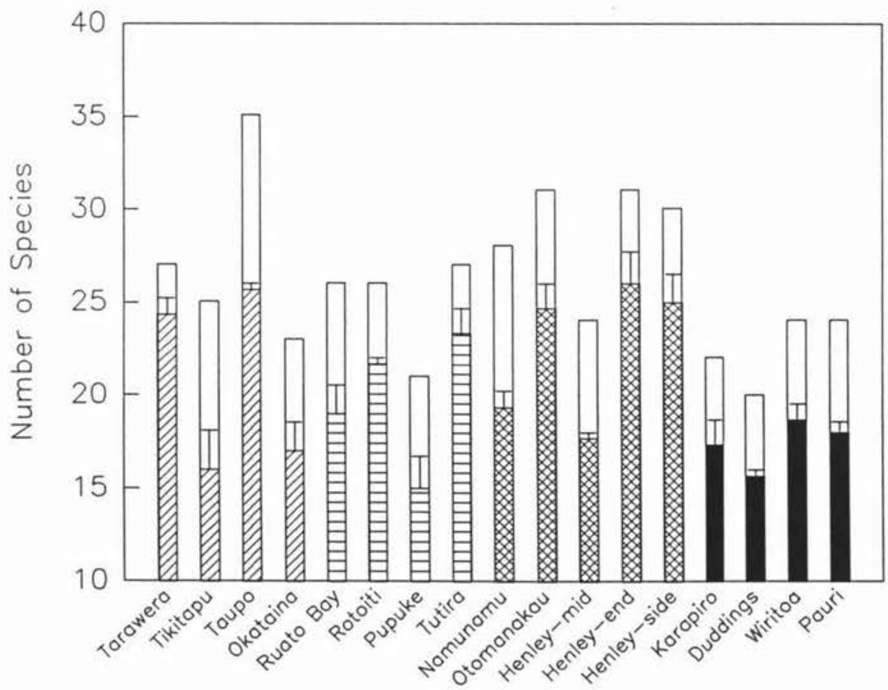


Fig. 3.3. Total number of species recorded per lake site ( ), together with the average number of taxa (± 1SE) from 3 replicate samples collected at the 17 lake sites in May 1994.

( ) Oligotrophic group ( ) Meso-oligotrophic group

( ) Meso-eutrophic group ( ) Eutrophic group

Overall the oligotrophic, and meso-eutrophic trophic groups had a greater number of taxa than the meso-oligotrophic and eutrophic groups ($F_{3,34} = 10.74$ $P < 0.01$). Margalef's index and rarefaction measures indicated the eutrophic group had a significantly lower taxa richness than the other three trophic status groups which were roughly similar ($F_{3,34} = 24.82$ $P < 0.001$, and $F_{3,34} = 22.76$ $P < 0.001$ for Margalef's index and rarefaction, respectively) (Fig. 3.4). Both insect and mollusc taxa in particular were relatively rare in this group compared with the other three groups ($F_{3,34} = 21.13$ $P < 0.001$, and $F_{3,34} = 14.16$ $P < 0.001$, respectively).

The eutrophic group had a significantly greater total abundance than the other lake groups ($F_{3,34} = 13.56$ $P < 0.001$) which did not differ from each other. Numbers of individuals per gram of macrophyte were also greatest in this group and lowest in the meso-eutrophic and meso-oligotrophic groups ($F_{3,34} = 38.63$ $P < 0.001$) (Fig. 3.5).

Lake Karapiro was the site most heavily dominated by a single taxon where Amphipod sp. were at densities of 600 individuals per gram of dried macrophyte. Overall, the meso-oligotrophic group had the least dominated communities with the highest Berger-Parker measures ($F_{3,34} = 11.95$ $P < 0.001$) (Fig. 3.6A). Lakes Namunamu, and Pupuke had the most evenly distributed communities using Simpson's index, however, although there were significant differences across all the trophic groups ($F_{3,34} = 6.56$ $P = 0.001$) (Fig. 3.6B) no one trophic group was distinctly different from all other groups.

The dominant taxa varied between lakes sites. Seven lakes were dominated by *Potamopyrgus antipodarum*, five were dominated by oligochaetes and the remaining were dominated by either *Paroxyethira* sp.(caseless instars), Amphipod sp., or *Physastra variabilis.* The most widespread sub-dominant taxa were oligochaetes and platyhelminthes (occurring as sub-dominants in five lakes each), with other sub-dominant taxa including *Potamopyrgus antipodarum*, *Paroxyethira hendersoni*, *Hydra*, cyclopoid copepods and amphipods. The mean relative abundances of the five most abundant species at each study site are listed in Table 3.

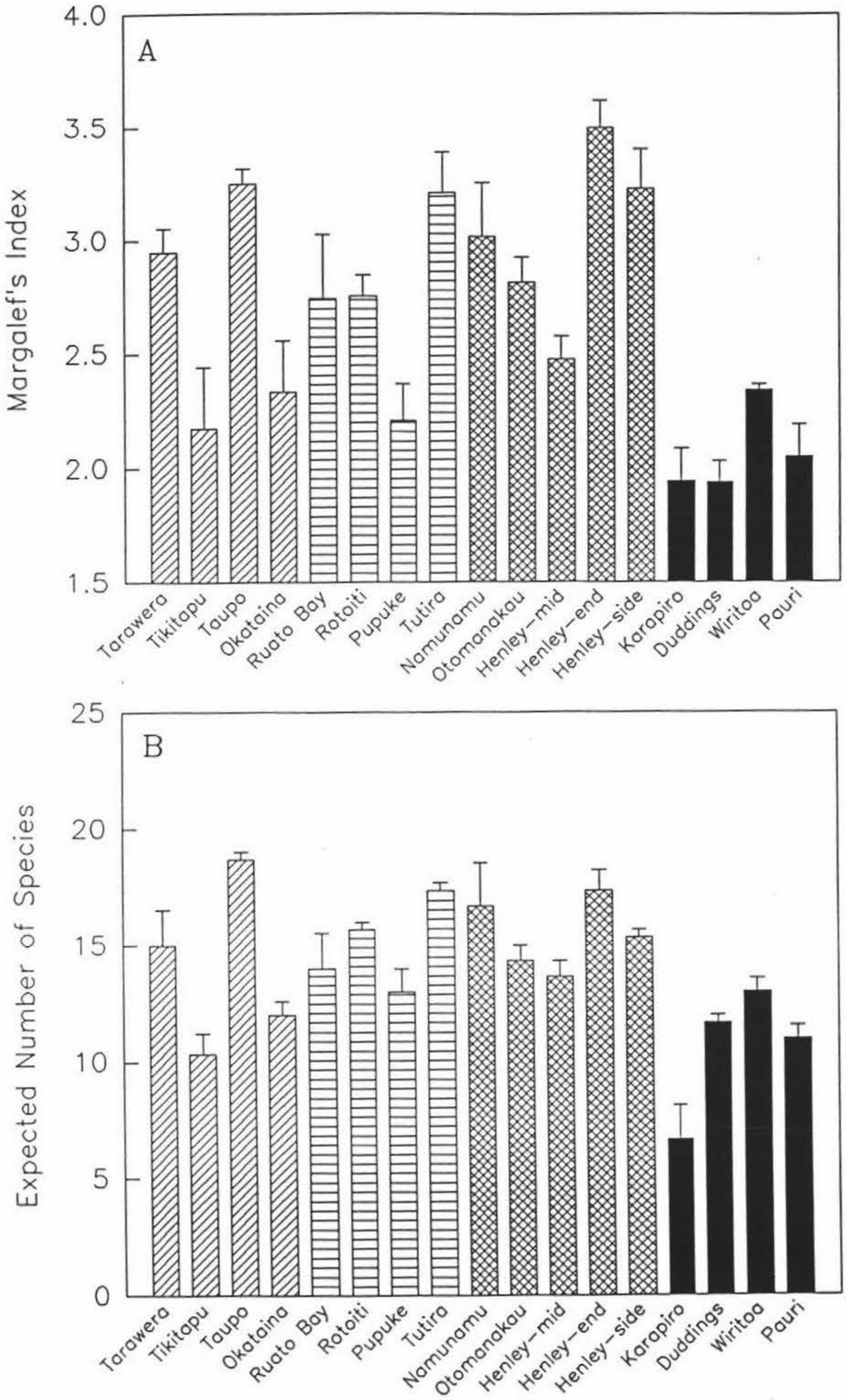


Fig. 3.4. Mean species richness (± 1SE) from three replicate macrophyte samples collected at each lake site in May 1994. **A**. Margalef's index **B**. Expected number of species.

( ) Oligotrophic group ( ) Meso-oligotrophic group

( ) Meso-eutrophic group ( ) Eutrophic group

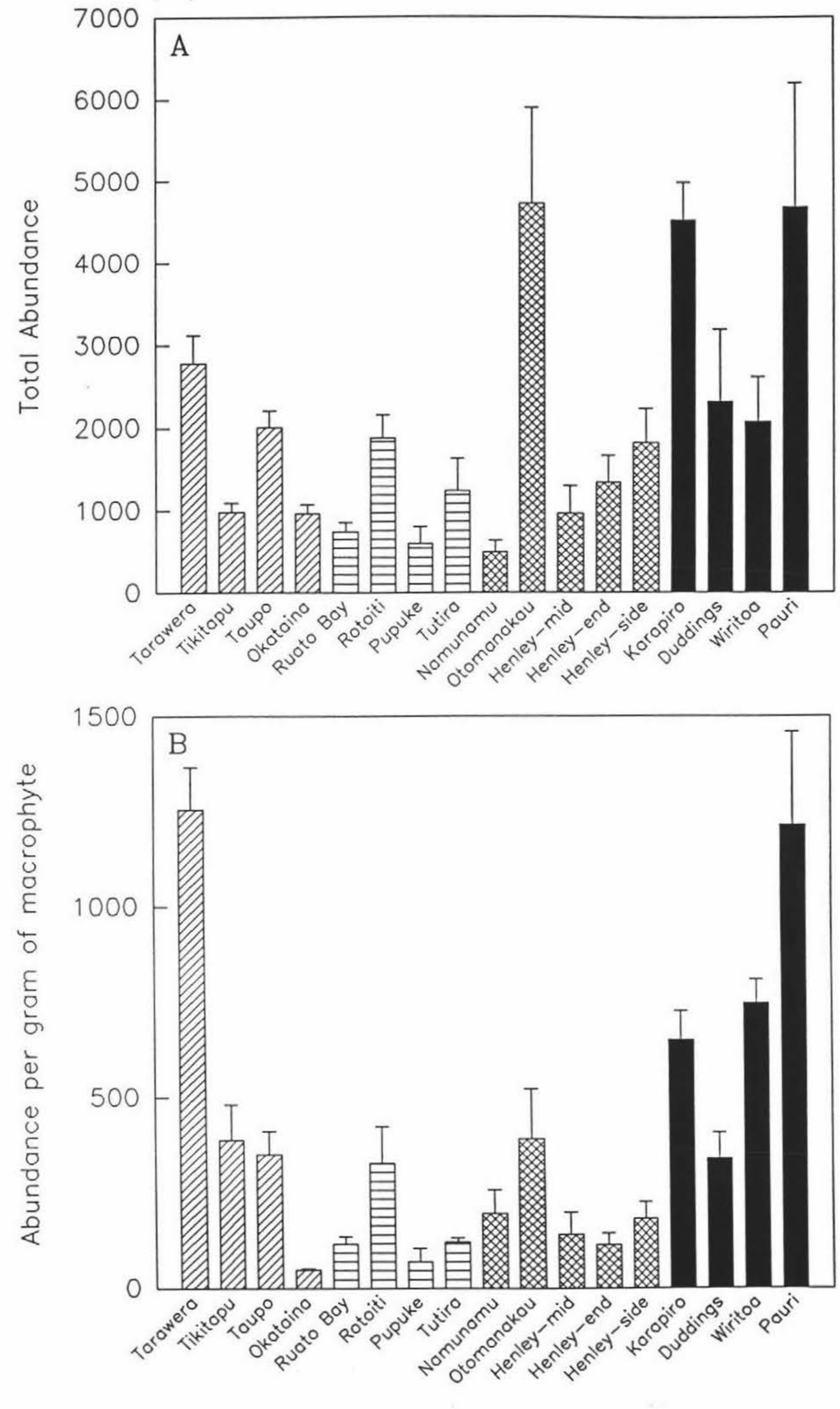


Fig. 3.5. **A.** Mean total abundance measures (± 1SE) and **B.** Total number of individuals per gram of dry macrophyte (± 1SE) from 3 replicate macrophyte samples collected at each lake site in May 1994

( ) Oligotrophic group ( ) Meso-oligotrophic group

( ) Meso-eutrophic group ( ) Eutrophic group

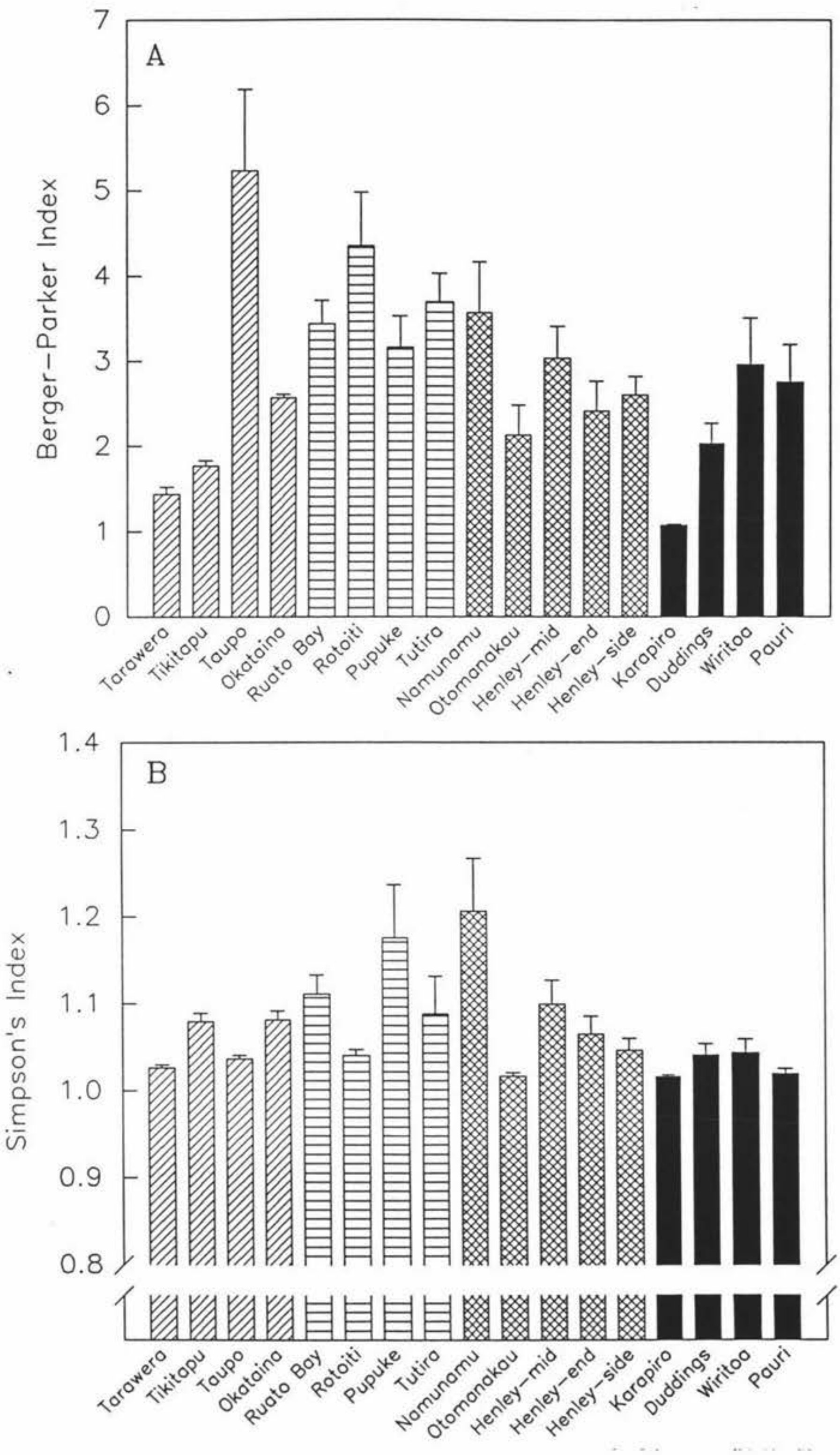


Fig. 3.6. Mean species evenness (± 1SE) from 3 replicate macrophyte samples collected at each lake site in May 1994. **A**. Berger-Parker index **B**. Simpson's index.

( ) Oligotrophic group ( ) Meso-oligotrophic group

( ) Meso-eutrophic group ( ) Eutrophic group

Table 3. Mean percentage of total abundance of the five most abundant species at each of the study sites. * indicates the species was one of the ten most abundant species at that site, and + indicates the presence of the species at that site. Tarawera (T), Tikitapu (TT), Okataina (O), Ruato Bay (RB), Rotoiti (R), Pupuke (PP), Tutira (TA), Karapiro (K), Duddings (D), Wiritoa (W), Pauri (P), Namunamu (N), Otomanakau (OT), Henley- mid (HM), end (HE) and side (HS).

TAXA	T	TT	TP	O	RB	R	PP	TA	K	D	W	P	N	OT	HM	HE	HS
Hydra	+		17.16	12.56	20.88	*	+	+	*	+	*	*	*	+	+	+	+
Brazoan	71.16	7.25	*	+													
Platyhelminthe sp. B	+	5.96							+					*	+	+	+
Platyhelminthes spp.	3.56	10.66	11.34	25.65	24.96	13.22	*	9.65	2.73	6.83	10.92	15.99	13.26	+	*	+	+
Potomopergus antipodium	6.41	+	8.89	39.09	30.02	+	*	17.59	+	52.67	+	36.99	+	44.81	20.4	40.87	37.53
Physa sp	*		+		*	*	+	+		+	+	+	*	2.75	16.27	11.47	15.21
Physa variabilis								28.18									
Gyralus corinna			+	5.26	+	+	6.38	+			+	+		+	14.36	4.31	4.54
Spaerium novaezelandie		+			+					+	+	+	3.78		+	+	
Ferrisia neozelanica	3.59		+		10.75	+											
Hirudinea sp. A			+					+	0.85	+	*	*	*				
Oligochaete sp. A	*	56.62	*	*	3.14	24.34	8.27	5.34	+	13.67	38.18	10.26	12.99	27.14	29.44	17.5	17.1
Lumbriculus variegatus	+	8.27	+	+	+	*	*	+	0.51	*	9.27	18.27	22.67	8.18	7.23	7.86	7.77
Chaetogaster sp.		+	+	6.13		+		+		+	*	+		+	+	+	*
Nematoda	+	+	+	+	*	13.2	+	+	*	*	+	*	7.51	+	+	*	*
Acarina sp. A	+	*	+		*	*	+	*	*	5.8	*	*	+	+	+	+	+
Amphipoda sp.			+						93.26		13.17	+		+	+	+	+
Bosminidae	+		*	+		+	+		0.35	7.49	+	+		+	*	*	*
Cyclopidea	+	*	+	*	+	12.9	*	*	+	*	9.9	8.33	17.73	+	*	*	*
Tanais stanfordi	4.02																
Paroxyethira hendersoni	+		*	+	+	21.93	+	+				+	+				
Paroxyethira sp.	*	+	19.1	*	*	*	31.15	*	+		+	+	*	+	+	*	*
Tanytarsus vesperatinus	+				+	+	13.64			+				+			
Orthocladinae sp. A			12.59	+				19.9	*		+		+	+		+	+
Orthocladinae spp.	*					*	8.66		*								

There were also significant differences between lakes and lake trophic groups in the relative abundances of higher level taxonomic groups (Fig. 3.7). The meso-eutrophic group had a significantly greater abundance of molluscs ($F_{3,34}$ = 21.12 P < 0.01) than any of the other three trophic status groups. The oligotrophic group had a higher proportion of Coelenterata ($F_{3,34}$ = 40.38 P < 0.01), and the eutrophic group had a higher abundance of Crustacea ($F_{3,34}$ = 75.81 P < 0.01). Acarina abundance was greatest in the meso-oligotrophic and eutrophic groups ($F_{3,34}$ = 15.82 P < 0.01) and insect abundance was greatest in the meso-oligotrophic group and lowest in the eutrophic group ($F_{3,34}$ = 146.79 P < 0.001). There was no significant difference in the proportions of annelids between the trophic groups ($F_{3,34}$ = 0.6 P = 0.62).

Community composition

Cluster analysis of both presence/absence and log transformed abundance data yielded three distinct groups with Lake Tikitapu, an oligotrophic lake, clustered on its own (Fig. 3.8). Communities from all three Lake Henley sites and Lake Otomanakau, both meso-eutrophic lakes with high flushing rates, clustered together (Group A). All the members of the eutrophic group formed the second cluster group (Group B), with the addition of meso-eutrophic Lake Namunamu, and meso-oligotrophic Lake Rotoiti (in the abundance data only). The third group (Group C) consisted of all of the meso-oligotrophic lakes (except Lake Rotoiti in the abundance data), and all the oligotrophic lakes except Lake Tikitapu.

Mean relative abundances for the most abundant taxa in each of the classification categories are presented in Table 4. Group A, Lake Henley and Lake Otomanakau, had a significantly higher proportion than the other cluster groups of *Potamopyrgus antipodarum*, *Physa* sp., *Gyraulus corinna*, *Antiporus* sp., and *Chironomus zealandicus*. Group B (Lakes Karapiro, Pauri, Wiritoa, Namunamu, Rotoiti and Dudding's Lake), had a greater abundance of Nematoda, and Amphipod sp. than the other lake groups. Group C lakes (Pupuke, Tutira, Taupo, Okataina, Ruato Bay and Tarawera) were characterised by the lowest abundance of oligochaetes, and the highest abundance of several insect species (*Hygraula nitens*, *Paroxyethira hendersoni*, *Paroxyethira* (caseless), *Tanytarsus* spp., and

Orthocladinae sp. A) and two mollusc species (*Physastra variabilis*, and *Ferrisia neozelanica*). Lake Tikitapu differed from the other lake groups in having a greater relative abundance of Platyhelminthes sp. A, *Hemicordulia australiae*, *Tanytarsus funebris*, and two morphospecies of oligochaetes and a significantly lower abundance of *Potamopyrgus antipodarum*.

Table 4. Significantly different mean relative species abundances (P <0.001*, or P<0.05) between lake cluster groups.

	Group A	Group B	Group C	Group D	$F_{3,34}$
Hydra	0.29	8.74	2.38	0.00	14.50*
Brazoan	0.00	12.37	0.00	7.25	36.31*
Platyhelminthes Sp. A	0.92	0.01	0.00	5.96	7.60
Platyhelminthes spp.	0.65	13.29	10.49	10.66	50.24*
Potamopyrgus antipodarum	35.90	17.33	15.19	0.03	160.08*
Physa sp	11.42	1.18	0.76	0.00	62.79*
Physasta variabilis	0.00	4.70	0.00	0.00	255.6*
Gyralus corinna	5.92	2.25	0.02	0.00	60.23*
Sphaeriidae	0.97	0.03	0.03	0.03	10.19*
Ferrisia neozelanica	0.00	2.46	0.02	0.00	92.68*
Hirudinea sp. A	0.57	0.10	0.01	0.03	9.16*
Hirudinea sp. B	0.00	0.09	1.65	0.00	10.30*
Oligochaete sp.A	22.80	4.72	16.61	56.62	15.20*
Oligochaete sp. B	7.76	0.84	9.53	8.27	11.93*
Oligochaete sp. C	0.31	0.00	0.00	2.08	38.76*
Nematoda	1.29	0.63	4.06	0.24	14.94*
Mite sp. A	0.42	0.65	2.08	2.79	34.06*
Amphipod sp.	0.24	0.17	17.79	0.00	215.14*
Cloderacean sp. A	2.00	0.92	1.63	0.00	15.10*
Calanoid copepod	0.00	0.00	0.08	0.00	7.38*
Hygraula nitens	0.10	1.27	0.05	0.00	27.86*
Hemicordulia australiae	0.00	0.11	0.00	0.72	130.02*
Paroxyethira sp.	0.52	0.30	0.02	0.00	8.38*
Paroxyethira hendersoni	0.01	4.46	0.10	0.00	78.09*
Paroxyethira tillyardi	0.00	0.26	0.00	0.00	10.99*
Paroxyethira (caseless instars)	0.98	9.59	0.88	0.03	42.16*
Antiporus sp.	0.10	0.00	0.00	0.00	39.72*
Chironomus zealandicus	0.66	0.01	0.04	0.00	18.21*
Tanytarsus funebris	0.00	0.29	0.00	0.31	112.50*
Tanytarsus spp.	0.00	0.43	0.00	0.00	40.79*
Orthocladinae sp. A	0.03	5.42	0.09	0.00	200.56*
Orthocladinae spp.	0.00	1.63	1.01	0.00	46.46*

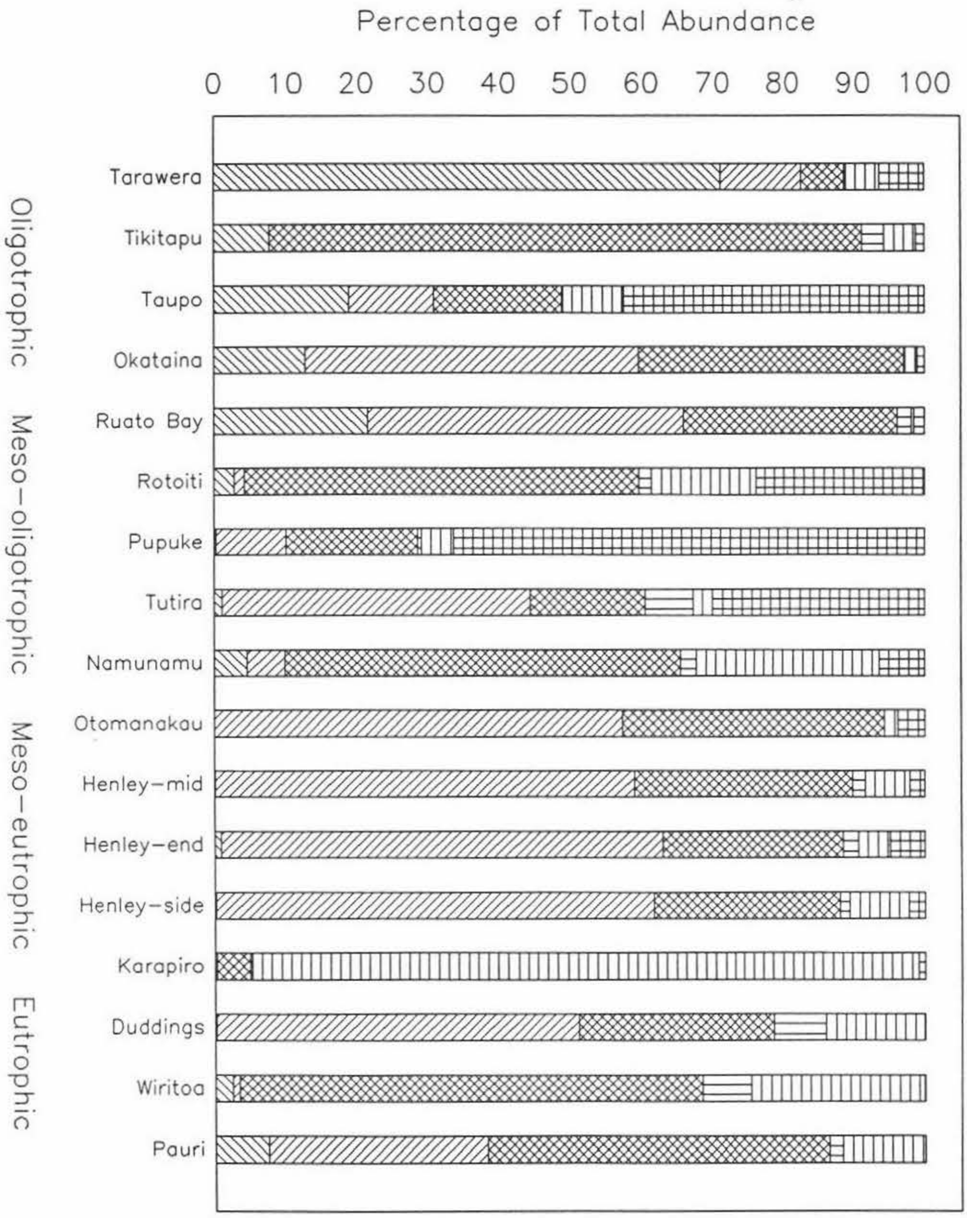


Fig. 3.7. The mean relative abundance of each major taxonomic group of invertebrates collected from each lake site in May 1994.

( ) Coelenterata ( ) Mollusca ( ) Annelida

( ) Acarina ( ) Crustacea ( ) Insecta.

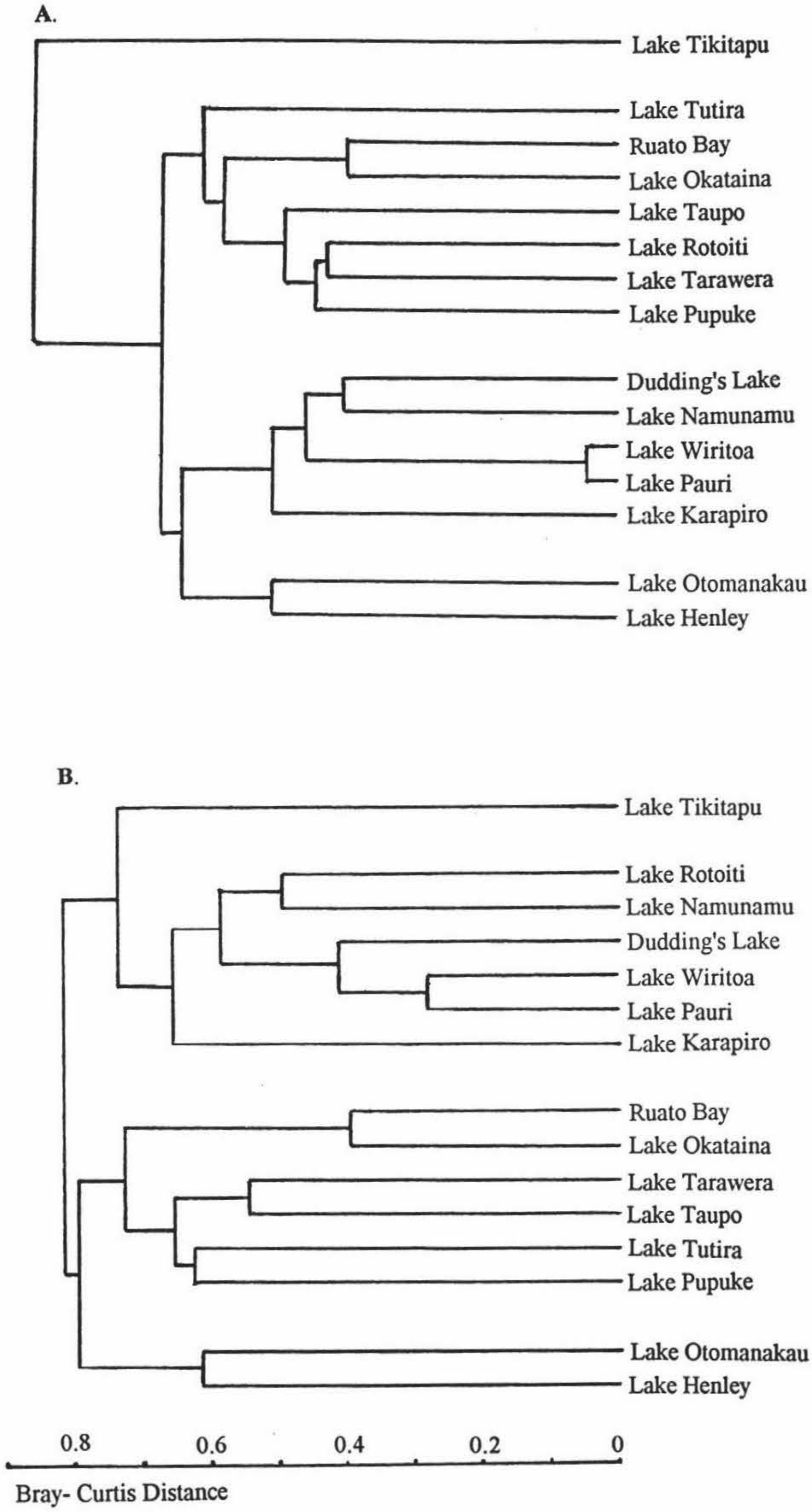
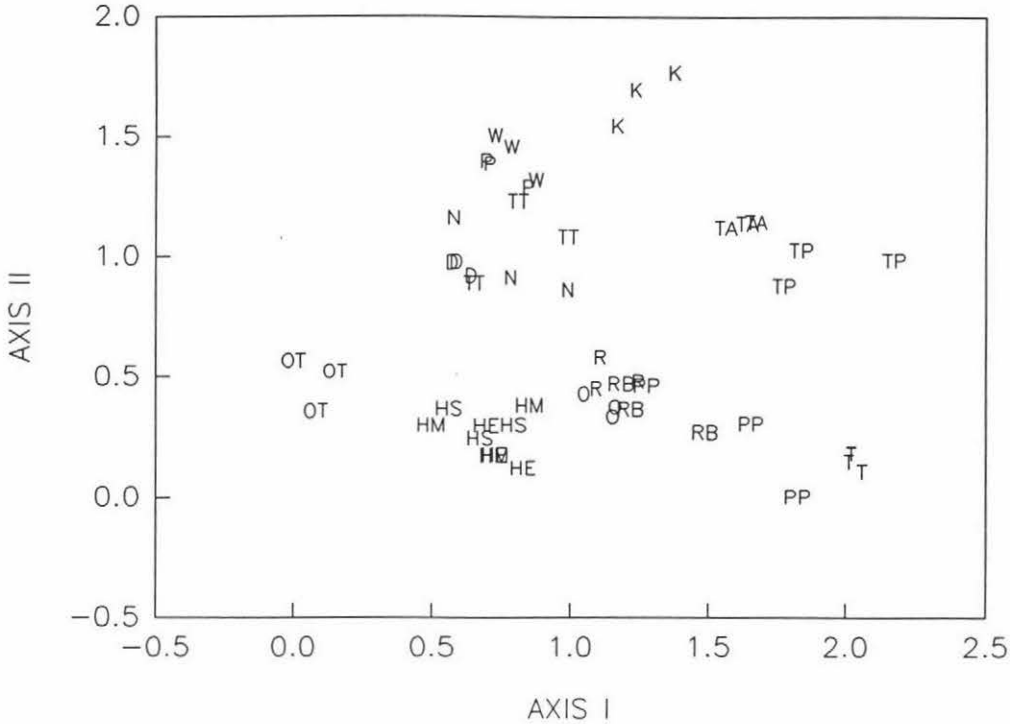


Fig. 3.8. Cluster analysis of the invertebrate communities collected from 17 lake sites in May 1994 using **A**. presence/absence data and **B**. log transformed abundance data.

The ordination analysis produced a similar pattern with three broad groups (Fig. 3.9). Although there was a considerable degree of intercorrelation between physical variables (Table 5) the invertebrate communities appear to separate along Axis 1 of the DECORANA according to the size, clarity and nutrient status of the lake. Mean depth, maximum depth, lake surface area, percentage of forested catchment, and mean, maximum and minimum secchi disk depths were all positively correlated with Axis 1 and dissolved reactive phosphorus, nitrate nitrogen, and percentage of catchment in pasture were negatively correlated with this axis (Table 5). Axis 2 appeared to separate communities on the basis of lake phytoplankton levels (as indicated by mean chlorophyll-*a* values) although this axis was also positively correlated with the amount of lake catchment in pasture, dissolved reactive phosphorus and negatively correlated with the amount of urban development in the catchment (Table 5).

Communities to the left of Axis 1 were those from lakes of the meso-eutrophic and eutrophic groups and were characterised by high abundances of Annelida and Acarina. Communities to the right of Axis 1 were those from the meso-oligotrophic and all but one oligotrophic lake (Tikitapu) and were characterised by a high abundance of Coelenterata and a high abundance and diversity of Insecta. Axis 2 yielded the third group splitting Karapiro, Pauri, Wiritoa, Dudding's (the eutrophic lakes), Lake Namunamu and Lake Tikitapu from the rest because of the comparatively higher abundance and diversity of molluscs in the remaining meso-eutrophic lakes, and the higher abundance of insects in the communities from the meso-oligotrophic and oligotrophic lakes. Specific taxa associated with each DECORANA Axis are listed in Table 6. Axis 3 separated Lake Tikitapu from all other lakes, and the species positively associated with this Axis were characteristic of this lake.



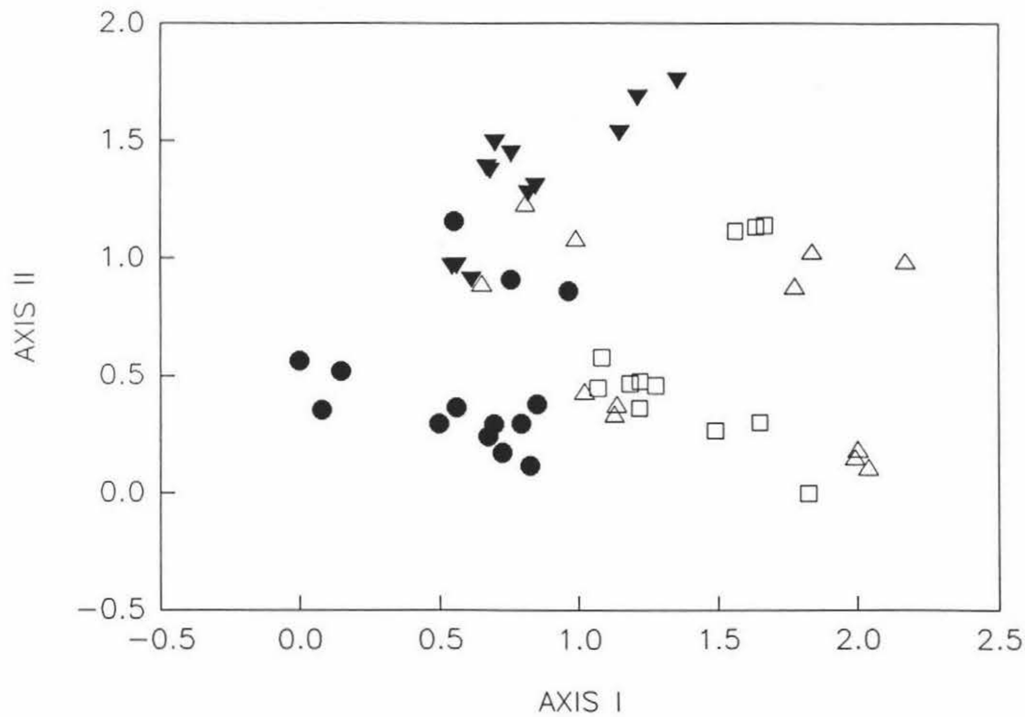


Fig. 3.9. Axis one of a detrended correspondence analysis as a function of Axis two for phytomacrofaunal communities from 17 lake sites collected in May 1994.
A. Lake Otomanakau (OT), Lake Namunamu (N), Lake Wiritoa (W), Lake Pauri (P), Dudding's Lake (D), Lake Tikitapu (TT), Lake Henley - mid site (HM), end site (HE) and side site (HS), Lake Rotoiti - North Western arm site (R), Ruato Bay site (RB), Lake Okataina (O), Lake Tutira (TA), Lake Taupo (TP), Lake Pupuke (PP), Lake Karapiro (K) and Lake Tarawera (T).

B. Oligotrophic group (△) Meso-oligotrophic group (□)

Meso-eutrophic group (●) Eutrophic group (▼).

Table 5. Correlation co-efficients between environmental variables.

	Mean depth	Max depth	Area	Altitude	MDRP	MNO3-N	MChl-a	Min seechi	Mean seechi	Max seechi	Pasture	Forest
Max depth	0.95*											
Lake area	0.83*	0.74*										
Altitude	0.24	0.23	0.25									
MDRP	-0.5*	-0.52*	-0.26	-0.05								
MNO3-N	0.48*	-0.58*	-0.21	-0.45*	0.17							
MChl-a	-0.11	-0.07	-0.08	-0.26	0.56*	-0.14						
Min seechi	0.86*	0.89*	0.78*	0.45*	-0.53*	-0.51*	-0.15					
Mean seechi	0.18	0.11	-0.03	-0.29*	-0.24	-0.16	-0.06	-0.12				
Max seechi	0.74*	0.75*	0.68*	0.61*	-0.29*	-0.48*	-0.23	0.86*	-0.06			
Pasture	-0.39*	-0.42*	-0.21	-0.45*	0.32*	0.19	0.12	-0.42*	-0.36*	-0.56*		
Forest	0.57*	0.63*	0.38*	0.74*	-0.09	-0.62*	0.12	0.70*	-0.21	0.74*	-0.52*	
Urban	-0.05	-0.15	-0.14	-0.42*	-0.11	0.28*	-0.11	-0.35*	0.86*	-0.24	-0.39*	-0.44*

Table 6. Species associated with DECORANA axes

AXIS I	AXIS II	AXIS III
Positive		
Brazoan	Platyhelminthes spp.	Oligochaete sp. C
Ferrisia neozelanica	*Glossiphonia* sp.	*Hemicordulia australiae*
Hygraula nitens	Pionidae	*Tanytarsus funebris*
Paroxyethira hendersoni	Calanoidae	Orthocladinae spp.
Paroxyethira sp.		
Orthocladinae spp.		
Negative		
Platyhelminthes sp. A	Platyhelminthes sp. A	Amphipod sp.
Sphaerium novaezelandiae	*Physa* sp.	Daphniidae
Lumbriculus variegatus	*Gyraulus corinna*	*Tanytarsus* spp.
Nematoda	*Paroxyethira* sp.	
Orbatidae	*Antiporus* sp.	
Chironomus zealandicus		

DISCUSSION

There appear to be three main groups of phytomacrofaunal communities that correspond approximately to lakes grouped on the basis of their trophic status. Invertebrate communities from the more enriched lakes (i.e., the eutrophic group) were characterised by high densities, low species richness and a high degree of numerical dominance by a few taxa. The communities with the greatest number of species were from the oligotrophic and meso-eutrophic lakes whereas the meso-oligotrophic communities were those with individuals distributed most evenly amongst the species. The communities from the eutrophic lakes were numerically dominated by amphipods, *Potamopyrgus antipodarum*, annelids and Acarina. Communities from the meso-eutrophic group were characterised by a higher abundance of molluscs, while communities from the oligotrophic and

Although in a survey of lakes to examine the influence of trophic status it is difficult to factor out all the confounding variables (i.e., lakes are not easily replicated) - for example it is hard to avoid the fact that large lakes tend to be oligotrophic simply because of size, it does appear that increasing trophic status leads to communities with a high dominance of one or two taxa. Nutrient enrichment has been found in a number of other studies in both terrestrial (eg. Bakelaar & Odum 1978) and aquatic communities (eg. Cyr & Downing 1988; Dougherty & Morgan 1991; Forsyth 1978; Forsyth & McCallum 1981; Johnson 1974; Lalonde & Downing 1992; Saether 1980; Ravera 1980) to increase productivity, biomass and dominance.

Changes in benthic community composition with nutrient enrichment has been attributed to a loss of species limited to oligotrophic systems and an increase in dominance of species with a wider tolerance of fluctuating environmental conditions (Saether 1990; Wiederholm 1984) such as the increased variation in dissolved oxygen levels likely to be experienced in a more eutrophic lake. In this study, a shift in the dominant species from Trichoptera or Coelenterata in nutrient poor lakes to amphipods and oligochaetes in the more enriched lakes could be attributed to such a shift. Similar observations have been recorded in Lake Constance with increasing eutrophication over a 20 year period with a decrease in burrowing mayflies, and the replacement of chironomids with oligochaetes as the dominant species (Raevell & Frenzel 1981).

However, increasing dominance by a few key taxa with a consequent decline in species richness may also reflect biotic interactions if increased productivity results in a greater quantity of a single resource rather than an increased variety of resources. Competitive dominants may utilise an increasingly greater amount of the resource accelerating the rate of any potential competitive exclusion (Huston 1979; Rosenzweig 1971). Thus, the lower species richness and the greater numerical dominance of the phytomacrofaunal communities in the more nutrient enriched lakes in this survey may reflect both changes in tolerance levels by specific invertebrates and/or biotic interactions.

While every attempt to sample 'Elodea' type macrophytes was made, the Characean macrophytes sampled from Lake Tikitapu were lower growing in a greater depth of silt than the other sampling sites from oligotrophic lakes. This may account for the close association of Lake Tikitapu communities with lakes from the eutrophic group in the DECORANA ordination, and the higher abundance of oligochaetes recorded from this lake. The difference in macrophyte type and architecture could also account for the Lake Tikitapu community constantly falling as an outlier from the other lakes in the cluster analysis, although other unmeasured physico-chemical characteristics could also be important such as the low levels of bicarbonate and calcium irons in Lake Tikitapu implicated in other studies as a reason for low mollusc abundance (Forsyth 1978). Phytomacrofaunal communities in Lakes Henley and Otomanakau group separately from the other lakes with high nutrient concentrations in both the classification and ordination analysis. In the case of these meso-eutrophic lakes the low chlorophyll-*a* levels, despite relatively high nutrient concentrations, are possibly the result of the rapid flow through of water in these lakes hindering any build up of phytoplankton levels. The distribution of the communities from the remaining group of lakes, i.e., those from the meso-oligotrophic and oligotrophic lakes, appear to be further split along Axis 2 into those from the Rotorua region and those from elsewhere.

Macroinvertebrate communities associated with the benthos of streams and rivers have a long history of being used to assess water quality (as reviewed in Metcalfe 1989; Stark 1994) as many invertebrates are differentially sensitive to various types of pollutants, and respond to them quickly allowing subtle changes in water quality to be detected. Changing trophic status of lakes, particularly with eutrophication, has been assessed with benthic communities by examining differences in total abundance, abundance of specific indicator species, and the relative abundances of oligochaetes and chironomids (Saether 1980). Application of such approaches in New Zealand lakes has however been limited. Forsyth (1978) found that total biomass was significantly greater in lakes of higher nutrient concentration, although Timms (1982) concluded that species composition and dominance were not related to lake trophic status and could find no clear evidence of trophic status indicator organisms.

The large number of taxa common to the phytomacrofaunal communities of all lakes indicates that use of differences in their relative abundances are more likely to be important than differences in the presence or absence of rare taxa. Furthermore, the generally low species richness of phytomacrofaunal communities makes attempts to use specific indicator species difficult. However, there were differences in the species associations of the three major trophic status groups which could be of assistance in the biomonitoring of changes occurring with lake enrichment. In particular *Hygraula nitens*, *Paroxyethira hendersonii*, *Paroxyethira* (caseless instars), *Tanytarsus* spp., Orthocladinae spp, and *Ferrisia neozelanica* were strongly associated with communities from the meso-oligotrophic and oligotrophic group. *Potamopyrgus antipodarum*, *Physa* sp., *Gyraulus corinna*, Platyhelminthes sp. A, *Antiporus* sp., and *Chironomus zealandicus* were associated with the meso-eutrophic lakes with high flushing rates, and Nematoda, *Glossiphonia* sp. and Amphipod sp. were associated with the eutrophic lakes.

In conclusion there were three clear groups of invertebrate communities collected from the shallow littoral zone of these 14 North Island lakes which corresponded approximately to lake trophic status. In addition to differences in species richness and dominance, the communities were characterised by specific species associations which could be useful in biomonitoring, however the generally low species richness in these communities is likely to make this difficult.

Chapter 4: Management recommendations for Lake Henley, Masterton, based on current ecological conditions.

OVERVIEW:

EXECUTIVE SUMMARY

1. Lake Henley is a shallow artificial lake in Masterton with a uniform depth, a rapid flow through of water and dense beds of macrophytes (submerged aquatic plants) which affect recreational activities such as boating and canoeing.

2. The key management problems are: the control of aquatic weed beds, the maintenance of water quality in the lake and the monitoring of the quantity of water entering the lake.

3. Options for weed bed management are: chemical control, mechanical harvesting, habitat manipulation and biological control methods. The latter is currently being implemented.

4. The use of Grass Carp (*Ctenopharyngodon idella*) to control, as opposed to eradicate, aquatic weeds constitutes an experiment in management. Ongoing monitoring of the weed control programme is necessary to allow for modification of the stocking rate as appropriate.

5. Due to the high flushing rate of water through the lake, the nutrient enriched water results in the prolific growth of filamentous algae, and not in a build up of microscopic green algae (phytoplankton). The highest loading of nutrients enters the lake through the main inflow, thus attempts to reduce nutrients in this water are likely to be more effective at reducing total lake nutrient concentrations than diverting the other inflowing streams away from the lake.

6. Monitoring the amount of river diverted into the lake from the Ruamahunga river is important to ensure a sufficient flow through rate is maintained to prevent the development of unwelcome phytoplankton blooms. However, excessive water takes must be avoided to ensure the ecological health of the river is not adversely affected.

4.1 INTRODUCTION

The primary objective of this chapter is to present recommendations for the management of Lake Henley, a small artificial lake in Masterton, based on physico-chemical and biological characteristics recorded for the lake between May 1993 and August 1994. To this end characteristic physical and ecological features of the lake will be described with an identification of each of the key issues pertaining to the management of the lake. These issues will be examined in detail drawing from data gathered from the lake and from general literature on lake management to generate specific recommendations for the ongoing management of Lake Henley.

4.1.1 Description of Lake Henley

The Lake Henley Park development consists of Lake Henley, in addition to 8 ha of wetland area. A vision in the late 1960's of a lake for family recreation lead to the establishment of the Henley Trustee Deed. Through the sale of quarried material from the site, and donations from the Masterton Charitable Trust, the lake and surrounding park area were developed. In 1992 the management of the park was handed over by the Henley Trust to the Masterton District Council (MDC 1993).

Lake Henley is a small (11.14 ha) lake with a relatively rapid flow through of river water and a short residence time estimated to range between three and five days. The lake has a uniform depth of about 2 m with patches that go down to a maximum depth of at least 3 m. The lake has a 1.6 km shoreline, and a volume of 223 000 m^3. It has been in existence in its present form for about eight years, and despite the young age of the lake, a continuous bed of *Elodea canadensis* grows across the entire lake area.

The lake is supplied with water from the nearby Ruamahunga River. At an inlet situated 600m up the river from the Te Ore Ore Road bridge, water is diverted from the river through a diversion channel and enters the lake at the main inflow (Fig.1.1). The bulk of this water is subsequently returned to the Ruamahunga River via the main outflow. Water also leaves the lake through two outflow pipes situated at the southwest and northeast ends of the lake to supply wetlands before eventually percolating back to the river.

4.1.2 Recreational uses of the lake and park

The diversion of water from the Ruamahunga River into Lake Henley has created a body of water that is used for a variety of water recreational activities as well as enhancing the visual appeal of the area. The shallow artificial lake of such close proximity to the township of Masterton is an ideal location for water recreation and educational activities. Lake Henley is used extensively for canoeing, sail boarding, model boats, and water education, activities that are not possible to the same extent on other water bodies in the region such as the Ruamahunga River. It provides the area with an excellent small lake fishery (I. Buchanan *pers comm.*) being annually stocked with 1500-2000 rainbow trout yearlings by the Wellington Fish and Game Council. The lake offers young anglers, and less mobile fishers, such as the elderly and physically disabled, an easily accessible recreational facility. In addition to the opportunities offered for fishing at the lake, the activities of the Wellington Fish and Game Council enhance fishing in the Ruamahunga River indirectly as trout that escape from the lake take up residence in the river.

Historically the Wairarapa contained a large amount of wetland area (T. Harrington, DoC, *pers comm*) but in the face of development such habitat has been destroyed. The value of wetlands as resources for biological diversity is now being given greater recognition. The artificial wetlands that form part of the Lake Henley park therefore constitute an important ecological restoration project to restore and preserve biological heritage of the southern Wairarapa region. The establishment of the wetlands has also allowed Masterton bird watchers the opportunity to observe and study bird life that would not have been present in the area previously including at least 30 species of birds (T. Dennison, Ornithological Society, *pers comm*).

4.1.3 Physico-chemical and biological features of Lake Henley.

The shallow nature of the lake, combined with high light visibility and silty substrate contribute to provide a highly suitable environment for the development of dense beds of submerged macrophyte (aquatic plant). The weed beds in Lake Henley are dominated by Canadian oxygen weed (*Elodea canadensis*) and scattered stands of Curly leaved pond weed (*Potamogeton crispus*) and *Nitella* sp.. Despite high levels of nutrients available for plant growth in the water (dissolved reactive phosphorus and nitrate nitrogen) the occurrence of phytoplankton (microscopic green algae) blooms are rare. This is most likely a result of the rapid flushing of water through the lake and the utilisation of these nutrients by macrophytes and filamentous algae. Dense mats of filamentous algae are frequently associated with the submerged macrophytes and also form thick floating mats on the water surface. Growth of the macrophyte beds is inhibited by the presence of these dense filamentous algal mats which cause flucuations in the biomass of the weed beds (Chapter One).

MANAGEMENT ISSUES FOR LAKE HENLEY

The management of an artificial, recreational lake such as Lake Henley requires balancing the needs of a wide variety of lake users, which are often conflicting with the requirements of natural ecological processes. The task of managing Lake Henley is to maintain environmental conditions in the lake in such a way as to allow a range of recreational activities to take place and to maintain the visual appeal of the lake. There are essentially three important issues pertaining to the ecologically sound management of the lake: the control of the aquatic weeds, the maintenance of water quality and the monitoring of the quantity of water entering the lake.

4.2 MANAGEMENT OF AQUATIC WEED BEDS

The dominant management issue for Lake Henley is the controlling of the weed beds in such a way that their presence in the lake does not interfere with the recreational activities carried out on the lake. The dense growth of macrophyte in the lake results from the shallow nature of the lake, the high water clarity, and the nutrient rich sediment. It is not, as popularly perceived, an indication of 'pollution' or excessive nutrient enrichment (Hughes 1976).

Exotic aquatic plants in many small New Zealand lakes cause similar problems as in Lake Henley - restricting swimming, angling and boating activities (Brown 1975; Howard-Williams *et al* 1987). Deoxygenated areas at the base of dense weed beds can become uninhabitable by animals, act as sediment traps and increase siltation (Henriques 1987).

Despite the obvious nuisance value of submerged macrophytes, their presence in a lake such as Lake Henley can have numerous beneficial effects to the overall ecology. Macrophytes act as nutrient filters, absorbing nutrients that could otherwise be used by phytoplankton (Carpenter & Lodge 1986; Pokrovskaya 1984). They stabilise the sediments reducing the resuspension of inorganic silts which would reduce water clarity, increase the range of oxygen concentrations, provide a substrate for the colonisation of invertebrates (many of which serve as food for resident fish), act as a protective cover for fish and form the basis of the diet for many waterfowl species.

The objective for management of the aquatic weeds must therefore be to control the macrophyte beds at a level where their detrimental value to the recreational use of the lake is minimised, and their beneficial value to the ecological health of the system is maximised. Difficulties are likely to arise in establishing this balance as small lake systems tend to be either in a macrophyte dominated state or a phytoplankton dominated state (Blindow 1993), the latter option being the less visually appealing.

4.2.1 Aquatic weed management options

Methods to control aquatic weeds are varied and depend on the type of water system that they are growing in, the amount of labour available, and the degree of nuisance value. Weed control techniques include mechanical removal of weeds, chemical control with herbicides, habitat manipulation to make the environment less attractive to aquatic plant growth, and biological control of weeds with the herbivorous fish, Grass Carp. More recently, the importance of developing integrated aquatic weed management programmes is being recognised. These are aimed at keeping weed growth below a threshold nuisance level by integrating several weed control techniques.

(i) Chemical control of aquatic weeds

The only chemical registered for use in New Zealand water ways for the control of submerged aquatic plants is diquat, a bipyridinium herbicide (O'Conner 1993). Diquat is a water soluble herbicide with a broad spectrum activity on both floating and submerged aquatic plants. It is a contact herbicide that is rapidly absorbed by foliage, and is inactivated by soil, clay, or organic detritus particles in water, so is therefore not effective in muddy water, or when plant surfaces are covered with silt or growths of attached algae. Diquat is most effective in the early part of the growing season when plants are actively photosynthesizing. Although diquat controls a number of filamentous algae species, some have also been found to be resistant under field conditions. An alternative formulation of diquat as a viscous gel product improves the placement accuracy of the herbicide allowing for the control of deeper vegetation (Clayton & Tanner 1982). The formulation can be applied using hand held spray guns underwater, or from a special applicator which sprays a pencil thin jet onto the water surface. This breaks up into discrete droplets which sink to the weed beds and adhere to the plants, releasing the active ingredient (Clayton & Tanner 1982).

The advantages of using a herbicide to control, or manage aquatic weeds are the lower costs incurred, and the more long term control achievable than through mechanical control techniques (Clayton & Wells 1989). Diquat is non-carcinogenic, non-mutagenic, and is short lived in the environment with no evidence of accumulation by animals (Clayton & Wells 1989).

The detrimental effects of herbicides in aquatic systems include direct effects on non target aquatic plants and animals, and indirect effects resulting from the removal of macrophytes. The most sensitive group of organisms to this chemical are amphipods, crustaceans that are found associated with the macrophytes in Lake Henley. Problems can occur when the dead vegetation decomposes, with decreases in dissolved oxygen levels and a subsequent release of nutrients from the plant material. The release of nutrients by decomposing plants following chemical control of macrophytes can lead to undesirable filamentous and phytoplankton algal blooms. The loss of habitat for animals can lead to a change in fish feeding habits, or a change in the community structure of non-target organisms.

Many of the detrimental effects of controlling weeds with chemical methods can be minimised by only treating a small area of the lake at a time, and/or retaining areas of weed beds to act as a refuge for wildlife. The rapid flushing rate of Lake Henley should be sufficient to prevent a build up of nutrients and the likelihood of a phytoplankton bloom. However, the use of chemicals for control of weeds in Lake Henley has in the past met with opposition from the Masterton public. The short time required for macrophytes to regrow also means that this method of weed control is unlikely to be effective for long term control.

(ii). Mechanical harvesting

Weed control using mechanical harvesters has been attempted in the past (D. Crockett *pers comm.*), but was found to be highly labour intensive. The high costs involved, and the short period of effective control before plant regrowth meant that mechanical harvesting methods of weed control are also not suitable for the long term control of submerged macrophytes in Lake Henley.

(iii). Habitat manipulation

Manipulating the physical characteristics of the environment can make it less conducive for aquatic plant growth. These include such techniques as water level drawdown, shading with coloured chemicals, lining the lake bottom, or removing sediment. None of these techniques would be an effective long term solution for the weed problems in Lake Henley.

(iv). Biological control

Having assessed the above options for weed control as inappropriate long term solutions for Lake Henley it was decided to investigate the possibility of biological control (Buchanan 1991). The only biological control agent available in New Zealand for the control of submerged aquatic weeds is the herbivorous fish, Grass Carp (*Ctenopharyngodon idella*). Trials with Grass Carp in New Zealand have found them to be highly effective at removing submerged vegetation (Rowe & Schipper 1985), and have been found to be especially effective at consuming the species of weed (Edwards 1974) and filamentous algae (Rowe & Schipper 1985) present in Lake Henley. Approval for the release of Grass Carp into Lake Henley was granted in April 1994 and an introduction of a low to medium stocking rate of 20 fish/ha was made in November 1994 following the establishment of an operational plan for the management of the fish (Miller 1994).

4.2.2 Aquatic weed management using biological control

(i) Establishment of stocking rates

Given the benefits of maintaining a proportion of submerged vegetation in Lake Henley the objective of using Grass Carp is then to manage weed bed levels rather than try to eliminate all aquatic plants. This has entailed stocking the lake at a rate at which a reduction in plant biomass, and the creation of a patchy distribution of plants, is achievable rather than complete elimination of all submerged vegetation. There is however not a straight forward formula with which to calculate optimum stocking rate, and to do so is extremely difficult given differences between lakes in macrophyte standing crop, macrophyte species, growth and regrowth of macrophytes, water temperature, fish growth and mortality, and the unpredictable annual variations of these factors (Mitchell 1980). A low stocking rate of 20 fish/ha has been made in Lake Henley initially in the hope that at this density the fish will eat holes in surface reaching vegetation to produce a patchy, diverse environment in which Grass Carp will coexist with wildfowl, other fish and invertebrates (Buchanan 1991). Alternative figures in the area of 50-100 fish/ha have also been suggested as necessary to meet this objective (N. McCarter *pers. comm.*), thus it appears that effective weed management will require close monitoring of the weed beds after the Grass Carp introduction, with readjustments of stocking rate (i.e., by adding or removing fish) as necessary.

A further influence likely to impact on the predicted effectiveness of Grass Carp grazing, and therefore on determining an appropriate stocking rate, is that of temperature. The feeding behaviour of Grass Carp is strongly influenced by water temperature (Rowe & Schipper 1985). Feeding rates of these fish increase logarithmically as temperature rises (Wiley *et al* 1987) with temperatures between 21°C and 26°C considered optimal, and negligible feeding occurring below 10°C (Rowe & Schipper 1985). Grass Carp can therefore be expected to be most active in Lake Henley between December and March (Fig.4.1), and are predicted to consume very little between May and August. However, this suggests that Grass Carp are going to be most active during the summer season when the need for weed control measures is greatest, implying that they are still likely to be highly effective weed control agents although an increase in the stocking rate may be necessary to compensate for the short consumption period.

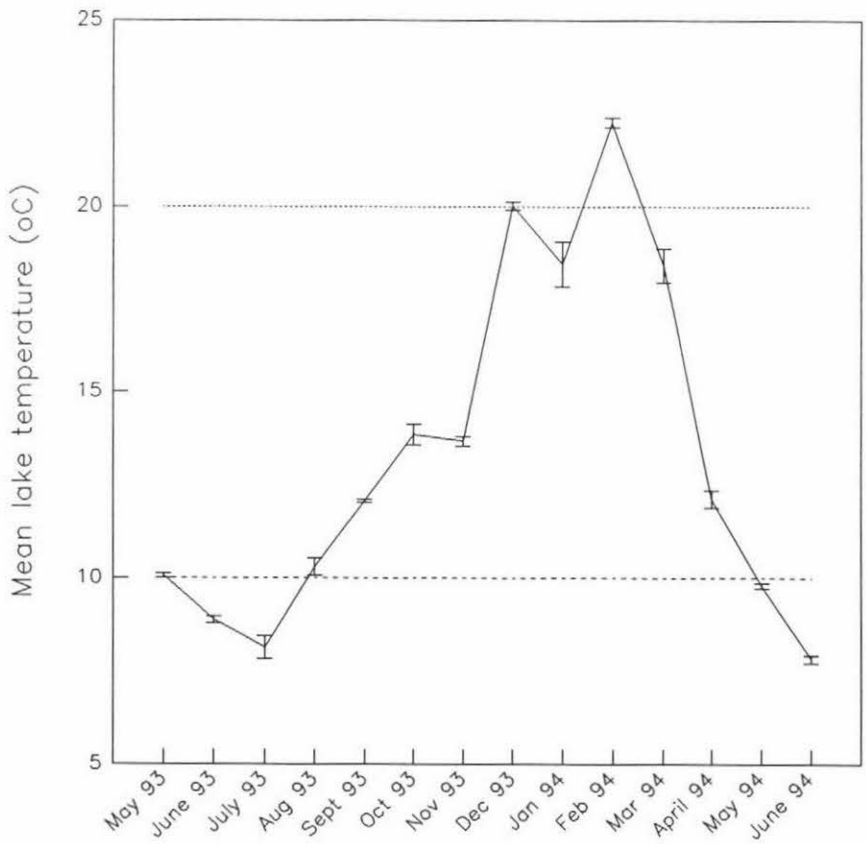


Fig. 4.1. Mean lake temperatures from Lake Henley between May 1993 and June 1994. Grass Carp feeding negligible below 10°C (------), and optimal above 20°C (.........).

(ii) Complying with security requirements for introduced fish

Security arrangements have been made to reduce the likelihood of Grass Carp escaping from the lake. Hiona and Te Ore Ore streams (flowing into the stream) both contain culverts that would effectively prevent fish from migrating upstream. No upstream movement is possible along the inflow back up to the Ruamahunga River, as the 1.5 m weir half way along the inlet channel forms an effective fish barrier. Permanent steel grids have been fitted onto both outlet pipes to the adjacent wetland areas. A rotating grid designed to prevent the passage of anything bigger than 2.5 cm has been fitted to the main outlet. This grid has been designed to allow for the clearing of weed fragments that accumulate at the outflow. The outflow grid system will prevent Grass Carp from escaping down the main outflow route and into the Ruamahunga River. The possibility of vandalism occurring to the outflow gird is reduced by the presence of the lake caretaker and security patrols, and any damage would be detected within eight hours and repairs begun immediately. If, however, Grass Carp did manage to escape from the lake because of vandalism on the outflow grid an immediate repair of the grid would need to be made to keep the number of fish escaping to a minimum. The Department of Conservation would be also need to be notified (Miller 1994).

(iii) Ongoing monitoring programme

The primary objective of the weed control programme is to decrease the submerged vegetation cover in the lake by 50% within six years, recognising the value of maintaining some aquatic plants. It is aimed to achieve this level of vegetation cover by keeping the Grass Carp density at appropriate stocking levels.

In all other lake Grass Carp trials in New Zealand the primary objective has been the total elimination of weed (Rowe & Schipper 1985). Partial removal and control at a specific level, such as desired for Lake Henley, therefore constitutes an experiment in management. Ongoing monitoring of the vegetation beds is required to assess the effectiveness of the Grass Carp with respect to this objective and to allow modification of the stocking rate if necessary. It is also important to monitor macrophyte species composition in the lake to ensure that holes created in the weed beds by the Grass Carp are not occupied by less palatable aquatic plants or species that may be more difficult to control. Maintenance of policies that prevent the introduction of other macrophyte species, such as restricting the use of motor boats on the lake, will help ensure that more vigorous growing macrophyte species, such as *Largarosiphon major*, do not add to the weed problem in the lake.

Vegetation surveys are recommended at two yearly intervals following the introduction of the Grass Carp to monitor weed management. The impact of the Carp is unlikely to be noticeable before this time. Vegetation surveys to assess the impact of the Grass Carp can take two forms:

(i) Qualitative assessment.

When water clarity is high and the vegetation is clearly visible below the water surface, the presence of holes in the vegetation cover can be easily assessed from the water surface. However, this method is not possible under conditions of low visibility, i.e., directly after a period of stormy weather.

(ii) Quantitative assessment.

The presence or absence of vegetation at 2m points along transect lines used in the 1993 survey (Chapter One) can be ascertained either by visually assessing each point if water visibility permits, or by using a hook to grab some vegetation at that point. The later method was used in the 1993 survey. A comparison with the 100% vegetation cover recorded in that survey will provide an indication of the Grass Carp effectiveness.

If there is no indication of Grass Carp activity after the first two years, i.e., no obvious holes in the vegetation cover, then an additional stocking of Grass Carp will need to be made. Raising the stocking density to 30 fish/ha by an additional release of 100 fish would be recommended. If more than 75% of the vegetation has been removed in the first two years then attempts to reduce the stocking density woulds need be made by removing some fish. Total collapse of the weed beds could potentially lead to a situation in which recolonisation of the lake by macrophytes would be prevented by high suspended sediment levels and an associated reduction in light availability. Should this eventuate the result would be a visually unappealing turbid lake. This situation is highly unlikely given the initial small size of the fish, low stocking rates, and the number of months predicted to be too cold for Grass Carp feeding.

After six years of biannual vegetation surveys if the reduction in weed distribution has not reached the desired 50% level, consideration will have to be given to increasing the stocking density of Grass Carp further, or to supplementing the Grass Carp activities with other forms of weed control. Incremental additions of 100 fish (i.e., increasing stocking rate by 10 fish/ha) is suggested. Further additions of fish to the base stocking rate will be necessary to maintain the population at an effective level because of natural mortality and possible predation events.

4.3 MANAGEMENT OF WATER QUALITY

Water of high quality has a high degree of clarity, an absence of filamentous or planktonic algal scums, low concentration of nutrients required for plant growth, and low faecal coliform counts. Water quality is a subjective variable that depends on the required uses for the water body (Vant 1987). In Lake Henley the clarity of the water is generally quite high with visibility to the surface of the weed beds. The water quality of Lake Henley, as perceived by the lake users, deteriates with excessive weed growths (as discussed above), excessive filamentous algal scums (that grow readily in warm, nutrient enriched waters), and with reduced clarity (which occurrs following resuspension of sediments by wind action, or with the addition of silt laden water from a flooded river).

4.3.1 Management of lake nutrient concentrations

In highly developed catchment areas the nutrient concentrations of the streams, rivers, ground water and lakes is high (Vant & Hoare 1987). The lake is fed by water that flows though developed and urban land, and is consequently nutrient enriched, allowing growths of filamentous algae associated with the submerged aquatic plants to occur. This algae forms dense floating mats (Hillebrand 1983) and is commonly found forming scums in slow flowing regions of low lying rivers during summer low flow periods.

Excessive macrophyte plant growth has, in the past, been related to increases in plant available nutrients such as nitrate nitrogen and dissolved reactive phosphorus (Hoare 1982; Hughes 1976). However, observations of weed problems in oligotrophic (nutrient poor) lakes caused by an over abundance of tall exotic weed species suggests that these plants cannot be managed by nutrient load reduction (Hoare 1982). Increased nutrient loads have instead been linked to increases in the growth of suspended (phytoplankton) and filamentous algae (Pokrovskaya 1984) as described above. Thus, although examination of the nutrient concentrations in Lake Henley is likely to be of little consequence to the management of the weed beds, nutrient loads of the lake do have a direct influence on water quality as assessed by the quantity of filamentous algal scums.

The flow through of water in Lake Henley is relatively rapid and consequently seasonal variation in nutrient and hydraulic loads is important (Hoare 1982). There are three major sources of nutrients to Lake Henley: Hiona stream, Te Ore Ore stream, and the main inflow. Nutrients entering via groundwater or rainwater are likely to be minimal. Between May 1993 and August 1994 an estimated 180, 2, and 504 kg/year of dissolved reactive phosphorus (DRP), and 1152, 108, and 6168 kg/year of nitrate nitrogen entered Lake Henley through Hiona stream, Te Ore Ore stream, and the main inflow, respectively (Fig.4.2). This indicates that the bulk of the nutrients entering Lake Henley did so through the main inflow, thus any attempts to reduce the nutrient loading of the lake by plans to divert one or other of the smaller inflows away from the lake (MDC 1993) are unlikely to affect the total nutrient concentration of water in the lake. The amount of both phosphorus and nitrogen leaving the lake were less than the amount entering it (Fig.4.2). These nutrients were taken up by algae and other plants growing in the lake and eventually became incorporated into the sediments. This suggests that water flowing through the lake is stripped of some of its nutrient load prior to re-entering the Ruamahunga River.

The average DRP and NO_3-N concentrations in the lake were19.5 and 163 mg/m^3, respectively, although there was considerable seasonal variation in these values. These nutrient levels are of the magnitude recorded for nutrient enriched lakes in highly developed catchments

(Fig.4.3) such as Dudding's Lake in the Wanganui region. However, the chlorophyll-*a* concentrations (a measure that indicates biomass of phytoplankton) were at a level expected only for oligotrophic lakes such as Lake Okataina. Thus, despite high nutrient levels in Lake Henley, there is very little microscopic green algae, probably because of the high flushing rate of water through the lake (Pridmore 1987). Instead, the high levels of nutrients appear to be utilised by growths of filamentous algae (Biggs 1985). In turn these thick mats of algae appear to inhibit the growth of the weed beds and are suspected to cause the macrophytes to decay (Chapter One; Pokrovskaya 1984)

4.3.2 Managing water quality through reduction of sediment load

Water clarity in Lake Henley is generally high allowing visibility to the surface of the weed beds. After a flood event, or following resuspension of silt by winds, the visibility is severely reduced. The volume of silt entering the lake through the river when it is in flood is contributing to the gradual filling in of the lake. Reducing this impact by manually closing the intake gate is presently the only way of preventing silt laden water entering the lake. Suggestions to install an automatic turbidity meter at the site of the intake have been made (G. Daniels *pers comm*; I. Buchanan *pers comm*) but difficulties, and the cost involved, with getting electricity to the area have thus far prevented these suggestions from being implemented.

Silt also enters the lake in large quantities during flood events from Hiona stream. The increased depth of silt beneath the macrophyte beds situated on the side of the lake near this stream contributes to the increased biomass of weed at this site (Chapter One), and the increased abundance of oligochaetes (worms)in the invertebrate community associated with the macrophytes (Chapter Two).

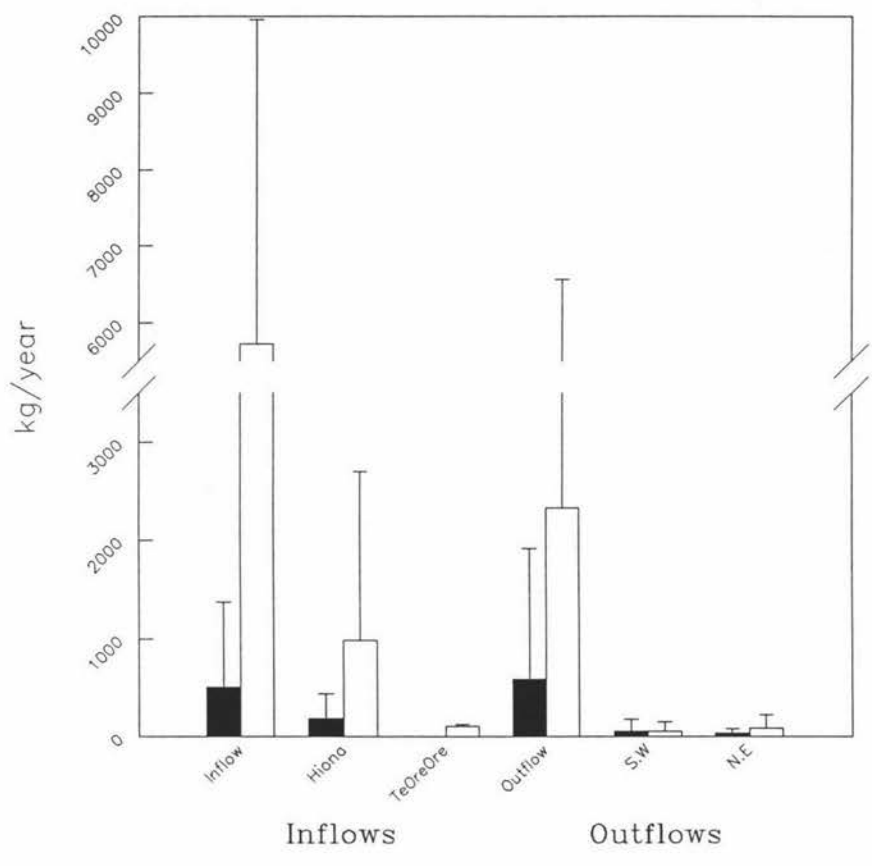


Fig.4. 2 Mean amount (in kg/year) of plant available phosphorus (■) and nitrogen (□) entering Lake Henley from the three inflows.

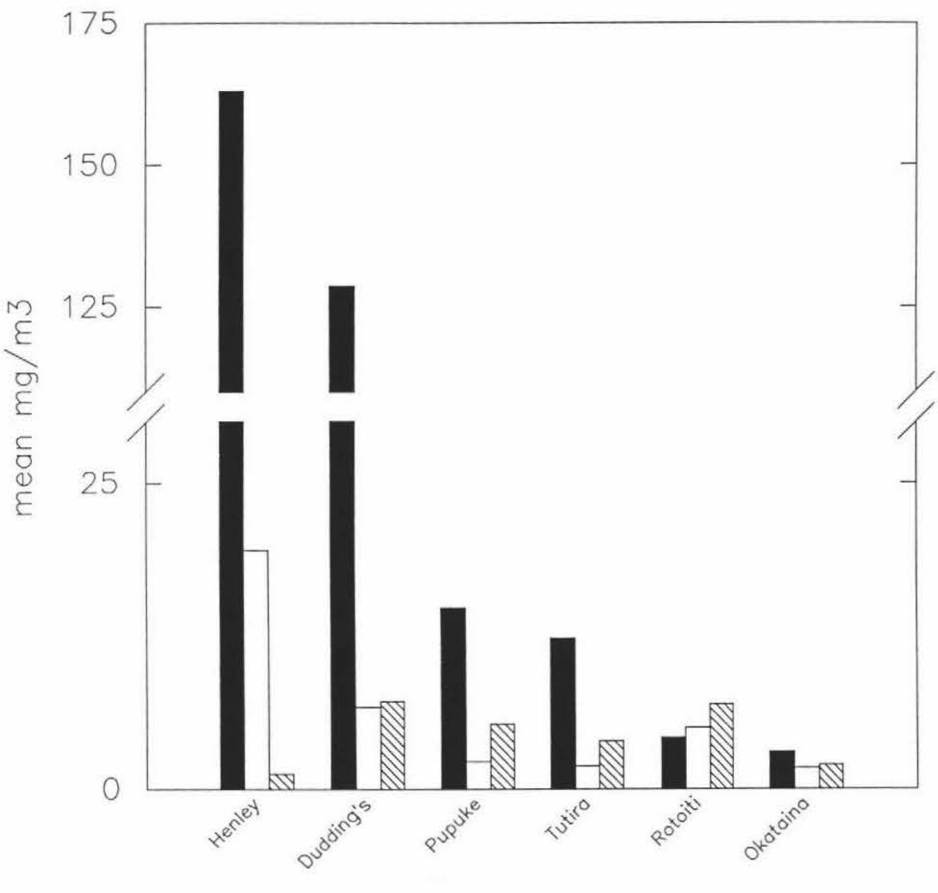


Fig 4.3. Mean Nitrate nitrogen ( ), Dissolved reactive phosphorus ( ), and Chlorophyll-*a* ( ) concentrations from a range of New Zealand lakes. Values for lakes other than Lake Henley obtained with permission from unpublished data courtesy of N. Burns, NIWA, Hamilton.

4.3.3 Using invertebrates to monitor water quality

While coliform counts are a standard measure of water quality, and form standards for permitting contact recreational activities, they only provide an indication of the conditions at the time and place of sampling (Vant 1987). Furthermore, the marked fluctuations that occur in population density of bacteria occur at short intervals, reflecting variation in catchment loads, and require intense constant sample monitoring for accurate assessment. A potentially better indicator of environmental condition is the composition of invertebrate communities as they respond to long term changes in water quality (Stark 1994). Macroinvertebrate communities associated with the benthos of streams and rivers have a long history of being used to assess water quality (Metcalfe 1989; Stark 1994). Benthic organisms are differentially sensitive to pollutants of various types, and as most macroinvertebrates have short life cycles they respond rapidly to habitat alteration, or to changes in water quality.

A 13 month study of the invertebrates associated with macrophytes in Lake Henley revealed that although there were some differences between community composition between different parts of the lake, seasonal differences were of greater significance (Chapter Two). The invertebrate communities collected from a site with greater depth of silt beneath the macrophytes contained a greater abundance of oligochaetes (worms) than the other sites. This suggests that as Lake Henley ages, and undergos increasing siltation the composition of the invertebrate communities will possibly undergo a shift to a community dominated numerically by oligochaetes (Minshall 1984).

Lake Henley appeared to have a lower diversity of insect taxa compared with other New Zealand lakes. This may relate to the young age of the lake. Invertebrates associated with running water environments could easily colonise the lake via the inflowing streams or river, while others have probably been carried there as eggs on the feet of aquatic birds. Larvae of insects that rely on colonisation by adults are however likely to take longer to reach this new habitat particularly if their adults do not disperse widely (Sheldon 1984). The lower diversity of insects could also be associated with the monoculture of *Elodea canadensis* growing in the lake. Proposals to manage the weed beds in Lake Henley at a level designed to create a more patchy distribution, with the possibility that this could increase the diversity of aquatic plants in the lake, are likely to lead to an increase in the biodiversity of macroinvertebrates associated with the weed.

When overall community composition was compared between Lake Henley and other North Island lakes it was found that the invertebrate communities associated with the macrophtyes in Lake Henley were quite similar to those from Lake Otomanakau. This hydro lake from the Central Volcanic Plateau region also has relatively high nutrient concentrations and low chlorophyll-*a* concentrations because of its low residency time. The invertebrate communities from both of these lakes were characterised by a high density and diversity of molluscs especially *Potamopyrgus antipodarum*, *Physa* sp., and *Gyraulus corinna* and two insect species: *Antiporus* sp., and *Chironomus zealandicus*.

Although there is currently no national standard or index to enable comparison of the invertebrate communities associated with submerged aquatic plants, it would appear that the diversity present in Lake Henley communities indicate reasonably high water quality. Comparison of invertebrate communities in the future with the species list established from the present study (Appendix 5) will provide a means of monitoring long term changes in lake water quality.

4.3 MANAGEMENT OF WATER QUANTITY

Resource consents are required under the 1991 Resource Management Act to divert water from the Ruamahunga River, carry out maintenance work on the gravel weir at the intake, and discharge water from the lake into the river. These were applied for in June 1995. During periods of summer low flows, the river is vulnerable to experiencing elevated temperatures, decreased oxygen levels and an increase in filamentous algae growth. Lake management decisions therefore need to consider the maintenance of appropriate water levels to safe guard the environmental state of the river, and to also ensure that the flushing rate through the lake is maintained at a rate at which phytoplankton blooms are inhibited.

The amount of water estimated to have been diverted into Lake Henley between May 1993 and March 1995 ranged from 1% to 30% of the river at the intake point (based on gaugings from Mt. Bruce and Wardell's stations, WRC unpublished data) (Fig. 4.4). The greatest percentage water diversion occurred between January and April 1994, a summer low flow period. Detailed flow measurements on March 8 1995, again during a summer low flow period, found that 16 % of the total flow above the diversion site was diverted into Lake Henley, and 82% of that volume was returned to the river directly through the main inflow, yielding a total loss from the river of only 6.5%. By ensuring that not more than 25% of the total volume at the intake site is diverted, the impact of the diversion on the river should be minimal.

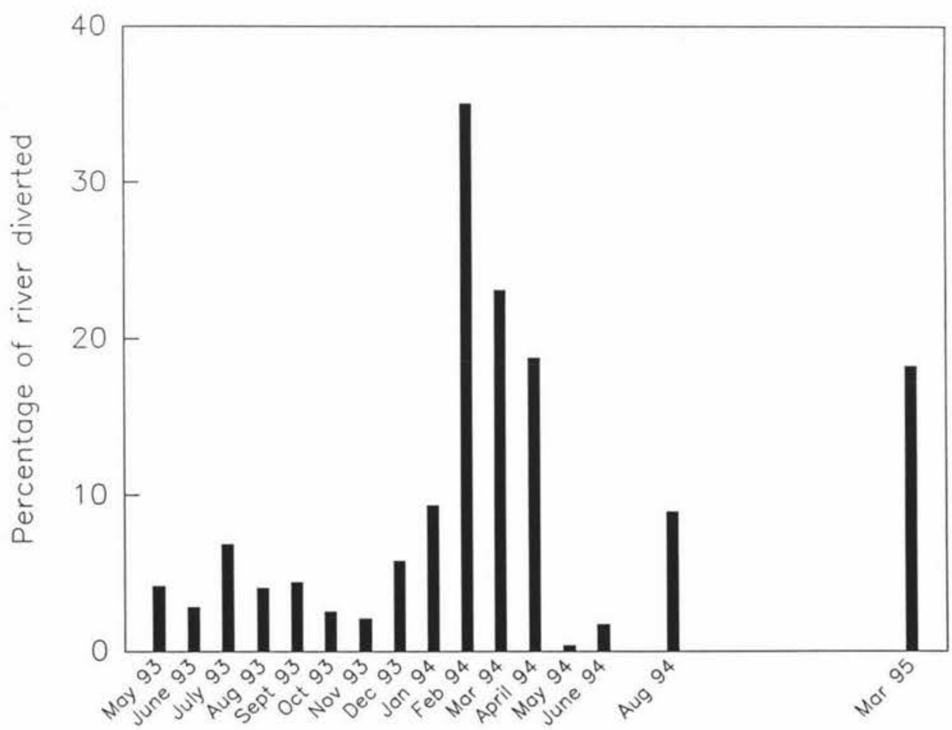


Fig 4.4. Percentage of total flow of Ruamahunga River at the location of the inlet diverted into Lake Henley. Estimates based on unpublished data courtesy of the Hydrology department, Wellington Regional Council.

SUMMARY OF RECOMENDATIONS

1. A weed bed monitoring programme should be established to assess the change in the degree of weed cover in the lake that result from Grass Carp grazing, and in the species composition of the aquatic weeds.

2. Based on the outcome of such further monitoring the weed control programme is envisioned to involve making further additions to the Grass Carp stocking rate and supplementing the grazing activities with minor chemical control of weed beds to satisfy the requirements of user groups.

3. Methods and benefits of reducing nutrient concentration of the water entering the lake via the main inflow require further investigation. This may involve consultation with the land owners to reduce stock access to the water way.

4. The amount of silt entering the lake needs to be reduced. The feasibility of establishing an automatic turbidity meter at the point of intake should be explored further on a cost-benefit basis.

5. A system for gauging the percentage of the Ruamahunga river diverted into Lake Henley needs to be established to ensure that not more than 25% of the river's flow is diverted, especially during periods of low flow. This will also be required to compile with resource consents applied for.

6. Through consultation with user groups a cost-benefit analysis needs to be devised to provide guide-lines for the appropriate size of a lake management budget to enable the carrying out of the above suggested monitoring and maintenance programmes.

CONCLUSIONS

The principal aim of the management of Lake Henley, as detailed in the Lake Henley Park Management Plan, is to provide for the sustainable management and development of Lake Henley Park for predominantly passive recreational use, while preserving its unique ecological character and balance (MDC 1993). In order to ensure that the lake's development follows a natural course, the type of management adopted needs to be as least intrusive as possible. However, because of the artificial nature of the lake a certain degree of active intervention is required. The result of no intervention for a lake of this nature would be a reversal to wetland, and this would not prove acceptable to the majority of recreational users. Therefore the requirements of lake management are to (i) monitor the progress of the adopted weed control methods, and take the appropriate action as described above, (ii) monitor water quality and take actions to reduce sediment loads or nutrient levels in order to maintain the water quality of the lake, and (iii) monitor the amount of water diverted into the lake, especially during periods of low flow, to ensure that the ecological health of the Ruamahunga River is maintained.

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Appendix 1: Discharge rates for inflows into Henley Lake (Hiona Stream, Te Ore Ore Stream and the Main inflow), and outflows from Henley Lake (to N.E. Wetlands, S.W. Wetlands, and the Main outflow) between May 1993 and August 1994. Discharges measured as m^3/s. Also shown is the percentage of inflowing discharge accounted for by the outflowing discharge.

	May 93	June 93	July 93	Aug 93	Sept 93	Oct 93	Nov 93	Dec 93	Jan 94	Feb 94	Mar 94	April 94	May 94	June 94	Aug 94
Hiona Stream	0.074	0.047	0.022	0.033	0.028	0.055	0.063	0.015	0.049	0.03	0.064	0.042	0.007	0.018	0.0868
Te Ore Ore Stream	0.0013	0.0013	0.0013	0.0017	0.0013	0.0011	0.0013	0.0013	0.0012	0.001	0.0011	0.0013	0.0014	0.0013	0.0013
Main inflow	0.497	0.843	0.97	0.288	0.325	0.553	0.351	0.282	0.844	0.686	0.664	0.596	0.127	0.311	0.778
Main outflow	0.3791	0.676	1.219	0.177	0.126	0.446	0.263	0.131	0.248	0.422	0.405	0.387	0.254	0.102	0.452
Outflow to N.E wetlands	0.0217	0.0217	0.0292	0.0117	0.0140	0.0227	0.0351	0.0241	0.0077	0.0261	0.0261	0.0206	0.0330	0.0154	0.0164
Outflow to S.E wetlands	0.0216	0.0216	0.0216	0.0216	0.0216	0.0216	0.0261	0.0226	0.0215	0.0171	0.0297	0.0206	0.0203	0.0131	0.0239
Total inflow	0.5723	0.8913	0.9933	0.3227	0.3543	0.6091	0.4153	0.2983	0.8942	0.7170	0.7291	0.6393	0.1354	0.3303	0.8661
Total outflow	0.4225	0.7193	1.2698	0.2103	0.1616	0.4904	0.3242	0.1777	0.2772	0.4652	0.4608	0.4282	0.3072	0.1305	0.4923
Percent	73.82	80.71	127.84	65.18	45.62	80.51	78.07	59.57	31.00	64.88	63.20	66.98	226.90	39.51	56.84

Appendix 2: Phosphorus concentrations in inflows (Hiona Stream, Te Ore Ore Stream, and the main inflow) to Lake Henley, and in outflows (to N.E. Wetlands, S.W wetlands, and the main outflow) during May 1993 to August 1994. Phosphorus loadings derived by multiplying concentration of DRP by discharge.

	May 93	June 93	July 93	Aug 93	Sept 93	Oct 93	Nov 93	Dec 93	Jan 94	Feb 94	Mar 94	April 94	May 94	June 94	Aug 94
A. DRP concentrations:															
Hiona Stream	19	43	10	23	18	39	30.5	1606	241.5	51.05	30.3	238.6	144.5	102.3	292
Te Ore Ore Stream	7	134	207	21	29	24	35.5	55.98	28.32	29.73	39.1	23.8	38	22.5	226
Main Inflow	14	11	11	12	23	42	25.3	4.51	8.4	3.37	3.44	8	22.8	104.1	148
Main Outflow	m	18	135	13	7	104	7.7	7.19	5	5.95	12.81	9	26.7	12.6	25
Outflow to N.E Wetlands	m	m	512	12	6	71	12.6	3.07	8.7	5.23	4.91	14.8	35.5	24	195
Outflow to S.W Wetlands	m	m	m	m	m	m	m	m	m	10	5.65	19.2	44	13.9	181
B. DRP loadings:															
Hiona Stream	1.41	2.02	0.22	0.76	0.50	2.15	1.92	24.09	11.83	1.53	1.94	10.02	1.01	1.84	25.34
Te Ore Ore Stream	0.01	0.17	0.27	0.04	0.04	0.03	0.05	0.07	0.03	0.03	0.04	0.03	0.05	0.03	0.29
Main Inflow	6.96	9.27	10.67	3.46	7.48	23.23	8.88	1.27	7.09	2.31	2.28	4.77	2.90	32.38	115.14
Main Outflow	m	12.17	164.57	2.30	0.88	46.38	2.03	0.94	1.24	2.51	5.19	3.48	6.78	1.29	11.3
Outflow to N.E Wetlands	m	m	14.93	0.14	0.08	1.61	0.44	0.07	0.07	0.14	0.13	0.31	1.17	0.37	3.20
Outflow to S.W Wetlands	m	m	m	m	m	m	m	m	m	0.17	0.17	0.40	0.89	0.18	4.32
Total DRP entering lake:	8.37	11.47	11.16	4.25	8.02	25.40	10.85	25.43	18.96	3.87	4.27	14.82	3.96	34.25	140.78
Total DRP exiting lake:	m	12.17	179.49	2.44	0.97	48.00	2.47	1.02	1.31	2.82	5.48	4.18	8.84	1.84	18.82

Appendix 3: Nitrate nitrogen concentrations in inflows (Hiona Stream, Te Ore OreStream, and the main inflow) to Lake Henley, and in outflows (to N.E. Wetlands, S.W wetlands, and the main outflow) during May 1993 to August 1994. Phosphorus loadings derived by multiplying concentration of DRP by discharge.

	May 93	June 93	July 93	Aug 93	Sept 93	Oct 93	Nov 93	Dec 93	Jan 94	Feb 94	Mar 94	April 94	May 94	June 94	Aug 94
A. NO3 concentrations (mg/m^3):															
Hiona Stream	2750	1875	700	400	0	0	506.3	26.96	22.37	59.85	34.2	50.5	221.9	1084.2	1022
Te Ore OreStream	3900	2900	2200	2100	2600	2100	2869.1	3172.5	3054	2174.9	3023	2086.5	2083.1	2222.1	2517
Main Inflow	750	375	400	400	600	0	313.8	248.63	197	93.26	189.03	245.5	378.3	442.2	599
Main Outflow	0	575	300	0	0	0	8.1	13.94	8.5	6.47	7.23	51.4	70.4	283	449
Outflow to N.E Wetlands	0	0	300	0	0	0	6	21.81	6.2	2.63	4.82	33.6	36.3	289.9	453
Outflow to S.W Wetlands	m	m	m	m	m	m	m	m	m	2.68	2.12	1.1	8.2	302.1	495
B. NO3 loadings (mg/s):															
Hiona Stream	203.50	88.13	15.40	13.20	0.00	0.00	31.90	0.40	1.10	1.80	2.19	2.12	1.55	19.52	88.69
Te Ore OreStream	5.07	3.77	2.86	3.57	3.38	2.31	3.73	4.12	3.66	2.17	3.33	2.71	2.92	2.89	3.27
Main Inflow	372.75	316.13	388.00	115.20	195.00	0.00	110.14	70.11	166.27	63.98	125.52	146.32	48.04	137.52	466.02
Main Outflow	m	388.70	365.70	0.00	0.00	0.00	2.13	1.83	2.11	2.73	2.93	19.89	17.88	28.87	202.95
Outflow to N.E Wetlands	m	m	8.75	0.00	0.00	0.00	0.21	0.53	0.05	0.07	0.13	0.69	1.20	4.48	7.44
Outflow to S.W Wetlands	m	m	m	m	m	m	m	m	m	0.05	0.06	0.02	0.17	3.95	11.81
Total NO3 entering (mg/s):	581.32	408.02	406.26	131.97	198.38	2.31	145.77	74.64	171.03	67.95	131.03	151.15	52.51	159.93	557.98
Total NO3 exiting (mg/s):	0.00	388.70	374.45	0.00	0.00	0.00	2.34	2.35	2.16	2.84	3.12	20.61	19.24	37.29	222.20

Appendix 4: Dissolved Reactive Phosphorus (DRP) and Nitrate Nitrogen (NO_3-N) concentrations at four sites within Henley Lake between February 1993 and August 1994.

(A)Dissolved Reactive Phosphorus (mg/m^3)

	Feb 93	Mar 93	May 93	June 93	July 93	Aug 93	Sept 93	Oct 93	Nov 93	Dec 93	Jan 94	Feb 94	Mar 94	April 94	May 94	June 94	Aug 94
Top Lake	11	32	41	21	192	10	34	29	m	24.37	18.1	8.25	5.52	11.2	17.5	11.3	151
Mid Lake	8	13	10	27	16	13	47	54	17.1	m	6.35	12.23	4.41	9.4	14.7	12.9	152
End Lake	m	m	m	18	12	14	47	33	7.1	9.53	7.4	5.78	7.2	13.3	9.2	9.2	148
Side Lake	m	21	16	21	12	10	49	49	14.9	5.36	19.96	9.39	11.33	14.5	19.8	17.2	175

(B) Nitrate Nitrogen concentrations (mg/m^3)

	Feb 93	Mar 93	May 93	June 93	July 93	Aug 93	Sept 93	Oct 93	Nov 93	Dec 93	Jan 94	Feb 94	Mar 94	April 94	May 94	June 94	Aug 94
Toplake	<250	<250	1750	625	300	<250	<250	<250	m	3.93	7.4	2.58	15.75	1.1	73.2	273.2	500
Mid Lake	70	<250	625	500	300	<250	<250	<250	65.3	m	7.59	3.74	30.99	138.7	102.5	261.1	387
End Lake	m	m	m	575	300	<250	<250	<250	40.8	25.62	7.3	2.68	8.76	53.7	95.1	287.3	481
Side Lake	m	<250	1150	350	300	250	<250	<250	59.3	27.54	15.27	3.91	25.32	37.6	211.9	276.9	572

Appendix 5: Species list of invertebrates collected from Lake Henley and Hiona Stream macrophytes between June 1993 and June 1994.

COELENTERATA
Hydra

PLATYHELMINTHES
Platyhelminthe sp. A
Platyhelminthe sp. B

NEMATODA

ANNELIDA
Oligochaeta: *Lumbriculus variegatus* (Müller 1774)
Chaetogaster sp.
Oligochaete sp. S
Hirudinea: *Glossiphonia* sp. (Mason 1974)

ARTHROPODA
Crustacea: Amphipoda sp.
Clodocera: Bosminidae
Chydoridae
Daphniidae
Ostracoda sp.
Copepoda: Cyclopoidae

Insecta: Ephemeroptera
Odonta: *Xanthocnemis zealandica* (McLachlan 1873)
Procordulia grayi (Selys 1871)
Ischnura aurora
Hemiptera: *Diaprepocoris zealandiae*
Coleoptera: *Antiporus* sp.
Diptera: Chironomidae
Tanypodinae
Orthocladiinae
Chironominae: *Chironomus zealandicus* (Hudson 1892)
Stratiomyidae
Ceratopogonidae
Simuliidae
Trichoptera: *Oxyethira albiceps* (McLachlan 1862)
Paroxyethira sp.
Oecetis unicolor (McLachlan 1862)
Hudsonema amabilis (McLachlan 1868)
Triplectides cephalotes (Walker 1852)
Lepidoptera: *Hygraula nitens*
Collembolla

Acari:	Pionidae Orbatidae Acarina sp. C Acarina sp. D Acarina sp. E
Mollusca:	*Gyraulus corinna* (Gray 1850) *Potamopyrgus antipodarum* (Gray 1843) *Physa* sp. *Sphaerium novaezelandiae* (Deshayes 1851) *Ferrisia neozelanica*

Appendix 6: Species List of invertebrates collected from survey of phytomacrofaunal communities from 14 North Island Lakes, May 1994.

COELENTERATA
Hydra
Brazoan

PLATYHELMINTHES
Platyhelminte sp. A
Platyhelminte sp. B
Platyhelmintes spp.

NEMATODA

ANNELIDA
Oligochaeta: *Lumbriculus variegatus* (Müller 1774)
Chaetogaster sp.
Oligochaete sp. C

Hirudinea: Hirudinea sp. A
Glossiphonia sp. (Mason 1974)

ARTHROPODA
Crustacea: Amphipoda sp.
Clodocera: Bosminidae
Chydoridae
Daphniidae
Ostracoda sp.
Copepoda: Cyclopidae
Calanoidae
Tanais stanfordi
Paranephrops sp.

Insecta: Odonta: *Xanthocnemis zealandica* (McLachlan 1873)
Procordulia grayi (Selys 1871)
Ischnura aurora
Hemiptera: *Hemicordulia australiae*
Diaprepocoris zealandiae
Coleoptera: *Antiporus* sp
Elmidae
Diptera: Chironomidae
Tanypodinae: Macropelopiini
Orthocladiinae sp. A
Orthocladiinae sp. B
Orthocladiinae spp.
Chironominae: *Chironomus zealandicus* (Hudson 1892)
Chironomus sp.a
Cryptochironomus sp.

Tanytarsus vesperatinus (Hutton 1902)
Tanytarsus funebris
Parachironomus cylindricus
Polypedilum

Stratiomyidae
Ceratopogonidae

Trichoptera: *Oxyethira albiceps* (McLachlan 1862)
Paroxyethira sp.
Paroxyethira hendersoni (Mosely 1924)
Paroxyethira tillyardi (Mosely 1924)
Oecetis unicolor (McLachlan 1862)
Hudsonema amabilis (McLachlan 1868)
Triplectides cephalotes (Walker 1852)

Lepidoptera: *Hygraula nitens*

Collembolla

Acari:
Pionidae
Orbatidae
Acarina sp. C
Acarina sp. D
Acarina sp. E
Acarina sp. F

Mollusca:
Gyraulus corinna (Gray 1850)
Potamopyrgus antipodarum (Gray 1843)
Physa sp.
Physa variabilis
Lymnaea columella
Lymnaea tomentosa
Sphaerium novaezelandiae (Deshayes 1851)
Ferrisia neozelanica

SYNTHESIS

Lake Henley is a shallow macrophyte dominated lake supplied with nutrient rich water from the Ruamahunga river. The high flushing rate through the lake prevents high phytoplankton densities although dense mats of filamentous algae do occur periodically. The presence of filamentous algae on the macrophyte beds appeared to influence macrophyte growth and decay patterns and may also be partially responsible for seasonal changes in the macrophyte-associated invertebrate communities. This is possibly because the filamentous algae altered the structural complexity of the macrophytes, provided additional food resources for the invertebrates, and encouraged decay of the macrophytes which in turn increased macrophyte palatability. The phytomacrofaunal community in Lake Henley had a lower proportion of insect taxa than similar lake communities in New Zealand, perhaps because of the low floral diversity, lack of depth variation, or the absence of a nearby colonization source.

Phytomacrofaunal community structure appeared to be influenced by the trophic status of the lake that they occurred in. Communities from lakes with a high trophic status were characterized by high abundance but low species richness whereas invertebrate communities from the remaining lakes had higher species richness but lower abundance. Specific groups of invertebrate taxa associations were found in communities from lakes of low trophic status suggesting they may be useful for monitoring changes in nutrient status, such as those resulting from a lake aging, or from increasing eutrophication. However, the general low species richness recorded for New Zealand phytomacrofaunal communities means that other attributes of the communities such as relative abundances are likely to be of more importance in detecting differences in the communities than merely the presence or absence of specific taxa.

Given the increased awareness of the importance of monitoring and evaluating impacts of human activities on New Zealand's freshwater resources it would appear that further investigations into the relationship of the phytomacrofaunal community to the nutrient status of the water body would be warranted.