Water Quality, Phytoplankton, Zooplankton and Fish in the University of Waikato Campus Lakes, Hamilton, New Zealand

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Abstract

The water quality, phytoplankton, zooplankton, fish, nutrient regimes, and mixing regimes of three small, shallow, hypertrophic, urban lakes situated on the University of Waikato campus, Hamilton were investigated. The lakes are spatially very close, in the same soil type, subject to the same climatic influences, and have similar nutrient concentrations. However, two of the lakes are shallow (Oranga and Knighton lakes – maximum depth 0.7m) and one of the lakes is slightly deeper (Chapel Lake – maximum depth 1.8m).

The lakes experienced strong temperature induced mixing. The water columns of the lakes were less stable at night than during the day. They were more stable in summer than in winter. The water column of Chapel Lake was significantly more stable than Oranga and Knighton Lakes. Oranga Lake was significantly more stable than Knighton Lake.

The concentrations of PO_4, NH_4, NO_2, and NO_x varied over the sampling period but were generally similar in all three lakes. Peaks of NH_4 occurred in Knighton and Chapel lakes. It is possible that low levels of dissolved oxygen caused nitrogen to be released from the sediment in the form of NH_4, which was converted to NO_2 and NO_x once oxygen levels increased. The exact mechanisms causing the changes in nutrient concentrations are unclear.

Water quality varied between seasons. Chapel Lake had consistently better water quality than Knighton and Oranga lakes. This was especially pronounced in summer when there were marked blooms of zooplankton and phytoplankton in Oranga and Knighton lakes but not in Chapel Lake. Significant differences were also found in the phytoplankton and zooplankton diversity and abundance between the lakes. Cyanobacteria and Euglenoids dominated the phytoplankton communities in Oranga and Knighton lakes while Chrysophytes and Chlorophytes were dominant in Chapel Lake. Rotifers and small cladocerans dominated the zooplankton communities in all of the lakes. The differences in phytoplankton communities between lakes were more pronounced than the differences in zooplankton communities.

Brown bullhead catfish (*Ameiurus nebulous*) and shortfinned eels (*Anguilla australis*) dominated the benthivorous fish communities of the lakes. There was a marked seasonal trend in catch rates with more fish being caught in the warmer months. There were significantly more catfish and eels caught in the two shallow lakes than in Chapel Lake, but significantly fewer common bullies (*Gobiomorphus cotidianus*). In addition, there were small eels caught in Knighton Lake and Oranga Lake in December but not in Chapel Lake. This indicates that shortfinned eels are able to navigate the stormwater system from the Waikato River to reach the lakes. The catch rates of catfish were lower than previous studies and the populations were dominated by fish aged 1-2 years. The catfish populations have the potential to increase.

The greater depth of Chapel Lake appears to protect it from the more extreme effects of hypertrophy experienced by Oranga and Knighton lakes. Artificial deepening is the option that is likely to provide the longest lasting improvements to water quality.

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Chapter 1

Thesis Introduction

GENERAL INTRODUCTION

Shallow lakes are important ecosystems in terms of the benefits they provide as well as their high abundance and distribution (Gulati & van Donk 2002); however there has traditionally been a lack of knowledge surrounding their ecosystem and structure, especially when compared to deeper lakes. Shallow lakes are very common worldwide, for example, in countries such as North Germany and Northern Poland (Nixdorf & Deneke 1997) and in Denmark and the Netherlands (Scheffer 1998). In the Waikato region shallow lakes are also very common (Barnes 2002). The mean depth of most Waikato lakes' is between 2.9 and 5 m (Green *et al.* 1987). Shallow lakes often suffer from eutrophication and declining water quality (Nixdorf & Deneke 1997; Sayer & Roberts 2001).

This thesis examines the general ecology of three small, shallow, hypertrophic, urban lakes in Hamilton, New Zealand, which experience problems typical of such lakes. In addition, lake water quality is related to the phytoplankton, zooplankton, and fish communities, and to the lakes' productivity, morphology, and mixing regimes.

BACKGROUND ON SHALLOW LAKES

The management of shallow lakes presents unique challenges. They are recognized as having high ecological, socio-economic and recreational values (Perrow *et al.* 1999) and can provide benefits through their hydrology and flood control (Gulati & van Donk 2002). However their relatively small volume makes them particularly prone to the negative effects of eutrophication, caused by internal and external nutrient loading (Wetzel 2001). They are also vulnerable due to the interactions between the lake sediment and water column (e.g., Kallio 1994), which can result in significant nutrient release from the sediment (e.g., Phillips *et al.* 1994).

Over the last few decades, there has been a worldwide increase in the incidence of anthropogenic eutrophication of lakes (Moss 1998) particularly in shallow lakes (Sayer & Roberts 2001). Lakes can apparently exist in one of two stable equilibriums; macrophyte-dominated or algal-dominated (e.g. Scheffer 1990; Scheffer *et al.* 1993; Perrow *et al.* 1999; Korner 2001). Eutrophication often leads to turbid, algal-dominated conditions (Moss 1998) with frequent blooms occurring during the summer months (e.g. Kallio 1994; Tryfon 2001) where cyanobacteria often dominate the phytoplankton community (e.g., Dokulil and Teubner 2000) Cyanobacteria often dominate shallow lakes that have hydraulic residence times of greater than 8-14 days (Olding *et al.* 2000). Shallow lakes with flat lakebed morphology are more likely to exhibit strong hysteresis between states compared with shallow lakes with varying depth profiles (van Nes *et al.* 2002).

There have been a number of attempts to remedy the effects of eutrophication in shallow lakes, through nutrient reduction, biomanipulation of fish species and sediment removal. A review of a shallow lake biomanipulations in the Netherlands is provided in Meijer *et al.* (1999) and management issues in the Norfolk Broads and in Scottish lochs are examined in Phillips *et al.* (1999) and Naysmith (1999) respectively. In New Zealand the trophic structure, water quality and ecosystems of shallow lakes are reasonably well known (e.g. Boswell 1985; Coffey *et al.* 1995, Mitchell and Wass 1996; Jeppesen *et al.* 2000; Barnes 2002). However, few New Zealand studies have examined whole lake interactions relative to lake depth and productivity.

SHALLOW VERSUS DEEP LAKES

The processes operating within lakes of different depths are often vastly different, which raise issues for lake management and restoration. Historically, studies have focused on deep, stratified water bodies at the expense of research on shallow lakes (Kufel *et al.* 1997). Restoration of eutrophied water bodies has focused particularly on reducing the external nutrient loading (Vant 1987), but this is not always effective when dealing with shallow lakes (Phillips *et al.* 1999). Nutrients often have a more pronounced effect on shallow water bodies than on deeper lakes due to the small volume and long hydraulic residence of shallow lakes (Wetzel 2001). Shallow lakes also generally have a higher nutrient input from sediments (Søndergaard *et al.* 1990). Where shallow lakes have their external nutrient loads reduced, internal recycling often explains the majority of phosphorus variation in the water bodies of shallow lakes (e.g., Van der Molan & Boers 1994; Kleeburg & Dudel 1997). Internal recycling of nutrients has been shown to account for 80% of the summer phosphorus budget of some shallow lakes (James *et al.* 2002) and can be nearly twice that of external phosphorus sources (Knuuttila *et al.* 1994). This often results in the lakes maintaining high levels of algal production despite large reductions in the external nutrient load (Moss *et al.* 1997; Graneli 1999). Nutrient dynamics are often varied and complex in shallow lakes and less inter-lake generalizations are possible compared with deeper, dimictic lakes (Havens *et al.* 2001). However, there has been less work done on plankton dynamics in frequently mixed water bodies than deep, stratified ones (Agbeti *et al.* 1997).

As the maximum depth decreases in urban lakes, so does the likelihood that phytoplankton assemblages can be predicted from the lakes' trophic status and the greater the influence that the hydraulic residence time and lake morphology have on algal communities (Olding *et al.* 2000).

URBAN LAKES

Urban lakes are often shallow, eutrophic to hypertrophic, and have limited conservation value. By definition, urban lakes drain highly developed areas and the quality of the runoff is often low (Hobman 2000). They are prone to heavy metal contamination from urban sources (Charlesworth & Foster 1999) and are subject to significant internal nutrient recycling (James *et al.* 2002). Many urban lakes have deteriorated in water quality and in plant and animal biodiversity (Moss *et al.* 2002). For these reasons, studies on urban lakes are often overlooked in favour of larger, more pristine water bodies (Birch & McCaskie 1999). Studies that have been undertaken have shown that urban lakes are typically dominated by cyanobacteria (e.g. Mayer *et al.* 1997; Dokulil & Teubner 2000) and often provide suitable habitat for rotifer communities (Ejsmont-Karabin & Kuczynska-Kippen 2001). The hydraulic residence time of urban lakes strongly influences their phytoplankton communities (Olding *et al.* 2000).

The relatively dense populations that often surround urban lakes utilize them to a high degree and the lakes provide vital contact to the general public (Birch & McCaskie 1999). Urban lakes can also provide ideal areas for research and education However, there is a recognized dearth of knowledge on urban ecology (Collins *et al.* 2000) and urban lakes are one of the least understood areas of limnology (Birch & McCaskie 1999).

FISH IN SHALLOW LAKES

Shallow lakes are usually dominated by benthic fish (Scheffer 1998), which have been linked with declines in macrophyte populations and reductions in water quality (Williams *et al.* 2002). Benthic fish can also cause a "catastrophic response" (Zambrano *et al.* 2001) in the water quality of shallow lakes if they exceed a critical biomass threshold. Increased foraging activities as their prey decline can cause an increase in sediment resuspension and a decline in water quality. Shallow lakes are particularly susceptible, whereas the lower benthic productivity of deep lakes limits the impact that this can have on them (Zambrano *et al.* 2001). Also, there are relatively more fish in shallow lakes than deep lakes. Fish biomass has been negatively correlated with mean depth, with a 10-fold decrease in depth resulting in a 48-fold increase in fish biomass, although this may be due to the different nutrient states of the lakes analysed (Jeppesen *et al.* 1997). There is a clear linear relationship between the biomass of benthic fish and the concentration of inorganic suspended solids (Meijer *et al.* 1994). Some benthic fish species can significantly increase the nutrient loading of a water body, even in the absence of sediment (Keen & Gagliardi 1981).

LOCATION OF THE CAMPUS LAKES

The three lakes studied in this thesis are located on the campus of the University of Waikato (S37° 47’ E175°19’), Hamilton, New Zealand. The lakes’ range in area from 4400 m^2 to 10100m^2 (Table 1.1).

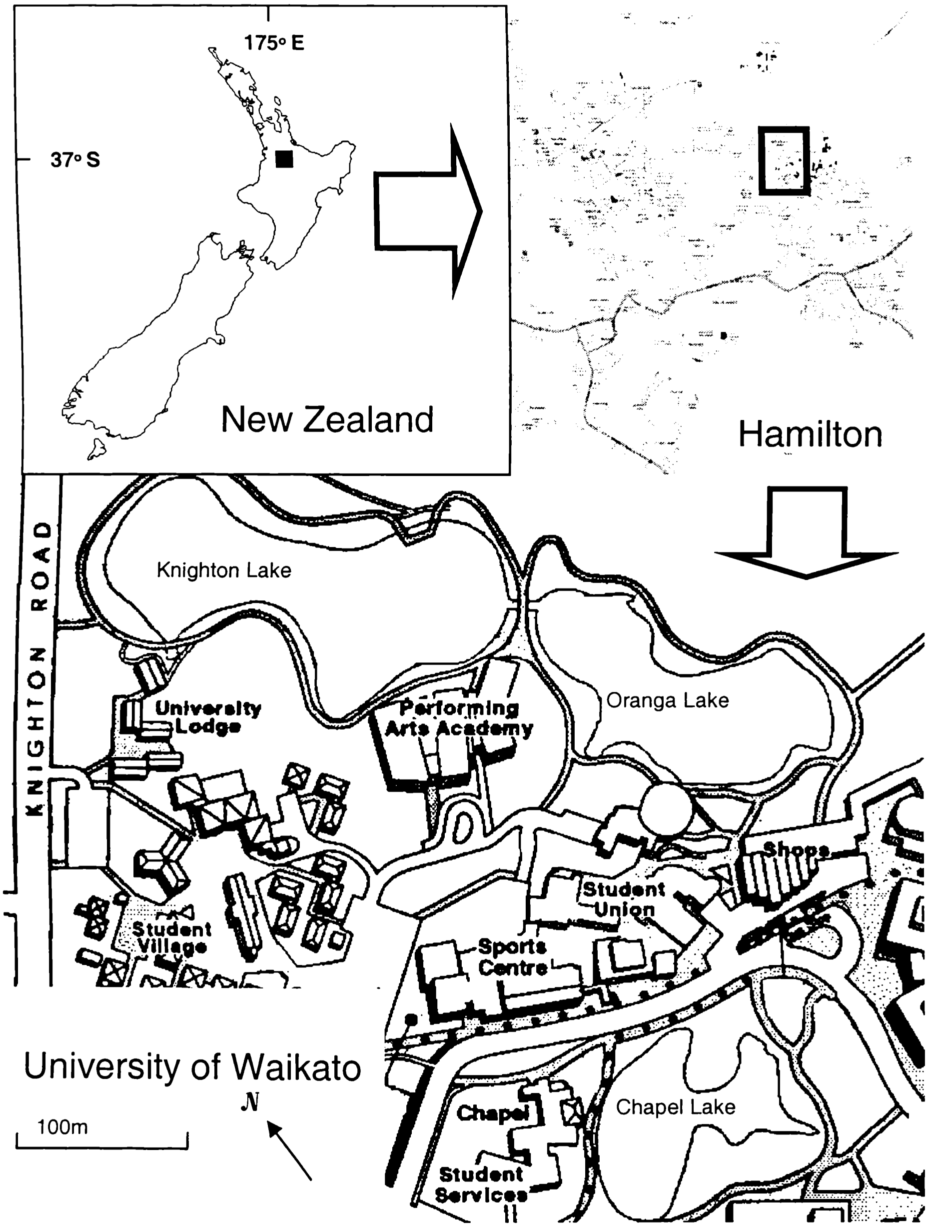


Figure 1.1: Location of the University of Waikato campus lakes.

Table 1.1: Physical characteristics of The University of Waikato campus lakes (from Willis, 1996).

	Chapel	Oranga	Knighton
Surface area (m^2)	4400	6900	10100
Maximum depth (m)	1.8	0.7	0.7
Water volume (m^3)	5200	4100	6060

AIMS AND STRUCTURE

This thesis deals with three lakes that are situated on the University of Waikato campus, Hamilton. Its objectives were to investigate: a) the mixing regimes, b) the seasonal distribution and abundance of water quality variables, phytoplankton, zooplankton and fish, and c) the physical and chemical parameters in the lakes. Each chapter is intended to be published as an individual paper, so some repetition of material is inevitable.

The aims of each chapter are:

Chapter 2: To determine any differences in mixing regimes between the lakes and to discuss possible relationships between climate and nutrient levels in the lakes.

Chapter 3: To determine any differences in the water quality, phytoplankton and zooplankton of the lakes and to identify any correlations between plankton communities and environmental variables.

Chapter 4: To determine any differences in the benthic fish communities of the lakes and to document aspects of their ecology in shallow lakes. .

Chapter 5: To summarise Chapters 2, 3 and 4 in relation to the lakes' individual morphology and mixing regimes, with a view for future lake management and restoration.

Chapter 2

Mixing Regimes, Climate and Nutrient Regimes

INTRODUCTION

The influence of mixing on shallow lakes

One of the defining features of shallow lakes is their mixing regime. Shallow lakes are prone to intense mixing (Nixdorf & Denke 1997) and this may contribute to their susceptibility to eutrophication. Intense mixing can negate the effects of a reduced nutrient load as sediment-bound nutrients can become released into the water column and form a significant portion of the total nutrient budget. Lakes that are shallow and intensively mixed have much higher surface phosphorus levels than deeper lakes, which experience mixing through a smaller fraction of their volume (Osgood 1988). Resuspension of bottom sediments in deep lakes has been shown to occur below wave bases during unstratified periods (Hawley *et al.* 1996) and also occurs in shallow water bodies as a direct result of wind induced mixing (Yanful & Catalan 2002). In lakes < 6 m deep, the likelihood of sediment resuspension decreases markedly with increasing lake depth and increases with higher wind speeds (Scheffer 1998).

The mixing regime can affect the plankton populations of shallow lakes. Intense mixing reduces the likelihood that algal populations will settle out of the water column or become light limited. Also, very shallow lakes often have a relatively good underwater light climate for phytoplankton, with light penetration throughout the entire water column. This can result in increased plankton production compared to deeper lakes (Nixdorf & Denke 1997). Mixing can increase the relative abundances of total chlorophyll, total autotrophic picoplankton (Rhew *et al.* 1999), and of autotrophic and heterotrophic protists (Garstecki & Wickham 2001). The shallow, frequently mixed Lake Opinicon was compared to the deeper dimictic Upper Rock Lake and found to contain denser populations of zooplankton, despite similar water chemistry (Agbeti *et al.* 1997). When buoyant cyanobacteria were present, vertical and horizontal mixing distinctly increased the productivity of algae. Vertical mixing has been found to result in a greater increase in primary production than horizontal mixing (Gervais *et al.* 1997). Intense mixing can often alter the limiting factor of algal populations and the degree of mixing can influence the community composition. In reservoirs with high inorganic turbidity, the ratio between the euphotic depth and the mixing depth was more instrumental in controlling overall productivity than the nutrient status of the water body (Grobbelaar 1992). The strength of wind-induced resuspension also has the potential to greatly influence the makeup of food webs in shallow water bodies through the preferential resuspension of smaller, lighter algae (Arfi & Bouvy 1995).

Strong mixing can also influence the resuspension of contaminants into the water column. Sediment resuspension has been found to release as much as 20% of arsenic contained in the sediment back into the water column (Linge & Oldman 2002). In one shallow lake, 43% of the suspended matter leaving the system originated from the polluted lake bottom (ten Hulscher & Lüers 1992). Wind-induced mixing can cause resuspension of contaminated mine tailings if the overlying water depth is less than 1 m. The critical depth for sediment resuspension in a large shallow tailings pond, with an average wind speed of 10 m s^{-1}, was estimated to be between 1.18 and 1.34 m (Yanful & Catalan 2002).

Nutrients in shallow lakes

The concentrations of nutrients in shallow lakes can vary temporally (e.g., Miller 2002) and spatially (Scheffer 1998). There is a general trend of increasing nutrient levels in lakes (Moss 1998), particularly shallow, urban lakes (Birch & McCaskie 1999). This can manifest itself in algal blooms and decreased water quality.

Shallow lakes are more prone to external nutrient loading due to their small volume (Wetzel 2001) and can often experience significant internal loading, in spite of reductions in their external nutrient load (e.g., Kleeburg & Dudel 1997). The reactions of various factors at the sediment-water interface control the degree of internal recycling of nutrients. There is a wide range of factors that affect nutrient release and their interactions are often very complex. Their relative influence also differs between deep and shallow lakes (Marsden 1989) and between polymictic and dimictic lakes (Osgood 1988). There has been a wide range of P release rates recorded in lakes (Marsden 1989) but less is understood about mechanisms of nutrient release in shallow lakes than hypolimnion release in deeper lakes (Phillips *et al.* 1994).

Phosphorus can be immobilized by binding with iron compounds in sediments (Jensen *et al.* 1992). However, if the oxygen levels in the overlying water fall to a level below the amount that is being used by the microbes in the sediment then anoxic conditions are created and phosphorus cannot be bound by iron (Scheffer 1998). Under anaerobic conditions the ratio of iron to phosphorus in the sediments is important in governing the rate of sediment P release. Iron is able to bind approximately 10% of its own volume of phosphorus in aerobic conditions. If the iron to phosphorus ratio is less than 15 and the sediment surface oxidised then phosphorus release is unlikely (Jensen *et al.* 1992). However, the presence of H_2S can negate this; H_2S can inactivate iron through the formation of FeS and FeS_2, and P release is then enhanced (Kleeburg & Dudel 1997).

The aerobic release of phosphorus from the sediments is governed by processes that can be very complex although it is usually reliant on water turbulence (Marsden 1989). Shallow lakes are prone to intense mixing (Nixdorf & Denke 1997), which has two opposite effects on the nutrient release It limits the degree of anaerobic phosphorus release by providing sufficient oxygen to the sediment-water interface but it encourages the aerobic release of sediment-bound phosphorus through resuspension (Scheffer 1998).

Intense mixing in shallow lakes usually provides enough oxygen to the sediments to allow the formation of an aerobic layer (Phillips *et al.* 1994; Scheffer 1998). Consequently, aerobic phosphorus release is the most dominant form of internal loading in shallow, non-stratifying lakes (Marsden 1989). Mixing can resuspend bottom sediments in both shallow (Scheffer 1998) and deep lakes (Hawley *et al.* 1996). Disturbed sediment can release as much as 30 times more phosphorus than undisturbed sediment (Søndergaard *et al.*1992). Wind-induced resuspension of sediment can affect nutrient levels, especially in shallow lakes. The likelihood of 50% of the bottom sediments becoming suspended increases with decreasing lake depth. Wind-induced resuspension of sediments can often explain differing nutrient levels in shallow lakes (Scheffer 1998).

pH can influence the amount of phosphorus release from sediments. High pH values can cause phosphorus to be released from iron. High pH values have been associated with high levels of phosphorus in lakes (Marsden 1989).

The biota of a water body can also influence the nutrient release rate. Higher temperatures often increase biological activity levels, which can increase phosphorus release rates (Kleeberg & Dudel 1997). Phosphorus release from sediments has also been linked with the biomass of benthic *Microcystis* colonies in some eutrophic shallow lakes (Brunberg & Boström 1992). High abundances of the macroinvertebrate *Nereis* can also influence sediment phosphorus fluxes as a result of their burrowing (Clavero *et al.* 1992).

The release of sediment-bound phosphorus can make up a significant part of shallow lakes' nutrient budgets (e.g., Knuuttila *et al.* 1994; James *et al.* 2002). Shallow lakes are less likely to recover after external nutrient reductions than comparable deeper lakes (Moss *et al.* 1997; Graneli 1999). Shallow lakes are more unpredictable in their phosphorus release and significant inter-annual variations are common. Meteorological conditions can strongly influence this variation (Marsden 1989).

It is less common for nitrogen to be a limiting nutrient in shallow lakes than phosphorus; however it can still influence shallow lake ecosystems (Scheffer 1998). When organic material falls out of the water column, bacteria, and protozoa degrade it and ammonium is released. Under aerobic conditions some nitrate is also formed (Moss 1998). The bottoms of shallow lakes often contain thick layers of nutrient-rich sediment. The nitrogen in the sediment is usually oxidized from organic N to NH_4^+ to NO_3^- and NO_2^-. If there is no oxygen present then the reaction stops at NH_4^+. If anoxic conditions occur in lakes then NH_4^+ is released from the sediment. It can often build up to toxic levels, especially at high pH values (Sprent 1987).

The main processes of internal nitrogen cycling in shallow lakes are given in Figure 2.1.

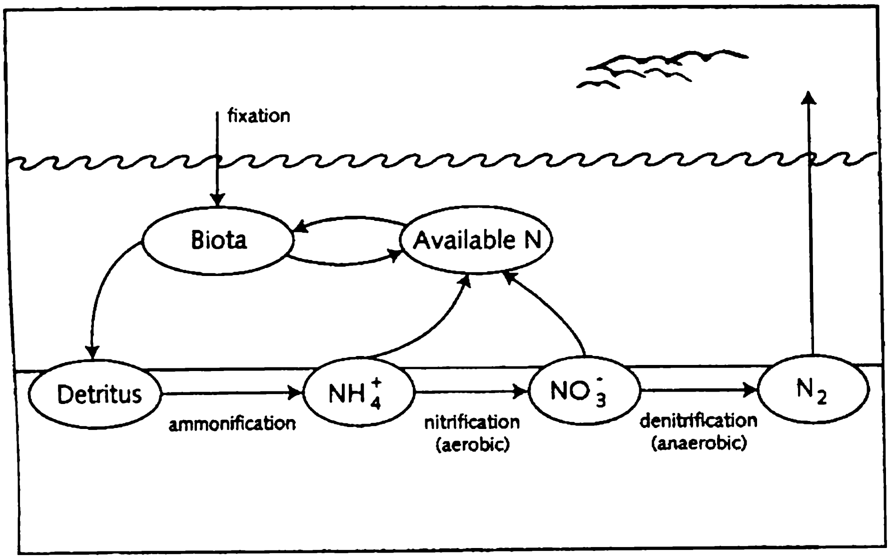


Figure 2.1: The main processes involved in internal nitrogen cycling in shallow lakes (taken from Scheffer 1998, p61)

Chapter aims

The aims of this chapter are:

1. To determine the climatic conditions during the study period.
2. To determine the mixing regimes of the lakes.
3. To determine the temporal variation of nutrient levels in the lakes.
4. To determine possible causes for variations in nutrient levels in the lakes.

METHODS

Site description

The campus lakes are small, shallow, hypertrophic, urban lakes (Willis 1996) that were created in the early 1970s. They are situated on the University of Waikato campus, Hamilton. The three lakes (Chapel, Oranga and Knighton) were originally intended "to aid site drainage, to be a visual amenity, and to be landscaped for botanical interest and recreational use" (Pridmore & Burns 1989). The lakes drain the University grounds to the Hamilton stormwater system and eventually the Waikato River (Pridmore & Burns 1989). The three lakes range in size from 4,400 m^2 to 10,100 m^2 (Willis 1996) and can also function as research sites for teaching at the University of Waikato, as habitats to species of native flora and fauna and for recreational fishing. Two of the lakes – Knighton and Oranga - are uniformly shallow (maximum 0.7 m depth) while Chapel Lake has two deeper basins (maximum 1.8 m depth) with a shallower section between. The lakes are fed by groundwater and also receive stormwater runoff from the University grounds. All lakes are physically very close (within 300 m), in the same soil type and are subject to the same climatic influences (Pollard 1995).

Background to lakes

Knighton and Oranga lakes have had long-standing issues with low water quality; mainly caused by summer blooms of *Microcystis* spp. and *Anabaena* spp. (Willis 1996) and the resuspension of bottom sediments through temperature and wind induced mixing of the water column. The two lakes are characterised by turbid water with low Secchi transparencies, high algal biomass and heavy blooms of the curly leaved pond weed *Potamogeton crispus*. Knighton Lake also borders the newly established Academy of Performing Arts and unsightly algal blooms detract from the value of this facility. Oranga Lake, however, is closer to the center of the university and is therefore seen regularly by large numbers of people. Chapel Lake is the deepest and generally has the best water quality. It has higher Secchi transparencies, less severe algal blooms in summer, lacks *Potomogeton crispus*, and is seen as the most attractive lake (Pollard 1995).

The fish present in the lakes include large populations of brown bullhead catfish (*Ameiurus nebulosus*), shortfinned eel (*Anguilla australis*) and goldfish (*Carassius auratus*), common bully (*Gobiomorphus cotidianus*) and mosquitofish (*Gambusia affinis*). The lakes provide habitat for large numbers of birds, predominantly ducks (*Anas* spp.) and a small number of shags (*Phalacrocorax* sp.). There are marginal stands of rushes (*Baumea articulata*) in Chapel Lake and water lilies (*Nymphea odorata*) in Oranga and Chapel lakes.

Sampling regime

Water was sampled over 14 months, from July 2001 to October 2002. Sampling frequency varied (Table 2.1). Two sites per lake were sampled (Figure 2.2) and lakes were always sampled in the same order; Knighton Lake, Oranga Lake then Chapel Lake. All sampling was conducted between 0900 and 1200 on the day of sampling.

Table 2.1: Sampling regime for the University of Waikato campus lakes.

Sampling period	Approximate sampling frequency	Sampling dates
July 2001- February 2002	Monthly	12/7/01, 9/8/01, 11/9/01, 8/10/01, 6/11/01, 11/12/01, 11/1/02, 25/1/02
February 2002 - April 2002	Fortnightly	20/2/02, 15/3/02, 25/3/02
April 2002 – July 2002	Weekly	5/4/02, 12/4/02, 18/4/02, 30/4/02, 10/5/02, 22/5/02, 31/5/02, 14/6/02, 21/6/02, 28/6/02, 5/7/02
July 2002 - October 2002	Fortnightly	18/7/02, 2/8/02, 16/8/02, 30/8/02, 4/10/02

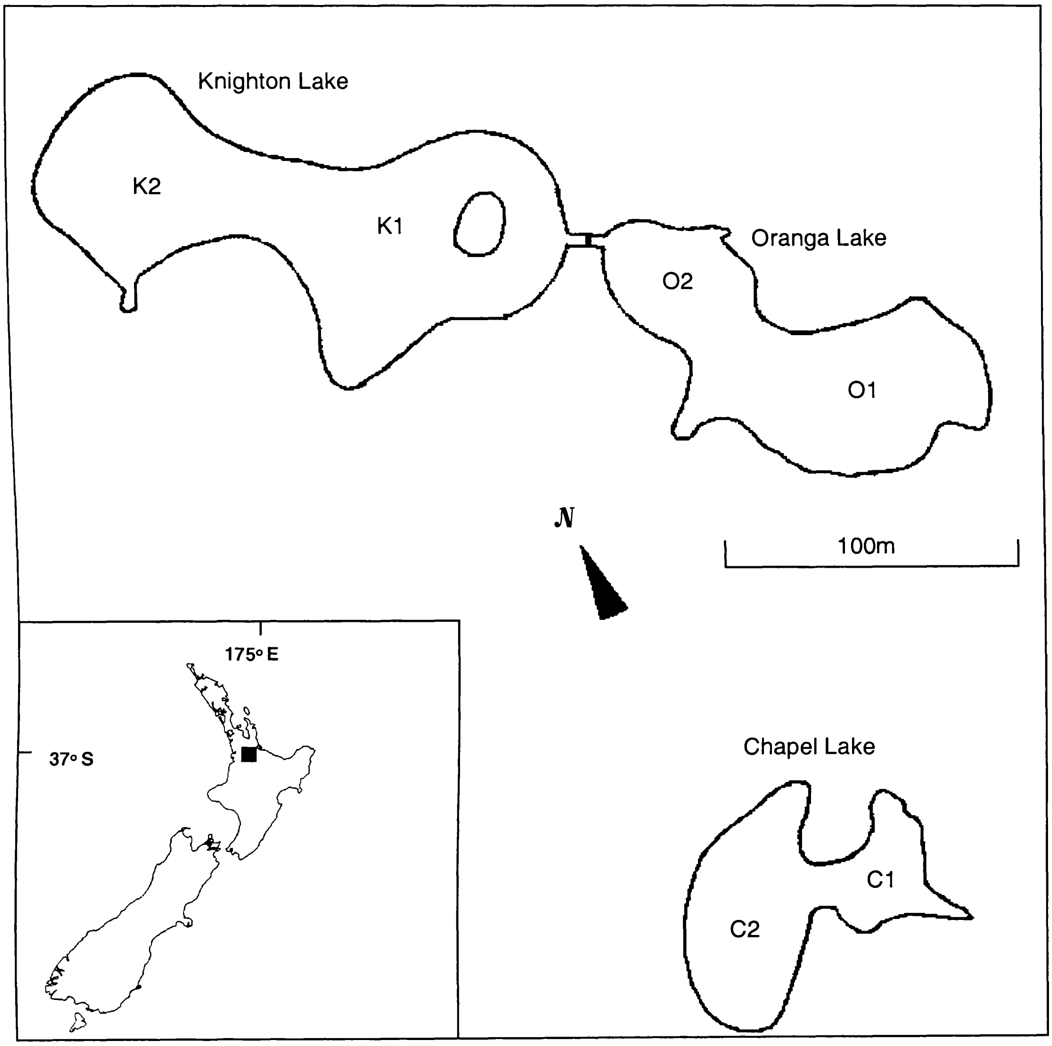


Figure 2.2: Sampling sites in the University of Waikato campus lakes.

Climatic data.

Hourly climate data were obtained from the Ruakura climate station, located approximately 1 km to the north of the lakes.

Stability

Water temperature was recorded at approximately 5 cm below the surface and approximately 10 cm above the bottom of the lakes on an hourly basis, for approximately one year. Onset Corporation, Tidbit temperature loggers were used and data were downloaded every 3 months.

The Brunt Väisälä equation was used to estimate the stability of the water column. It is commonly used as it can be easily calculated using temperature data (Viner 1985). The formula is:

$$N^2 = (g/q')(\mathrm{d}q/\mathrm{d}z)$$

where $(\mathrm{d}q/\mathrm{d}z)$ = mean water density gradient between the surface and bottom waters; q' = mean density given by $(q_{\mathrm{surface}} + q_{\mathrm{bottom}}/2)$ and g = gravitational acceleration (Nixdorf 1994).

Surface water samples - for nutrient analyses were taken by filling a 1-litre container at each site. Containers were acid washed in a 10% HCl solution, filled with water that had undergone reverse osmosis, and put on ice prior to sample collection. After water had been collected, samples were stored on ice in the dark until filtration could occur. The samples were filtered and frozen on the same day as collection, with the exception of samples taken before December 2001, which were frozen prior to filtering. PO_4, NH_4, NO_2, and NO_x were analysed using a Lachat QuickChem FIA 8000 series auto-analyser. Oxygen was measured in the lakes at 20 cm depth intervals using an YSI Model 50 DO meter.

RESULTS

Mixing

The mixing regimes of Chapel Lake were significantly different to the mixing regimes of Oranga and Knighton lakes. Although all three lakes showed strong diurnal variations in stability, there were extended periods when the water column was relatively stable in Chapel Lake, which did not occur in Knighton and Oranga Lakes. There were also brief periods in Oranga and Knighton lakes where the surface water was denser than the water below it, indicating very intense mixing. The lakes showed stronger daytime stratification in summer than winter (Figure 2.3). The lakes were less stable at night than during the day (a high N^2 value indicates periods of relative stability) (Figure 2.4). On average, Chapel Lake was significantly more stable than Oranga and Knighton lakes. Oranga Lake was significantly more stable than Knighton Lake ($P < 0.001$) (Figure 2.5).

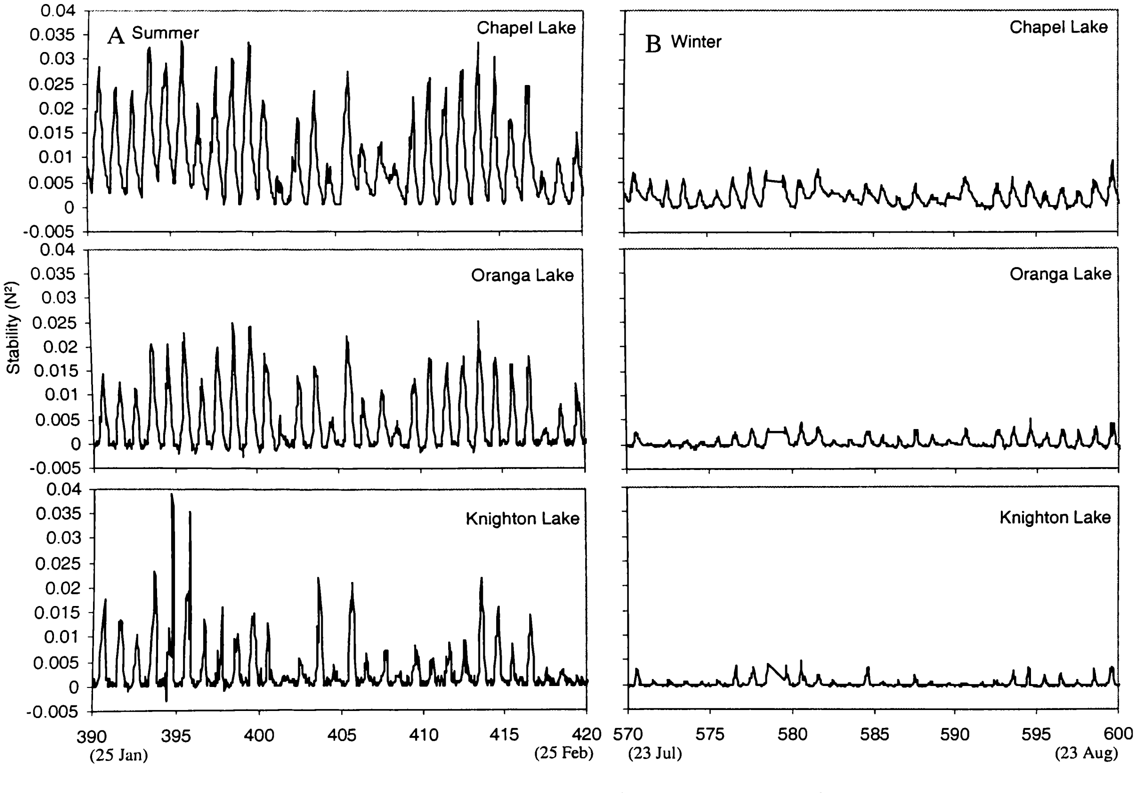


Figure 2.3: Typical Brunt Väisälä stabilities in the University of Waikato campus lakes in **A**. summer and **B**. winter.

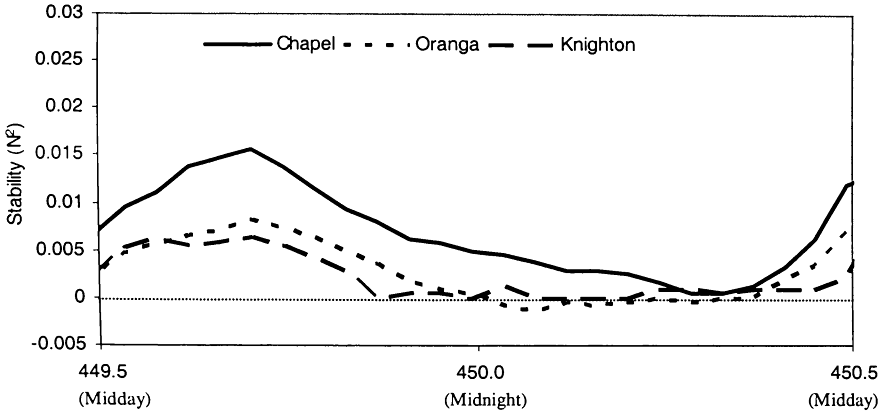


Figure 2.4: Typical Brunt Väisälä stabilities in the University of Waikato campus lakes over a 24-hour period.

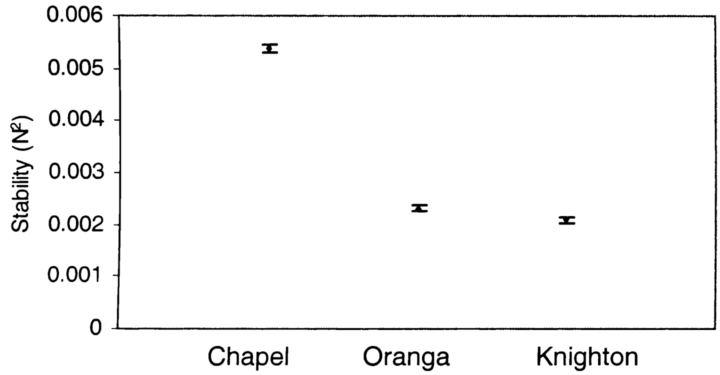


Figure 2.5: Average Brunt Väisälä stability in The University of Waikato campus lakes between July 2001 and October 2002, calculated from hourly temperature data. Error bars are ± 1 S.E.

Climate

The pooled climatic data are given in Table 2.2. There were marked diurnal variations in wind speed and air temperature over the sampling period. Air temperature followed a seasonal trend with hotter temperatures in summer and cooler temperatures in winter. There were several intense rainfall events and there was generally more rainfall over winter than summer. Most wind came from 90° to 270° (E, S and W) (Figure 2.6).

Table 2.2: Climatic data at the Ruakura climate station between July 2001 and October 2002.

	Wind speed ($m\ s^{-1}$)	Air temperature (°C)	Rainfall ($mm\ day^{-1}$)	Radiation ($MJ\ m^{-2}$)
Mean	2.6	13.8	2.9	0.5
Maximum	14.3	26.5	34.0	3.9
Minimum	0.0	-1.4	0.0	0.0

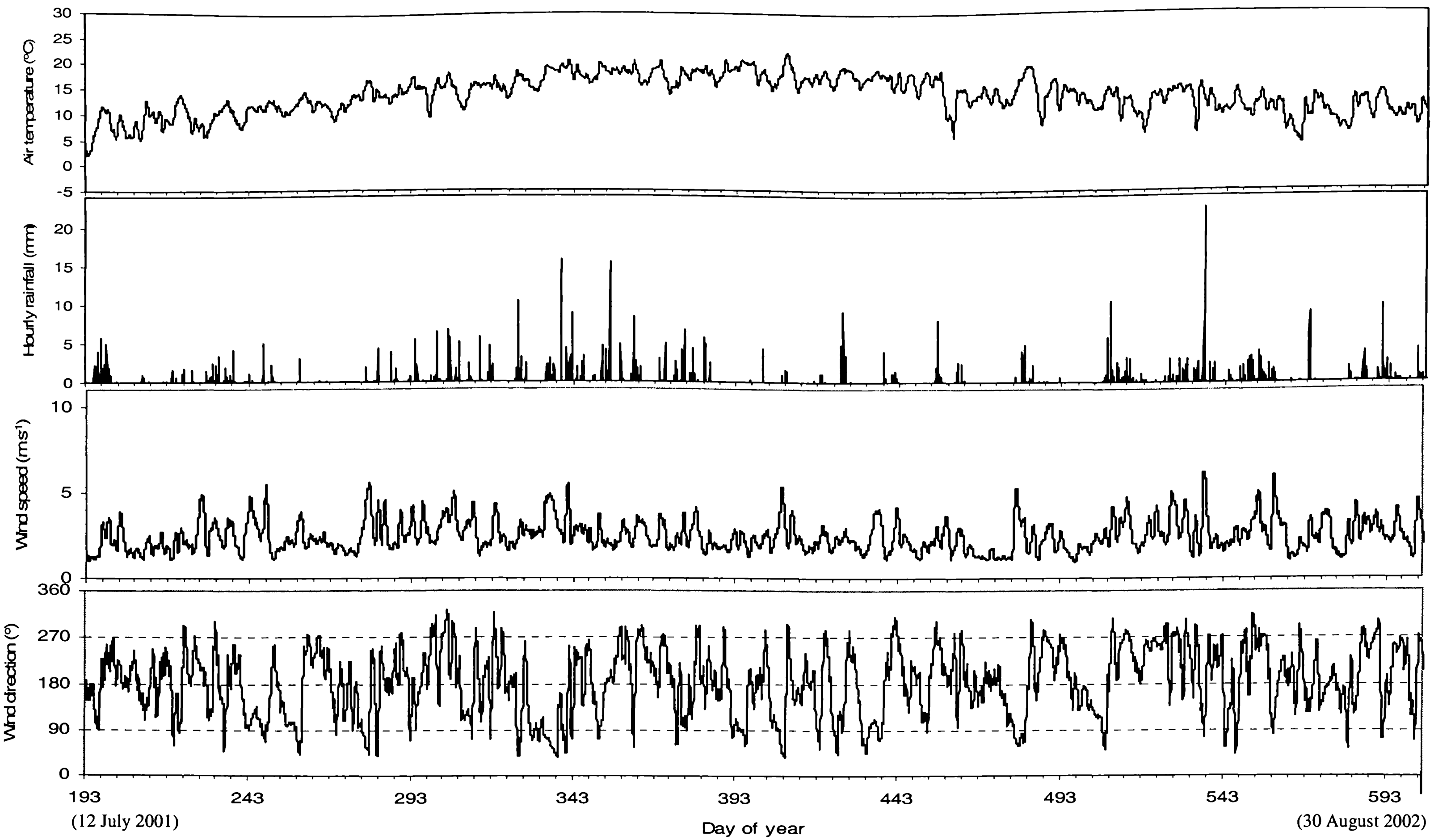


Figure 2.6: Hourly air temperature, rainfall, wind speed, and wind direction at the Ruakura climate station between 12 July 2001 and 30 August 2002. Air temperature, and wind speed and direction data have been 24-point smoothed.

Nutrients

The concentration of nitrogen in its various forms was low for most of the year. However there were pronounced peaks in the ammonium concentrations in Chapel and Knighton lakes (Figure 2.7), followed by peaks in NO_x and NO_2 (Figure 2.8A & 2.7B). This coincided with periods of low oxygen in the lakes. Although there were comparable periods of low oxygen, there was no ammonia peaks recorded in Oranga Lake (Figure 2.7).

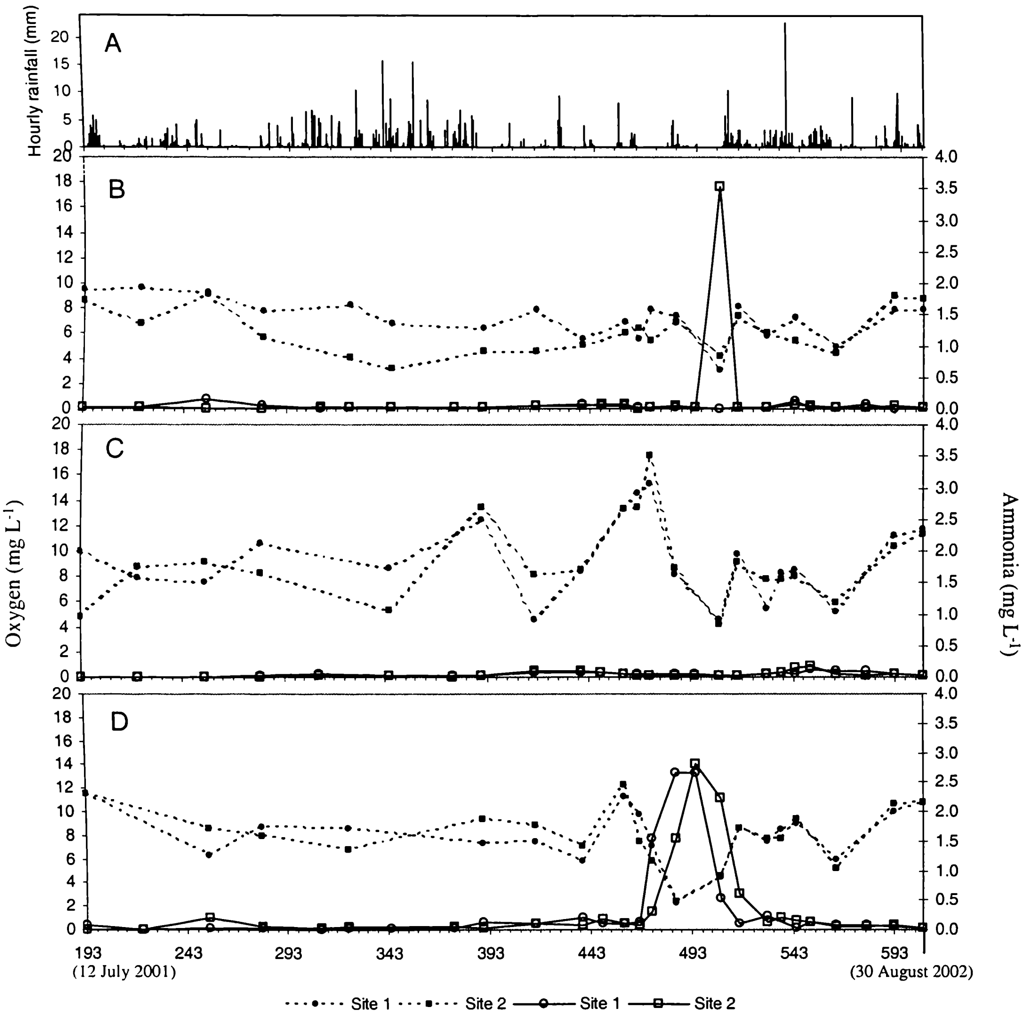


Figure 2.7: **A.** Hourly rainfall at the Ruakura climate station; concentrations of dissolved oxygen in the bottom 20 cm of the water column and ammonia in the surface water in **B.** Chapel **C.** Oranga and **D.** Knighton lakes. - - - - = Oxygen; —— = NH_4^+

The phosphate concentrations in all three lakes varied over the sampling period. There were generally low levels in the lakes until February 2002 where the levels increased markedly. After this the phosphate levels in the lakes alternated between periods of high and low concentrations. The phosphate concentrations matched each other closely in the lakes (Figure 2.8C).

There was no direct correlation between the nutrient concentrations and the mixing regime, rainfall, wind speed, or air temperature affecting the lakes.

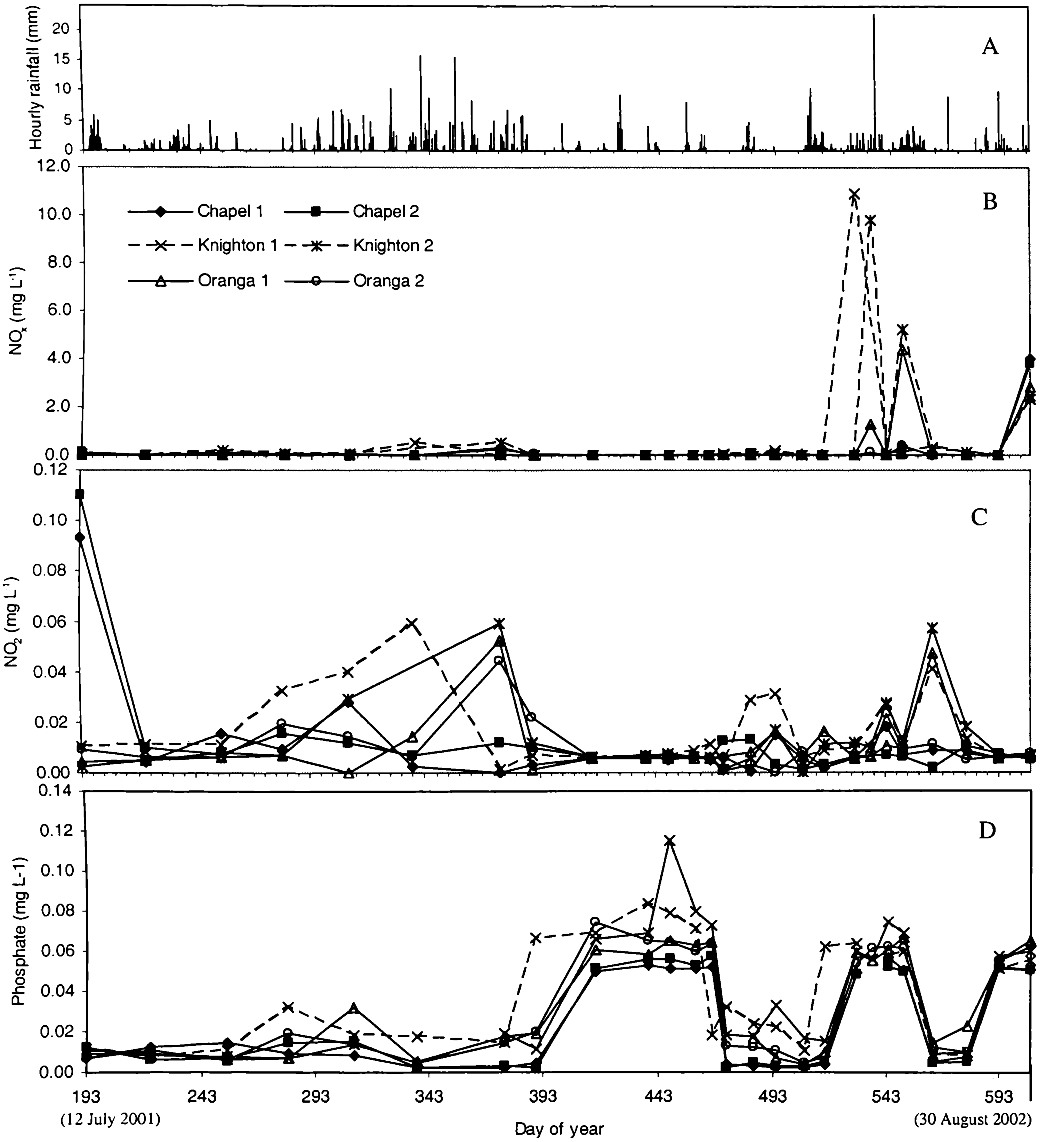


Figure 2.8: **A**. Hourly rainfall at the Ruakura climate station and concentrations of **B**. NO_2, **C**. NO_x, and **D**. PO_4 in the University of Waikato campus lakes.

DISCUSSION

Mixing

The mixing regimes differed in all three lakes. This was most likely due to the different morphologies and depths of the lakes. The shallower lakes with flat bottoms (Oranga and Knighton) were often mixed throughout their entire depth whereas the deeper, basin shaped Chapel Lake only experienced brief periods of complete mixing. Chapel Lake also experienced extended periods when the water column was relatively stable. Oranga and Knighton lakes were unstable during this time. The water in all three lakes was less stable during the night than the day.

The stability of the lakes' water columns was linked with the temperature regimes of the lakes. During the day the surface waters heated up at a faster rate than the bottom waters and the water column was relatively stable. During the night, the surface waters lost heat faster than the bottom waters. The surface water continued to lose heat until it became denser than the water below it. The surface water then sunk, forcing the bottom waters up, which caused intense mixing. This occurred for up to five hours each night in summer in the shallower lakes. In summer, the water in the lakes was more stable during the day than in winter. There was a greater difference in temperature between the surface and bottom waters, caused by more intense heating. This was not as pronounced in winter as the heating of the water surface did not occur to the same degree.

Climate

The climate affecting the campus lakes during the study period was not extraordinary for the Waikato. There was slightly less average daily rainfall over the study period than mean values of 3.2 – 4.3 mm day^{-1} recorded in the Waikato (Brestrum 1998). There were several intense rainfall events and there was generally more rainfall over winter than summer, which is common in the North Island (Duncan 1987). The majority of wind speeds recorded were between 0.5 and 5 m s^{-1}, which agreed with other recorded values (Brestrum 1998). The mean air temperature was also within the expected range of 12.6 - 15°C (Brestrum 1998).

Nutrients

The changes in nutrient levels in all three lakes were very closely matched to each other, in spite of their differing depths, morphologies, mixing regimes and plant and animal communities. These data suggests that either an external factor or internal processes that simultaneously affect all three lakes cause the changes.

The ammonia levels in the lakes peaked in late autumn. This was not likely to be caused by rainfall. 1.8 mm of rain fell in the lakes' catchment in the week prior to the ammonia peak in Knighton Lake. This volume is well below the average evaporation rates of ~ 3 mm day^{-1}. 17.8 mm of rain fell in the week prior to the ammonia peak in Chapel Lake. Therefore, approximately 2000 m^3 fell over its entire catchment (111,300 m^2). Even if no evaporation occurred during this time and the entire volume of water made it into the lakes, the runoff would have to contain an ammonia concentration of approximately 9180 mg m^{-3} to produce the peaks seen in Chapel Lake. As this is more than twenty times higher than the maximum NH_4^+ concentrations recorded in runoff from the adjacent Hillcrest catchment (380 mg m^{-3}) (Williamson 1985) it is highly unlikely. The ammonia is likely to have come from another source. This source is likely to be the sediment.

Organic nitrogen is present in plant litter and organic wastes. Ammonia is formed from the breakdown of organic nitrogen. If there is oxygen present then nitrification occurs and nitrate is formed. However, in the absence of oxygen, nitrogen cannot be converted to other forms and is released as ammonia from the sediment (Lijklema 1994). The ammonia peaks in the campus lakes occurred when there were low dissolved oxygen levels in the bottom waters. It is possible that anoxic release from the sediment is causing the peaks of ammonia in some sites in the campus lakes. Peaks of NO_2 and NO_x followed the peaks of ammonia in the lakes. It is likely the ammonia was converted to these compounds once the oxygen levels reached a sufficient level. The presence of NO_2 between July and December 2001 in the absence of any other forms of nitrogen raises questions about the validity of these results. For reasons that are not apparent, ammonia peaks were not observed in Oranga Lake.

Atmospheric deposition of nitrogen – usually in the form of ammonia - can be significant in areas of high energy consumption or animal breeding (Lijklema 1994). It can be an important source of nutrients in some New Zealand lakes (Rutherford *et al.* 1987). However, it is not likely that atmospheric deposition influenced the nitrogen peaks in the campus lakes. If the average New Zealand deposition rates of NH_4-N from rainfall (85 g m^{-3} yr^{-1}) (Rutherford *et al.* 1987) occurred over the campus lakes then 1 - 2 mg m^{-3} day^{-1} of nitrogen would be deposited directly into the lakes in the week preceding the ammonia peaks. This is well below the peak concentrations recorded in the lakes.

It is possible that rainfall influenced the variations in the phosphate levels. Nutrient peaks are often associated with intense rainfall events (Collier *et al.* 1995) and rainfall was one factor that influenced all three lakes equally and simultaneously. However, the peaks in phosphate did not always coincide with intense rainfall events. There were some periods (e.g., February 2002) when the phosphate was high in the lakes but there was very little rainfall. Conversely, there were periods of reasonably heavy rainfall that did not translate into high concentrations of phosphate in the lakes. It is possible that there was a lag time between rainfall events and high phosphate concentrations. This lag would be less pronounced in winter when the soil would be more saturated and runoff would flow faster into the lakes. More detailed studies are needed to clarify this process.

The wind speed varied throughout the study period. There were no significant correlations between wind speed or wind direction and nutrient concentration in the lakes. Also, Chapel Lake is deeper and more sheltered than the other two lakes; yet it experienced similar changes in nutrient concentrations. It is unlikely that the wind was the main driver of the nutrient regimes in the campus lakes.

CONCLUSIONS

The campus lakes have different morphologies and experienced significantly different mixing regimes as a result. The stability of the water columns in all three lakes was controlled by diurnal changes in air temperature. The lakes were generally less stable at night than during the day and were less stable in winter than in summer. On average the shallower lakes (Oranga and Knighton) were less stable than Chapel Lake, which is slightly deeper.

The climate that affected the lakes during the study period was not extraordinary with slightly below average rainfall but average temperature and windspeed.

Nutrients concentrations were similar in all three lakes. It is possible that low levels of oxygen allowed ammonia to be released from the sediment and cause increases in its concentration in the surface waters. Ammonia may then have been converted to NO_2 and NO_x as oxygen levels rose. The exact mechanisms causing the changes in concentrations of PO_4 are unclear. There were no direct correlations between the climate and the nutrient concentrations in the lakes.

Chapter 3

Water Quality, Phytoplankton and Zooplankton

INTRODUCTION

Although shallow lakes provide a range of benefits to human populations (Perrow *et al.* 1999; Gulati & van Donk 2002), studies have traditionally focused on deep, stratified lakes, at the expense of research on shallow lakes (Kufel *et al.* 1997). It is only in the last decade or so that shallow lakes have been more intensively studied and their ecological and physical complexities appreciated (Phillips *et al.* 1999; Havens *et al.* 2001). Shallow lakes are common throughout the world (e.g., Jeppesen 1998) and make up the majority of the lake types within the Waikato region (Barnes 2002). They are particularly prone to eutrophication. They are more prone to nutrient loading than deep lakes due to their small volume (Wetzel 2001). Also they often experience high internal nutrient loading through sediment resuspension (Søndergaard *et al.* 1990; Phillips *et al.* 1994) that can manifest itself in phytoplankton and zooplankton blooms and reduced water quality.

Ecology of shallow lakes

Shallow lakes have a unique biota. The primary producers are often algal-dominated with cyanobacteria dominating the phytoplankton community (Wiedner & Nixdorf 1998; Gulati & Van Donk 2002; Wiedner *et al.* 2002). Oligochaetes often dominate the benthic marcroinvertebrate community and there is typically a small submerged macrophyte population. These assemblages can decrease the phosphorus settling rate, which can exacerbate the effects of eutrophication (Havens & Schelske 2001), although this is not always the case (Beklioglu *et al.* 1999).

Cyanobacteria are common in eutrophied urban lakes (e.g., Mayer *et al.* 1997; Beyruth 2000; Dokulil & Teubner 2000) and their abundance is increasing near some urban centres (Azevedo et al. 1994). They are often associated with decreased water clarity and cyanotoxin occurrences (e.g., Azevedo *et al.* 1994; Wiedner *et al.* 2002) and in some shallow lakes, and they can make up 60 - 90% of the total microbial biomass in summer (Brunberg & Boström 1992). In shallow urban lakes, the dominance of cyanobacteria has been linked with the hydraulic residence time (Olding *et al.* 2000).

Cyanobacterias' adaptations to low levels of light, their nitrogen fixing capacity and their ability to alter their photosynthetic rate in response to the mixing regime provides them with advantages over other forms of algae in shallow lakes (Kronkamp *et al.* 1992; Nixdorf & Deneke 1997). Their ability to withstand large variations in their phosphorus content (Gulati & van Donk 2002) and their buoyancy control (Bonnet & Poulin 2000) also contribute to their success.

Overseas, shallower lakes have been found to have higher densities of zooplankton than deeper lakes of similar water chemistry (Agbeti 1997). Shallow lakes are often dominated by rotifers and small cladocerans (Jenkins 1995; Liu *et al.* 2002), whose populations can show marked seasonal trends in community composition and abundance (Miller 2002; Romero *et al.* 2002). Zooplankton can be instrumental in maintaining clear-water states in shallow lakes; however their influence can vary between lakes.

Recent zooplankton research in New Zealand is limited (e.g., Chapman & Green 1999; Greenwood *et al.* 1999; Duggan *et al.* 2001; Duggan *et al.* 2002; Burger *et al.* 2002). There is generally low species diversity (Chapman & Green 1987) although this may be due to a lack of intensive studies (Duggan *et al.* 2002). Rotifers are the least studied of the New Zealand zooplankton (Chapman & Green 1987) despite their dominance of some zooplankton communities (e.g., Burger *et al.* 2002) and their linkages with trophic state gradients in lakes (Duggan *et al.* 2002). There is often a relatively low zooplankton to phytoplankton ratio in lakes (Malthus & Mitchell 1990). Small cladocerans are common in some shallow lakes in New Zealand (Etheridge 1987, Greenwood *et al.* 1999)

Urban lakes

Urban lakes are often shallow, eutrophic and have degraded water quality (Birch & McCaskie 1999; Moss *et al.* 2002) due to introduced species and the urbanization of their catchments. Although they may be of limited conservation value, urban lakes are of particular importance in lake management and education due to their high accessibility, recreational value and dense surrounding population. They can also provide excellent areas for research. However, less is known about them than any other type of lake (Birch & McCaskie 1999).

Chapter aims

This chapter compares and contrasts the water quality and phytoplankton and zooplankton communities of three small, shallow, hypertrophic, urban lakes that are situated on the University of Waikato campus, Hamilton. Correlations between species abundances and environmental variables are analysed.

The aims of this chapter are:

1. To determine the spatial and temporal differences in water quality and the phytoplankton and zooplankton communities within the campus lakes
2. To determine the associations between environmental factors and plankton community composition and whether these differ between lakes.

METHODS

The site description, background to the lakes and sampling regime is given in Chapter 2.

Physical and chemical parameters

Dissolved oxygen, temperature, and conductivity were measured in the lakes at 20 cm depth intervals using an YSI Model 50 DO meter and a YSI Model 30/10FT conductivity meter. pH was measured in an unfiltered, depth integrated sample of the water column using a Eutech Instruments Ltd pH Scan WP1 meter. Secchi depth was recorded using a 20 cm diameter Secchi disk suspended from the side of a boat.

Surface water samples for chlorophyll *a*, suspended sediment, and nutrient analyses were taken by filling a 1 litre container at each site. Containers were acid washed in a 10% HCl solution, filled with water that had undergone reverse osmosis, and put on ice prior to sample collection. After water had been collected, samples were stored on ice in the dark until filtration occurred. The samples were filtered on the same day as collection, with the exception of samples taken before December 2001, which were frozen prior to filtering.

Chlorophyll *a* was extracted with acetone. Concentrations were measured by fluorescence before and after acidification with 0.1M HCl using a Turner Designs 10-AU fluorometer (Arar & Collins 1992). Suspended sediments (total suspended solids dried at 103-105°C and total volatile and fixed solids dried at 103-105°C then ignited at 550°C) were tested for in accordance with the guidelines set out in Eaton *et al.* (1995) using Whatman GF/C or Advantec glass fiber filter papers. PO_4, NH_4, NO_2, and NO_x were analysed using a Lachat QuickChem FIA 8000 series auto-analyser. Light was measured on the 11 February 2002 using a LI-COR type LI-192SA underwater quantum sensor.

Plankton

A depth-integrated sample of plankton was taken using an open-ended polythene pipe (75 mm diameter) inserted vertically into the water column descending the entire depth of the lake. A rubber bung was inserted in the top of the pipe, the pipe was withdrawn to just below the water surface and another rubber bung was inserted in the bottom of the pipe. The water was emptied into a bucket. This was repeated until at least 7 litres of water had been sampled. Phytoplankton samples were taken by filling 400 ml containers with a well-mixed sample and immediately preserving in 10% ethanol and lugols solution. Zooplankton were sampled by sieving a known volume of water through a 37 μm net, which can adequately sample smaller zooplankton taxa and while retaining the efficiency otherwise lost through suspended sediment and phytoplankton clogging the mesh (Orcutt & Pace 1984; Berner-Fankhauser 1987; Duggan 1999; Burger *et al.* 2002). The concentrated liquid was transferred into 400 ml containers and immediately preserved with 10% formalin until enumeration could take place.

Zooplankton enumeration was performed by first concentrating the sample through a 37 μm net so that the formalin could be thoroughly rinsed away. The sample was then made up to a known volume with water that had undergone reverse osmosis. Due to the high concentrations of zooplankton and suspended sediment, it was impractical to count entire samples and sub-samples were taken instead. The zooplankton were suspended by stirring the sample in a figure of eight motion, which has been found to be a more accurate method of mixing zooplankton for sub-sampling (Cryer 1983). While the zooplankton were suspended 5 ml aliquots of the sample were taken and pipetted onto a Perspex zooplankton counting tray measuring 60 mm by 25 mm. The zooplankton were counted under 60x magnification using an Olympus SZ60 microscope. Counting continued until at least 100 of each of the three most dominant organisms had been recorded. Identifications were made using Shiel (1995).

The phytoplankton of the campus lakes were identified using Etheredge (1987); Entwistle (1997); Moore (2000); Baker & Fabbro (2002) and John (2002). They were counted using the Utermöhl sedimentation technique (Utermöhl 1958) on a 10-40ml aliquot taken from a well-mixed sample. Samples were settled for approximately one day for every eight centimeters of settling height (Furet & Benson-Evans 1982). Diameter belt transects were counted on an Olympus inverted microscope at 400x magnification. To obtain an acceptable level of accuracy, counting continued until each major species had been recorded at least 100 times (Lund *et al.* 1958). Empty or damaged cells (if less than half of the normal cellular contents were missing) were not included in the analyses.

Phytoplankton abundance was calculated using the formula (Grace Analytical Laboratory 1994):

$$\text{Cells/ml} = \frac{CxTA}{LxWxVxS}$$

where C = cell count, L = length of strip (mm),W = width of strip (mm), V = volume of chamber (mL), S = number of strips counted and TA = total area of chamber bottom (mm^2).

Analyses

Where the data met the assumptions of parametric statistics, one-way ANOVAs were performed. Where the data did not conform to at least one assumption, a non-parametric Kruskal-Wallis test was performed. Non-parametric tests are more common in analyzing water quality variables as the data are often not normally distributed and therefore neither are their regression residuals. Parametric tests are also limited in the analyses of marked seasonal variability, which often occurs in water quality samples (Vant & Wilson 1998). The statistical program Minitab version 12 was used to perform data analyses.

Multi-dimensional scaling (MDS) was used to estimate zooplankton and phytoplankton community similarity. This method takes into account data on both species composition and abundance. Species were only included if their relative abundance was $\geq$ 4% of any sample (Duggan *et al.* 2002). MDS was performed using a Bray-Curtis similarity. A similarity of percentages test (SIMPER) was used to examine the taxa that were responsible for the similarity within samples and the taxa primarily responsible for the differences between samples. A one-way analysis of similarities (ANISOM) was used to find out if the species assemblage in the samples differed from random distributions. Bray-Curtis similarity matrices were used as a base for the ANISOM tests and 999 permutations were performed.The R value was calculated using the formula (Clarke & Gorley 2001):

$R = (r_B - r_W)/(M/2)$

where r_B = the average rank of similarities from all pairs of replicates across all groups, r_W = the average of all rank of similarities among replicates within groups, M = n(n-1)/2; where n = the total sample number.

The closer the R value is to 1, the more dissimilar are the groups to be compared, whereas a lower R value indicates that the groups are more similar. The statistical package PRIMER was used to perform the above tests (Clarke & Warwick 1994).

Redundancy analyses (RDA) were performed to identify the influence that environmental variables may have had on the phytoplankton and zooplankton communities. Species were only included if their relative abundance was $\geq$ 4% of any sample. RDA was used instead of canonical correspondence analyses as the species showed a linear response to the environmental gradients. Species data was square-root transformed prior to analyses to down-weight dominant species. To ensure that a comparable scale existed between different variables, environmental data were standardized prior to analyses using the equation (Jongman *et al.* 1987):

$$Y_{Ki} = (Y_{ki} - Y_{K+n})/S_k$$

where Y_{Ki} = standardized variable, Y_{ki} = measured variable, Y_{K+n} = mean of data, S_k = standard deviation of data.

Temperature, specific conductance, pH, NO_x, NO_2, phosphate, and ammonia were tested as variables against zooplankton and phytoplankton species abundances. Phytoplankton abundance was used as an additional variable to explain zooplankton abundance. Any missing environmental variables were replaced with the average value of the data set. Variables that were found to highly co-vary, as indicated by inflation factors >20 were not included in analyses. Forward selection and Monte Carlo permutations (using 199 permutations) were carried out to identify the statistical significance of any environmental-species correlations. RDA analyses were performed using the statistical package CANOCO v. 4.02. Bi-plots of the analyses are presented where points represent samples and arrows represent species data and environmental variables. The length of the arrows indicates how strongly it is correlated with an ordination axis (ter Braak 1998).

RESULTS

Water Quality

Specific conductance was significantly higher in Oranga and Knighton lakes compared to Chapel Lake ($P < 0.001$) (Table 3.1). Specific conductance, expressed as a depth-integrated mean, was consistently higher in Oranga and Knighton lakes, and had a larger range, than in Chapel Lake (Figure 3.1A). There were no obvious seasonal patterns in specific conductivity, pH or Secchi depths in the lakes (Figure 3.1A-C). pH ranged from 6.9 to 10.1 in the three lakes and was alkaline for most of the measurements (Figure 3.1B). Knighton Lake had a significantly higher pH than the other two lakes ($P < 0.001$). Chapel Lake had a significantly deeper Secchi depth ($P < 0.001$) than Oranga and Knighton lakes. Oranga Lake had a significantly deeper Secchi depth than Knighton Lake (Table 3.1). Oxygen was typically high in all three lakes, reaching levels of supersaturation on a number of occasions. The temperatures of the lakes were closely matched to each other and showed a marked annual variation (Figure 3.2A). Chlorophyll *a* was very high in all three lakes but showed a marked summer peak in Knighton and Oranga lakes while staying relatively stable in Chapel Lake (Figure 3.2C). For all significantly different variables, Kruskal-Wallis tests showed no significant variation between sites within a lake but significant variation between different lakes ($P < 0.001$).

Table 3.1: Mean values of selected physio-chemical parameters for the University of Waikato campus lakes between 12 July 2001 and 4 October 2002. Values that do not share a letter are significantly different from each other at the *P* values given.

Lake	N	Secchi depth (m)	Specific conductance (μS cm^{-1})	pH	Oxygen (g m^{-3})	Temperature (°C)	Chlorophyll *a* (mg m^{-3})
Chapel	26	0.58^a	54^a	8.0^a	8.0^a	15.4^a	49^a
Oranga	25	0.49^b	158^b	8.2^a	10.9^b	15.6^a	52^a
Knighton	24	0.36^c	163^b	8.6^b	8.9^{ab}	15.3^a	67^a
P		≤ 0.001	≤ 0.001	≤ 0.001	≤ 0.001	> 0.05	> 0.05

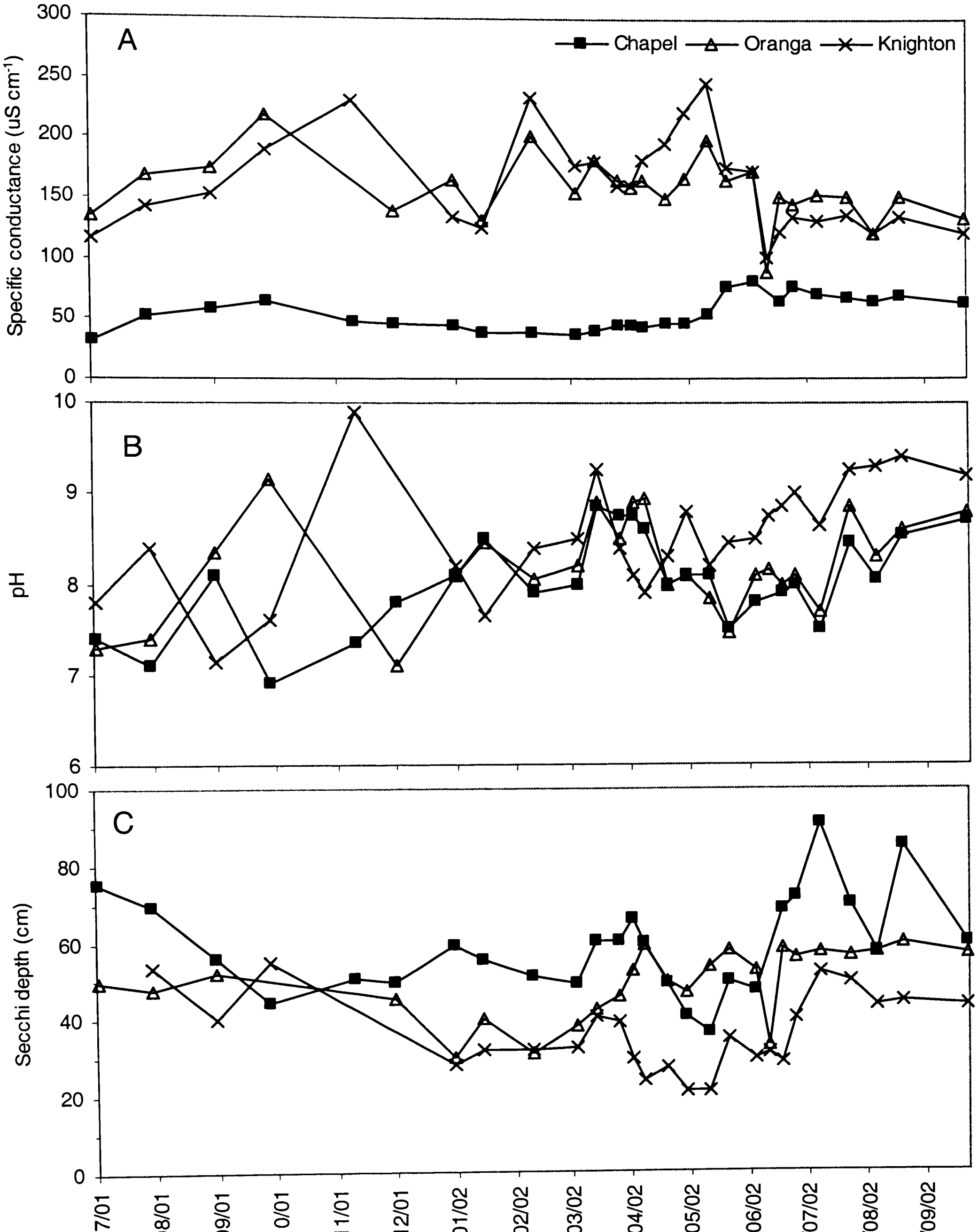


Figure 3.1: Temporal variation in **A**. specific conductance, **B**. pH, and **C**. Secchi depth in the University of Waikato campus lakes between July 2001 and October 2002.

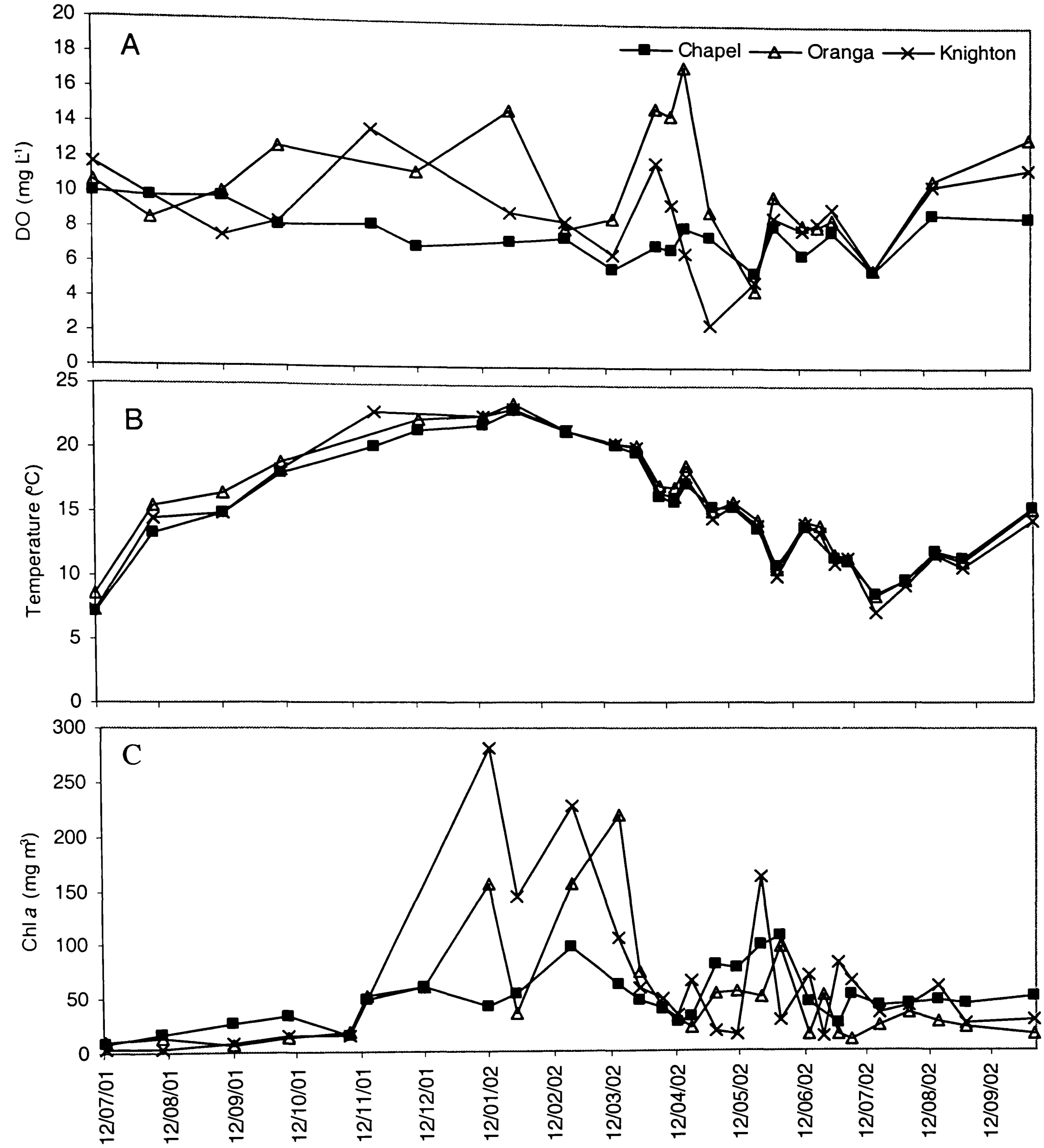


Figure 3.2: Temporal variation in **A.** dissolved oxygen, **B.** temperature and **C.** chlorophyll *a* in the University of Waikato campus lakes between July 2001 and October 2002.

More suspended solids were present in Oranga and Knighton lakes compared to Chapel Lake ($P < 0.001$) (Figure 3.3A). The majority of the suspensoids in Oranga and Knighton lakes were inorganic, whereas the majority of the suspensoids in Chapel Lake were organic (Figure 3.3B).

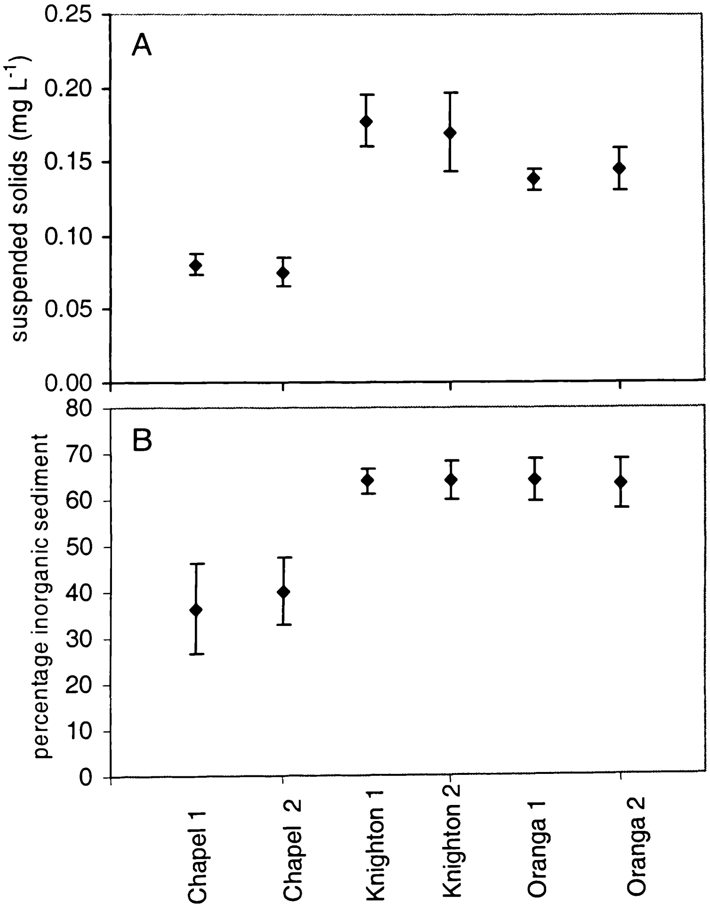


Figure 3.3: Suspended solids in the University of Waikato campus lake, sampled at two sites per lake. N = 7. **A.** Total suspended solids; **B.** percent of suspended solids that were inorganic. Error bars are ± 1 S.E

Light was rapidly attenuated in all three lakes. There was less light available at the bottom of the water column in Chapel Lake than in Oranga and Knighton lakes (Figure 3.4).

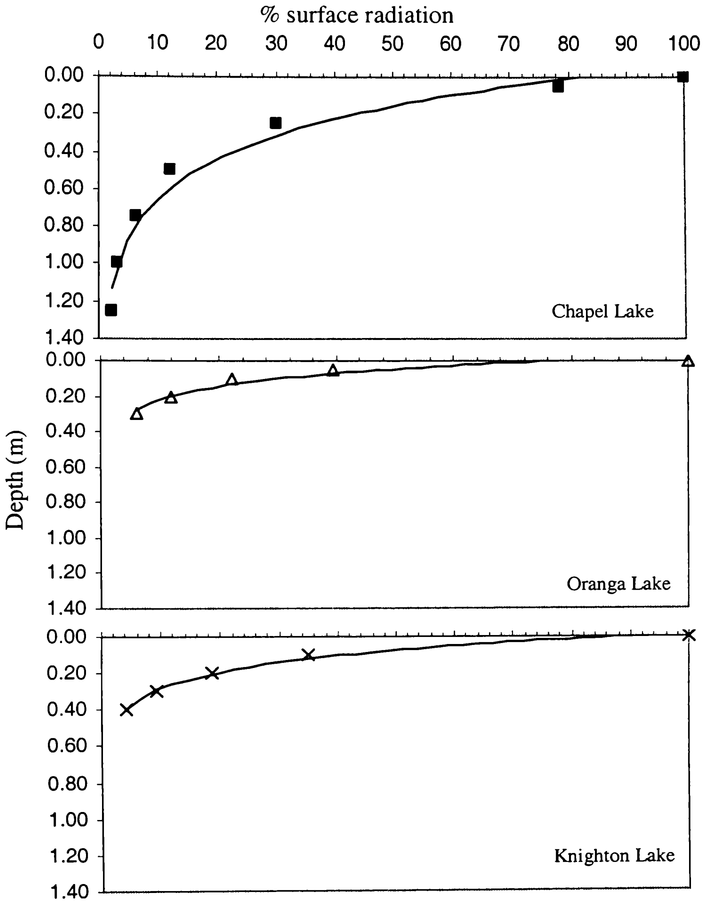


Figure 3.4: Light attenuation in the University of Waikato campus lakes on 11 February 2002.

Phytoplankton

Phytoplankton from 33 genera were identified from the three lakes (Appendix 1a). There were dramatic summer algal blooms recorded in Oranga and Knighton lakes, dominated by *Microcystis* spp. and *Trachelomonas* spp. The density of phytoplankton in Chapel Lake stayed relatively stable during this time (Figure 3.5). There were large amounts of nanoplankton observed in Oranga and Knighton lakes, especially in the summer months however this was not quantified in this study.

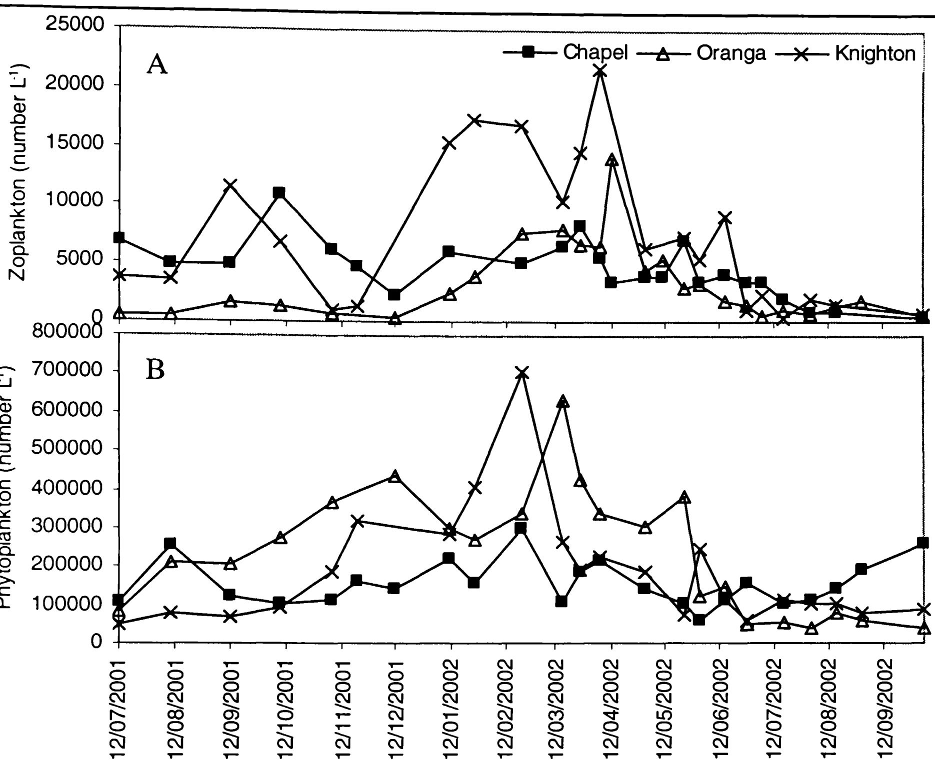


Figure 3.5: Temporal variation in **A**. total zooplankton and **B**. total phytoplankton in the University of Waikato campus lakes between July 2001 and October 2002

The phytoplankton communities of Knighton Lake and Oranga Lake were significantly different to the phytoplankton communities of Chapel Lake. In Knighton Lake, cyanobacteria dominated the algal groupings with strong blooms of *Microcystis* spp. occurring throughout the year. Cyanobacteria and Euglenophyta were the most dominant taxa in Oranga Lake, specifically *Microcystis* spp. and *Trachelomonas* spp. The algal groupings in Chapel Lake were significantly different with Chlorophyta dominating, specifically *Closterium acutem* var. *linea* and *C. acutem* var. *variabile* with occasional blooms of the Chrysophyte *Dinobryon sertularia* (Figure 3.6).

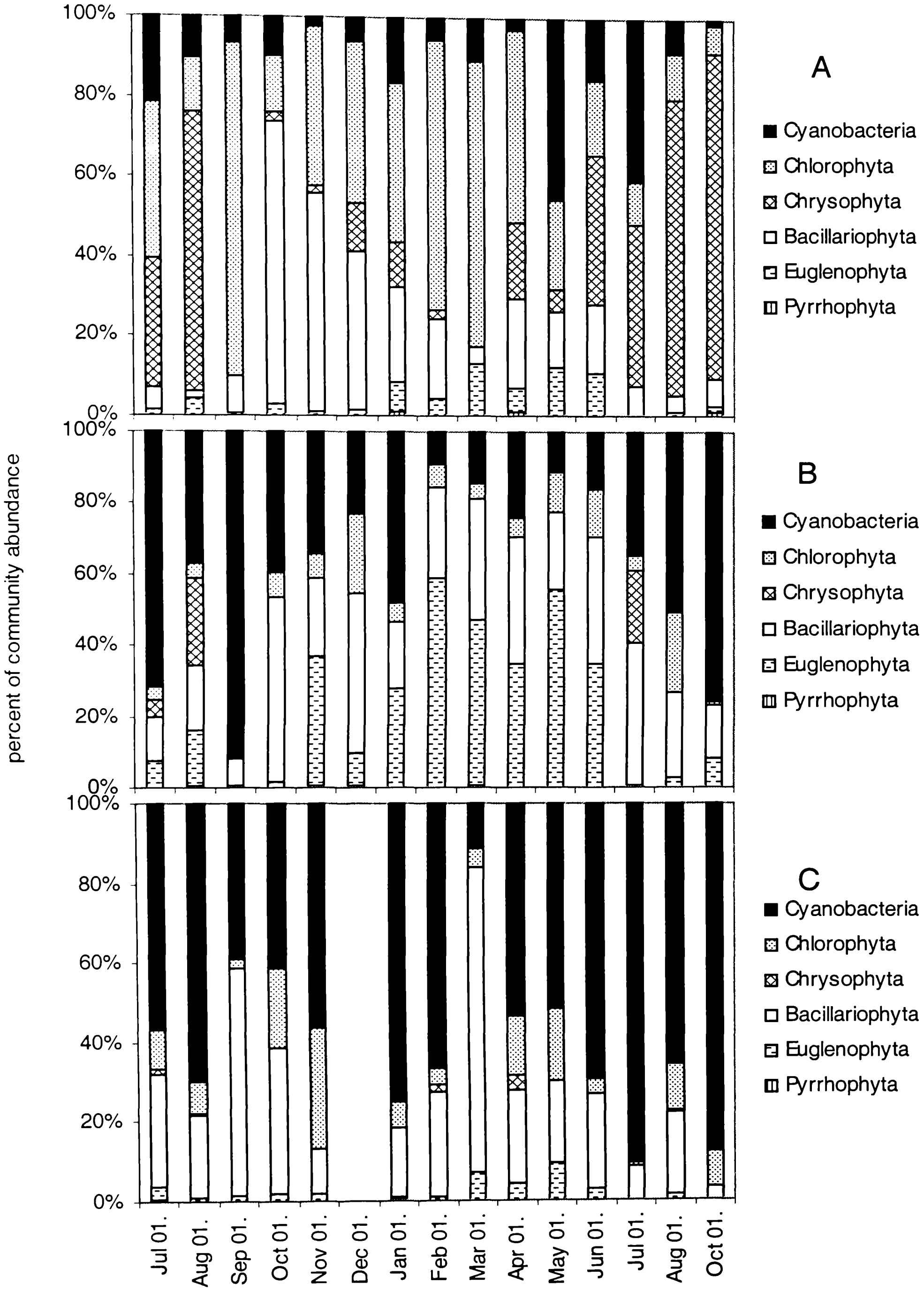


Figure 3.6: Temporal variation in the dominant phytoplankton taxa and their relative abundances in **A**. Chapel Lake, **B**. Oranga Lake and **C**. Knighton Lake. Samples were not taken from Knighton Lake in December 2001 as macrophyte removal occurred during this time, which may have influenced results.

Phytoplankton multivariate analyses

The MDS ordinations showed strongly defined groupings between the phytoplankton communities of Chapel Lake and Oranga and Knighton lakes (Figure 3.7).

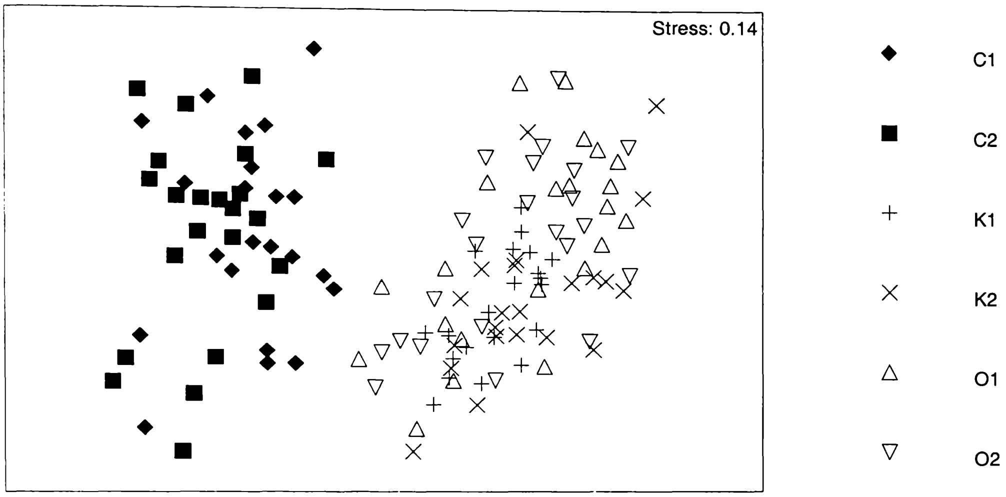


Figure 3.7: MDS plot of phytoplankton community similarity in the University of Waikato campus lakes between July 2001 and October 2002. C1 = Chapel Lake, Site 1; K1 = Knighton Lake, Site 1; O1 = Oranga Lake, Site 1.

The results of a one-way ANISOM analysis indicate that there was significant differentiation in phytoplankton community composition and abundance between all three lakes but not within lakes, as shown by the R values. All between lake comparisons were significant at $P \leq 0.05$. The only sites that were not significantly different were sites within lakes. The phytoplankton populations at the sites within Chapel Lake were more dissimilar to the other two lakes (shown by higher R values) than Knighton and Oranga lakes compared with each other (Table 3.2).

Table 3.2: Summary of the one-way analysis of similarities for phytoplankton community composition and abundance July 2001-October 2002. The closer that the R statistic is to one, the more dissimilar are the samples.

Groups	R statistic	Significance level
Chapel 1, Chapel 2	-0.026	0.817
Chapel 1, Knighton 1	0.850	**0.001**
Chapel 1, Knighton 2	0.822	**0.001**
Chapel 1, Oranga 1	0.763	**0.001**
Chapel 1, Oranga 2	0.773	**0.001**
Chapel 2, Knighton 2	0.903	**0.001**
Chapel 2, Knighton 1	0.892	**0.001**
Chapel 2, Oranga 1	0.843	**0.001**
Chapel 2, Oranga 2	0.847	**0.001**
Knighton 1, Knighton 2	-0.008	0.517
Knighton 1, Oranga 1	0.227	**0.001**
Knighton 1, Oranga 2	0.209	**0.002**
Knighton 2, Oranga 1	0.181	**0.003**
Knighton 2, Oranga 2	0.178	**0.001**
Oranga 1, Oranga 2	-0.036	0.944

SIMPER analysis showed that the taxa that contributed the greatest to the dissimilarity between Chapel Lake and Knighton and Oranga lakes were *Microcystis* spp. Taxa that contributed the greatest to the similarity within Chapel Lake were *Closterium acutem* var. *linea* and *Closterium acutem* var. *variabile* with *Microcystis* spp. contributing greatest to the similarity within the shallower lakes (Table 3.3).

Table 3.3: Summary of SIMPER analysis on phytoplankton abundance and community composition in the campus lakes; **A**) taxa that contribute the greatest percent to the similarity within samples and **B**) taxa that contribute the greatest to the dissimilarity between samples.

A)

Taxon	Contribution (%)
Chapel Lake	
C. acutem var. *linea*	31.3
C. acutem var. *variabile*	19.1
Dinobryon sertularia	15.4
Oranga and Knighton lakes	
Microcystis spp.	57.7
Aulacoseira spp.	10.7
Nitzchia spp.	8.5
Trachelomonas volvocina	8.1

B)

Taxon	Contribution (%)
Microcystis spp.	20.7
C. acutem var. *linea*	14.2
Dinobryon sertularia	12.3
C. acutem var. *variabile*	11.2

The influence that various environmental variables have on species abundance, as shown by the results of the forward selection and Monte Carlo permutation tests, are given in Table 3.4. The conditional effects (Lambda A) column "shows the environmental variables in order of their inclusion in the model, together with the additional variance each variable explains at the time of inclusion". The marginal effects (Lambda 1) column "lists the individual environmental variables in the order of the variance they explain singly" treating each variable as if it were the only one used. Axis one will be influenced the strongest by the first Lambda 1 variable and axis two will be influenced the strongest by the second variable (ter Braak & Smilauer 1998). The first two axes of the redundancy analysis (RDA) on the phytoplankton abundance

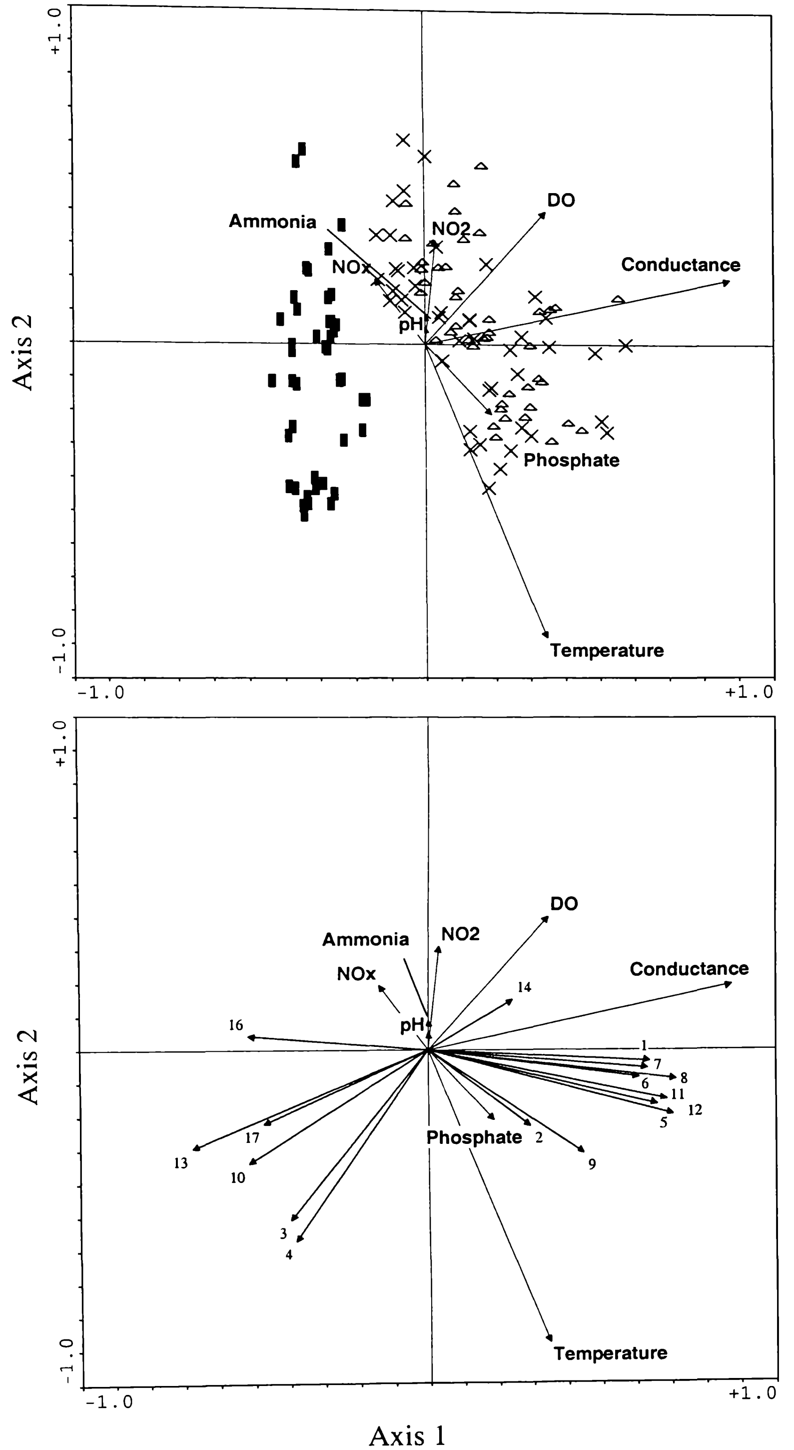
(Figure 3.8) have eigenvalues of 0.346 and 0.058 respectively, which cumulatively accounted for 34.6% of the total variation in the species abundance data. Axes 1 and 2 account for 94.4% of the relationships observed between species and environmental variables (Table 3.4). In the phytoplankton populations of the campus lakes specific conductance and temperature were found to have the greatest influence on the distribution of species abundance. When tested singly they accounted for 27% and 9% of the variation in the phytoplankton populations and 27% and 7% respectively when other variables were taken into account. The dissolved oxygen levels and pH were also found to have significant correlations with phytoplankton abundance. The level of phosphate was not significant at $P \leq 0.05$ but did have a significant correlation at $P = 0.06$ (Table 3.5). None of the environmental values tested were found to significantly co-vary with any others, therefore none were omitted.

Table 3.4: Summary of RDA ordination for phytoplankton abundance data.

Axis	1	2	3	4
Eigenvalues	0.346	0.058	0.014	0.005
Species-environmental correlations	0.860	0.657	0.371	0.294
Cumulative variance (%) of				
species data	34.6	40.4	41.8	42.4
species-environmental relation	80.6	94.4	97.7	99.0

Table 3.5: Results of the forward selection and Monte Carlo permutation tests for phytoplankton abundance data.

Variable	Conditional effects (Lambda A)	Marginal effects (Lambda 1)	P
Specific conductance	0.27	0.27	**≤0.005**
Temperature	0.07	0.09	**≤0.005**
pH	0.03	0	**≤0.005**
DO	0.03	0.05	**≤0.005**
Phosphate	0.01	0.02	0.06
NO_x	0.01	0.01	0.165
Ammonia	0.01	0	0.645
NO_2	0	0	0.180



Code	Species
1	*Microcystis* spp.
2	*Paedeastrum duplex*
3	*Coeleastrum* spp.
4	*Staurastrum* spp.
5	*Scenedesmus* spp.
6	*Cymbella* spp.
7	*Epithemia* spp.
8	*Gomphonema* spp.
9	*Trachelomonas* spp.
10	*Closterium acutem* var. *variabile*
11	*Aulacoseira* spp.
12	*Closterium gracile*
13	*Closterium acutem* var. *linea*
14	*Frustrulia* spp.
16	*Dinobryon sertularia*
17	*Quadrigula* spp.

Figure 3.8: Ordination diagrams based on redundancy analyses of phytoplankton abundance data. Symbols correspond to sampling occasions in each lake. Numbers correspond to species data. Some species data that showed weak association with axes 1 or 2 have been omitted for clarity.

The RDA ordination showed strong distinction between samples collected from Chapel Lake compared with samples collected from Oranga and Knighton lakes. All of the Chapel Lake samples showed a negative association with axis 1 and most of the samples from Knighton Lake and Oranga lakes showed a positive association with axis 1. Samples from Chapel Lake were distributed along axis 2, both positively and negatively. The species that were characteristic of Chapel Lake, as shown by SIMPER analyses, were the species that were negatively correlated with axis 1 and 2. The species that were characteristic of Oranga and Knighton lakes were positively correlated with axis 1. Essentially all the phytoplankton species were negatively correlated with axis 2 (Figure 3.8).

Zooplankton

Rotifers covering nine genera and cladocerans from four genera were identified from the three lakes (Appendix 1b). Cyclopoid copepods, calanoid copepods and copepod nauplii were present in all the samples but were not identified below this taxonomic level. There were dramatic summer blooms of rotifers in Oranga and Knighton lakes – consisting mainly of *Keratella* spp. and *Filinia* spp., while the zooplankton abundances of Chapel Lake remained relatively stable (Figure 3.5). The zooplankton communities in the three lakes were dominated by rotifers and small cladocerans for most of the year, with *Keratella*, *Filinia*, *Brachionus*, and *Bosmina* species being numerically dominant in the two shallow lakes. *Trichocerca*, *Keratella*, *Lophocanthous*, and *Bosmina* species were the most abundant zooplankton genera in Chapel Lake. There was a marked shift in dominance in Knighton Lake from rotifers to cladocerans of the genus *Bosmina* in May 2002 and in June 2002 in Oranga Lake (Figure 3.9). Large bodied *Daphnia* sp. c.f. *pulex* were not present in 2001; however their numbers increased in Oranga Lake from July 2002 until the end of the sampling period.

Zooplankton multivariate analyses

In contrast to the phytoplankton assemblages of the three lakes, there was no marked distinction in zooplankton community composition between lakes. An MDS ordination plot showed limited grouping of the Chapel Lake samples together with a relatively large spread of the Knighton Lake and Oranga Lake communities (Figure 3.10). There were significant differences in zooplankton community composition and abundance between Chapel Lake and the shallower lakes but there were no statistically significant differences between Knighton and Oranga lakes (Table 3.6). The taxa responsible for the dissimilarities between Chapel Lake and the two shallower lakes were *Bosmina* spp. *Keratella tecta* and copepod nauplii. The three taxa that were most responsible for the similarities within Chapel Lake are *Bosmina* spp. *Trichocerca* sp and copepod nauplii. The taxa that were most responsible for the similarities within Knighton and Oranga lakes were *Bosmina* spp. copepod nauplii and *Keratella tecta* (Table 3.7). The fact that the same species accounted for similarities within different lakes indicates that it is their relative abundances that is driving the differences in community composition.

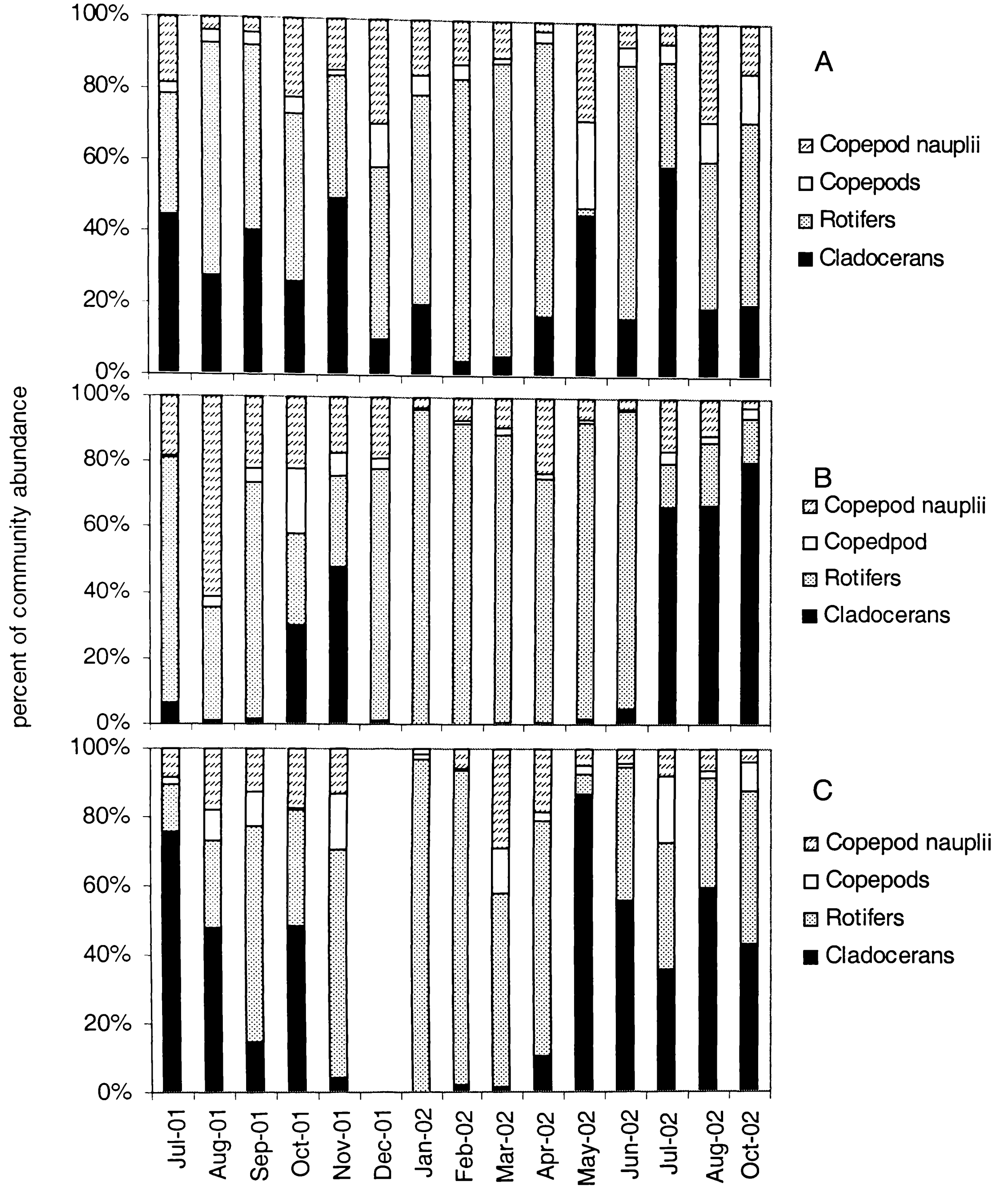


Figure 3.9: Temporal variation in the major zooplankton taxa in **A**. Chapel Lake; **B**. Oranga Lake and **C**. Knighton Lake and their relative abundances. Samples were not taken from Knighton Lake in December 2001 as macrophyte removal occurred during this time, which may have influenced results.

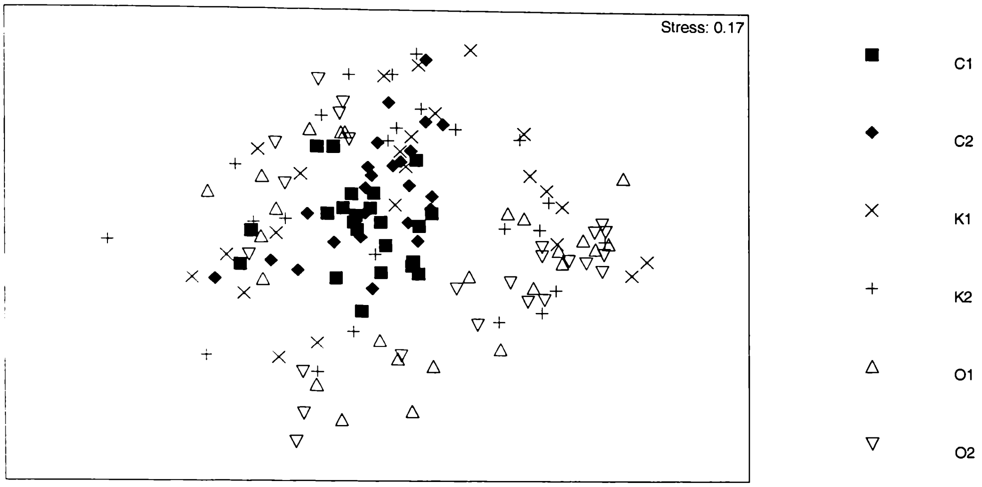


Figure 3.10: MDS ordination plot of the University of Waikato campus lake zooplankton communities between July 2001 and October 2002. C1 = Chapel Lake, site 1; K1 = Knighton Lake, site 1; O1 = Oranga Lake, site 1.

Table 3.6: Summary of one-way analysis of similarities analysis for the zooplankton community composition and abundance in the campus lakes, July 2001 - October 2002. The closer that the R statistic is to one, the more dissimilar the samples.

Groups	R statistic	Significance level
Chapel 1, Chapel 2	0.045	0.066
Chapel 1, Knighton 1	0.296	**0.001**
Chapel 1, Knighton 2	0.273	**0.001**
Chapel 1, Oranga 1	0.292	**0.001**
Chapel 1, Oranga 2	0.320	**0.001**
Chapel 2, Knighton 2	0.233	**0.001**
Chapel 2, Knighton 1	0.219	**0.001**
Chapel 2, Oranga 1	0.331	**0.001**
Chapel 2, Oranga 2	0.330	**0.001**
Knighton 1, Knighton 2	-0.420	0.995
Knighton 1, Oranga 1	0.034	0.120
Knighton 1, Oranga 2	0.022	0.175
Knighton 2, Oranga 1	0.033	0.129
Knighton 2, Oranga 2	0.035	0.117
Oranga 1, Oranga 2	-0.024	0.819

Table 3.7: Summary of SIMPER analysis on zooplankton abundance and community composition in the campus lakes. **A**) Taxa that contribute the greatest percent to the similarity within samples and **B**) taxa that contribute the greatest to the dissimilarity between samples.

A)

Taxon	Contribution (%)
Chapel Lake	
Bosmina spp.	25.2
Trichocerca spp.	24.5
Copepod nauplii	14.0
Oranga and Knighton lakes	
Bosmina spp.	28.1
Copepod nauplii	19.7
Keratella tecta	16.4

B)

Taxon	Contribution (%)
Bosmina spp.	21.3
Keratella tecta	11.8
Trichocerca spp.	10.3

The first two axes of the RDA ordination on the zooplankton data have eigenvalues of 0.253 and 0.063 respectively, which cumulatively accounted for 31.5% of the species data. Axes 1 and 2 account for 79.1% of the species-environmental relations (Table 3.8). The significance that various environmental variables had on species abundance, as shown by forward selection and Monte Carlo permutation tests, are given in Table 3.9. Significant correlations ($P \leq 0.05$) were found between zooplankton abundance and the levels of phytoplankton, specific conductance, temperature, pH and phosphate, dissolved oxygen and NO_X. The density of phytoplankton had the greatest influence on the distribution of zooplankton species abundance. When other variables were taken into account (Lambda A values), it accounted for 18% of the variation in zooplankton populations. Temperature accounted for 6% of the variance and temperature, pH and phosphate each accounted for 2% of the variance. When each variable was treated singly (Lambda 1 values), phytoplankton abundance accounted for 18% of the variation. Temperature and specific conductance, accounted for 16% and 7% of the variation respectively. pH, phosphate, dissolved oxygen and NO_x each explained 3% of the variation. There is a weak separation of samples from Knighton and Oranga lakes compared to Chapel Lake. The majority of Chapel Lake samples are correlated negatively with both axes. The species that have the strongest relationships with temperature are the species that dominated the summer blooms in the two shallower lakes (Figure 3.11).

Table 3.8: Summary of RDA ordination for zooplankton abundance data.

Axes	1	2	3	4
Eigenvalues	0.253	0.063	0.048	0.016
Species-environment correlations	0.734	0.595	0.667	0.541
Cumulative percentage variance				
of species data	25.3	31.5	36.3	37.9
of species-environment relation	63.4	79.1	91.1	95.0

Table 3.9: Results of the forward selection and Monte Carlo permutation tests for zooplankton abundance data.

Variable	Conditional effects (Lambda A)	Marginal effects (Lambda 1)	*P*
Phytoplankton	0.18	0.18	**≤0.005**
Temperature	0.06	0.16	**≤0.005**
Conductance	0.05	0.07	**≤0.005**
pH	0.03	0.03	**≤0.005**
Phosphate	0.02	0.03	**≤0.005**
DO	0.03	0.03	**≤0.005**
NO_x	0.01	0.03	**0.035**
Ammonia	0.01	0.02	0.075
NO_2	0.01	0	0.125

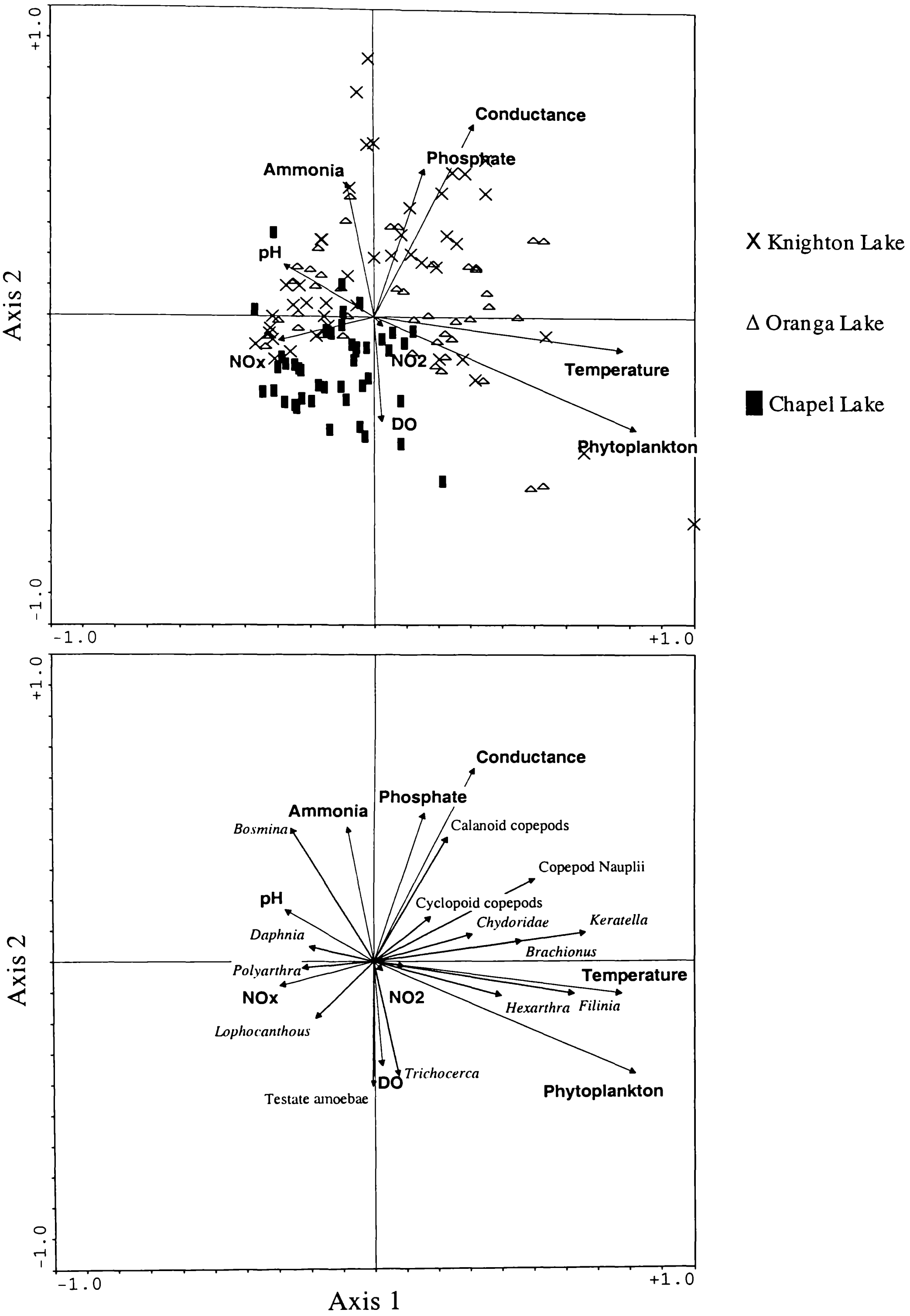


Figure 3.11: Ordination diagram based on RDA of zooplankton abundance data. Symbols correspond to sampling occasions in each lake. Some species data that showed weak association with axes 1 or 2 have been omitted for clarity.

DISCUSSION

Water quality

The water quality parameters tested indicated that Chapel Lake had higher water quality than Oranga and Knighton lakes. In particular, the significantly deeper Secchi depths in Chapel Lake compared with Oranga and Knighton lakes confirms earlier studies (e.g., Pollard 1995; Willis 1996) that have identified the two shallow lakes as problematic and Chapel Lake as being in a more desirable state.

Mean specific conductance values for Oranga and Knighton lakes were significantly higher than Chapel Lake. The values were also higher than other small Waikato lakes (Boswell 1985). This was probably a result of intense mixing (Chapter 2) and sediment resuspension and from the runoff from the University grounds.

The pH levels in the three lakes were within the range reported by Boswell (1985) in shallow Waikato lakes (6.2 - 12.3). pH showed no seasonal trends although Knighton Lake had significantly higher pH throughout the year. High levels of photosynthesis increases pH (Scheffer 1998) and it is possible that this occurs in Knighton Lake.

The variations in temperature in the lakes closely followed the seasonal variations in air temperature recorded at the Ruakura climate station data (Chapter 2). Temperature ranges in the three lakes are greater than the expected values for North Island lakes of 13.0 ± 3.1°C (Green *et al.* 1987) but have similar summer values to other shallow lakes in the Waikato (Boswell *et al.* 1985; Miller 2002). This is due to their shallow nature, therefore low surface area to volume ratio (Green *et al.* 1987) and lack of shading.

High dissolved oxygen concentrations were higher than the ranges found by Boswell (1985) but comparable to Lake Moana-nui, which is of a similar depth (Miller 2002). High photosynthesis rates and the intense mixing regime are likely to be responsible for the high concentrations of dissolved oxygen.

Chlorophyll *a* concentrations over the entire year were not significantly different in the lakes despite very high summer values recorded in Oranga and Knighton lakes but were not found in Chapel Lake. The lack of a significant difference was likely to be influenced by the relatively low values in all three lakes for much of the year. These would have obscured any significant difference that may have existed over summer.

Suspended sediment was significantly higher in the two shallow lakes than in Chapel Lake. This is likely to be partly due to the higher algal biomass in the shallow lakes. However, as the majority of the suspended sediment in the two shallow lakes was inorganic, sediment resuspension is likely to be the major cause. This may be caused by wind and temperature induced mixing. Yanful & Catalan (2002) found that the critical depth for sediment resuspension was between 1.18 and 1.34 m in large shallow ponds with an average wind speed of 10 m s^{-1}. This agrees with the data presented in this thesis, in that the two lakes with maximum depths less than 1.18 m (Oranga and Knighton lakes) have significantly higher suspended sediment than Chapel Lake, which is deeper than 1.34 m. Water bodies with the surface area and depth of Knighton and Oranga lakes need wind speeds of between 2 and 3 m s^{-1} to subject 50% of the bottom sediments to resuspension. A lake with the surface area and depth of Chapel Lake would need a wind speed of approximately 8 m s^{-1} for sediment resuspension to occur (Scheffer 1998). The mean wind speed at the nearby Ruakura climate station was 2.5 m s^{-1} (Chapter 2). Chapel Lake is less exposed than Knighton and Oranga lakes and it is likely that the wind speeds were even less over its surface. These data strongly suggest that there is strong mixing (see Chapter 2) and a significant amount of sediment resuspension occurring in the two shallow lakes, which did not occur to the same degree in Chapel Lake.

Seasonal variations in phytoplankton abundance

Total phytoplankton abundance peaked in summer in the two shallow lakes. This is probably a result of increased light intensities and higher temperatures for longer periods of the day that occur during summer (Viner & White 1987). Summer blooms of phytoplankton have been recorded in many lakes worldwide (e.g., Kallio 1994; Tryfon 2001) and in some New Zealand lakes (Viner & White 1987).

Cyanobacteria were present in all three lakes and dominant in Oranga and Knighton lakes for much of the year. Cyanobacteria often dominate shallow lakes (Gulati & van Donk 2002) and a hydraulic residence time of < 8-14 days is thought to be a key factor for their dominance in shallow (< 5 m), urban lakes. The hydraulic residence time can be as long as 68 days in Knighton and Oranga lakes (Hicks unpublished data). Therefore, the cyanobacterial dominance was expected.

The adaptations of cyanobacteria to a low underwater light climate, nitrogen fixing capacity (Nixdorf & Deneke 1997) and their ability to alter their photosynthetic rate in response to the mixing regime (Kronkamp *et al.* 1992) provide them with advantages over other forms of algae. In the campus lakes it is possible that the intense mixing regime, together with the high amount of suspended solids resulted in low light levels and large variations in the amount of light available to individual alga over the course of a day. This may have provided suitable conditions for cyanobacteria to dominate.

Buoyancy regulation by *Microcystis* colonies is known to be a major factor controlling its dominance in some systems (Bonnet & Poulin 2002). The campus lakes were weakly stratified during the day for much of the year (Chapter 2) and it is possible that the positive buoyancy of *Microcystis* allowed it to take best advantage of the available light in the lakes.

Cyanobacteria such as *Microcystis* can also store nutrients in times of excess and use them during future shortages. (Sbiyyaa *et al.* 1998; Boumnich *et al.* 2001). In the campus lakes it is possible that *Microcystis* colonies are able to store nutrients that may have been released from the sediment as a result of night-time temperature induced mixing (Chapter 2) and utilize them during the day as the light levels increase.

The significantly different phytoplankton communities throughout the year in all three lakes are likely to be a function of the different morphology, retention times, light penetration, nutrient regimes and mixing regimes within these three lakes. As expected, the two lakes that had similar morphology and were hydraulically linked had more similar phytoplankton communities than Chapel Lake, which was hydrologically isolated.

Phytoplankton associations with environmental variables

The phytoplankton populations varied in community composition and abundance in all three lakes throughout the year. The RDA ordination showed that the phytoplankton of Chapel Lake had the greatest distribution around axis 2. The forward selection and Monte Carlo permutations showed that temperature was the variable that constricted axis 2 to the greatest extent. Therefore, in Chapel Lake, the variable that was the greatest driver of phytoplankton abundance was temperature. Samples from Oranga and Knighton lakes were distributed around axes 1 and 2. Temperature and specific conductance were the strongest community drivers in these lakes.

The strong influence of temperature on phytoplankton community abundance in the campus lakes concurs with Brylinsky (1980) who recorded increasing primary production in lakes with higher temperatures. Water temperature has been found to explain the most variation (10%) of phytoplankton in a shallow, frequently mixed lake (Agbeti *et al.* 1997). These data also agrees with Willis (1996) who stated that it was unlikely that production in the campus lakes was nutrient limited.

Specific conductance also influenced the abundances of phytoplankton populations in Oranga and Knighton lakes. This is likely to be linked to the increased nutrients available in the lake water and is possibly linked to the sediment resuspension that occurs in these lakes.

Seasonal variations in zooplankton abundance

Total zooplankton abundance peaked in summer in the two shallow lakes. This peak is likely to be closely related to the amount of edible algae present and also influenced by higher temperature and a subsequent increase in growth rates. The zooplankton communities were dominated by rotifers for much of the year. This is similar to other studies, for example rotifers (mainly *Keratella* spp. and *Trichocerca* spp.) have made up 85% of the of the zooplankton communities in the Waikato River (Burger *et al.* 2002).

Shallow urban lakes have been shown to be suitable areas for rotifer populations to become established (Ejsmont-Karabin & Kuczynska-Kippen 2001) and rotifers can dominate zooplankton communities in newly constructed ponds (Jenkins 1995). Together, small cladocerans and rotifers made up 78.3% of the total zooplankton abundance in a shallow Waikato lake (Greenwood *et al.* 1999). The dominance of rotifers in the campus lakes for much of the year concurs with work done in a shallow eutrophic lake, with dense blooms of *Microcystis aeruginosa* (Liu *et al.* 2002). It was found that the populations of rotifers and smaller cladocerans were favoured, as these taxa were thought to take best advantage of the "organic matter from *M. aeruginosa* through the detritus food chain" (Liu *et al.* 2002). Another possible reason for the dominance of rotifers may be the fact that they are less vulnerable to blockages of their feeding apparatus from suspended sediments, mechanical abrasion and from filamentous algae due to their morphology, particularly their small size (Shiel *et al.* 1982; Threlkeld 1986; Gliwicz 1990). Rotifers are able to selectively avoid clay particles in laboratory conditions and suspended sediment is not thought to inhibit rotifer population growth (Kirk & Gilbert 1990). High suspended sediment loads can decrease cladocerans' ingestion of some species of algae; however there is evidence that suggests nutrition can be gained by cladocerans from the biota attached to the inorganic particles in turbid reservoirs (Arruda *et al.* 1983).

In the two shallow lakes, cladocerans were dominant only during periods of increased water clarity. This could be due to their efficient filtration rates reducing the algal biomass or that they can only survive in clear water due to the restrictions discussed above.

The abundance of bacteria in the lakes could also influence the zooplankton community composition. There has been higher numbers of bacteria recorded in water bodies that are more eutrophic. Zooplankton that are able to utilize the bacterial cells for nutrition have an advantage over other taxa. Some species of *Brachionus* and *Filinia* have the ability to consume bacteria (Pejler 1983; Ooms-Wilms *et al.* 1995). Although it was not quantified in this study, it is very likely that there were significant bacterial populations in the campus lakes and this may have influenced the zooplankton community composition.

New Zealand zooplankton communities usually show little seasonality except for a general trend of lower total densities in cooler months. New Zealand's temperate climate and temperature regimes that allow year-round growth and reproduction mean that it is often the quality and quantity of food that dictates the zooplankton community composition (Chapman & Green 1987). In the campus lakes the summer peak of algae occurred after a large peak in the numbers of zooplankton in the two shallow lakes, which was somewhat unexpected (Figure 3.5). However, when the species composition was examined, the summer bloom of phytoplankton was dominated by *Microcystis* spp. which have been found to be largely inedible to many zooplankton taxa and provide limited nutrition if they are ingested (Boing *et al.* 1998). The summer peak of zooplankton was more likely to be influenced by the blooms of more edible taxa of phytoplankton e.g., *Trachelomonas* spp., *Fragilaria* spp., or *Aulacoseira* spp. Other research has found that blooms of phytoplankton are reduced initially by increased zooplankton grazing on edible forms of algae. Although there have been exceptions to this rule, with edible algae blooming in the presence of high densities of zooplankton (Agrawal 1998), blooms of inedible algae after intense predation have been well documented (e.g., Porter 1973; Haney 1987). The reduced algal competition allows the inedible algal taxa to thrive and reduce zooplankton predation rates through ingestion of toxic *Microcystis* cells (Ferrao *et al.* 2000), reduced foraging efficiency and the clogging of filter feeding apparatus (Gliwicz 1990).

New Zealand zooplankton can survive in a wide range of ecological conditions (Chapman & Green 1987). This is demonstrated in the campus lakes by the unclear distinction shown by the MDS ordinations between the three lakes compared to the strong distinction in the phytoplankton community (Figures 3.7 & 3.10). This is possibly an indication of the large range of food sources that the zooplankton taxa present were able to utilize and the large range of environmental conditions in the campus lakes that they were able to tolerate.

Zooplankton associations with environmental variables

The RDA ordinations of the zooplankton data showed that the amount of phytoplankton present was the greatest driver of zooplankton abundance in the campus lakes. This concurs with the 'quality and quantity of food' limitations discussed above (Chapman & Green 1987) and the higher zooplankton abundances observed in lakes that are more productive (Chapman *et al.* 1985). Food has also been shown to limit small cladocerans in a turbid lake (Greenwood *et al.* 1999). In New Zealand, increases in copepod and cladoceran abundances are mainly caused by reduced mortality as opposed to increased birth rates (Chapman & Green 1987). There may be fewer deaths from starvation in the campus lakes over summer, which may account for the population increases. In other shallow, eutrophic lakes, rotifer abundance has been linked with temperature and phytoplankton densities. Factors that controlled the primary productivity had the greatest influence on the macrozooplankton abundance in a shallow

Waikato lake (Miller 2002). Overseas, *Trichocerca pusilla* has been strongly related to the abundances of the diatom *Aulacoseira* spp. and temperatures over 12°C (May *et al.* 2001).

CONCLUSIONS

There were marked differences between the physical conditions of Chapel Lake and Oranga and Knighton lakes. The water quality was generally better in Chapel Lake, especially in summer. The physical differences were mirrored by significantly different phytoplankton and zooplankton communities. Oranga and Knighton lakes experienced significant summer blooms of zooplankton and phytoplankton. Chapel Lake did not. Cyanobacteria and Euglenoids dominated the phytoplankton communities in Oranga and Knighton Lake. Chrysophytes and Chlorophytes were dominant in Chapel Lake. Rotifers and small cladocerans dominated the zooplankton communities in all of the lakes. The differences in phytoplankton communities were more pronounced than the differences in zooplankton communities. It is likely that the different morphologies and mixing regimes of the three lakes (Chapter 2) strongly influences the water quality and plankton communities.

Urban lakes are valuable freshwater ecosystems and are an important point of contact with lakes for the general public. The campus lakes provide ideal research areas in terms of their location and ecosystem complexity. Further research should be strongly encouraged.

Chapter 4

Benthivorous Fish; Abundance and Distribution

INTRODUCTION

The abundance and diversity of fish communities in lakes has been shown to affect water quality and lake ecology in a number of ways. Fish can affect water quality through their predation on zooplankton (Jeppesen *et al.* 2000), which can subsequently reduce the grazing pressure on phytoplankton (Perrow 1999) and allow nuisance blooms to occur (Scheffer 1998). This is especially a problem in shallow lakes as the predation pressure of zooplankton by fish increases in lakes of decreasing depth. Top-down control by fish has been found to be more important in shallow lakes compared to deep lakes, unless there is a dense cover of macrophytes (Jeppesen *et al.* 1997).

Consumption of submerged macrophytes by fish can lead to macrophyte bed decline and eventual collapse (Williams *et al.* 2002). Selective herbivory by fish may also prevent the reestablishment of some macrophyte species in New Zealand (Lake *et al.* 2002). In one shallow lake in Hamilton, macrophyte removal by fish after herbicide application was thought to have lead to the lake switching between clear, marcrophyte dominated conditions to turbid algal-dominated conditions (Coffey *et al.* 1995). Fish can also contribute to nutrient regeneration in lakes through bioturbation and excretion (Cooke *et al.* 1993).

Shallow lakes generally contain more fish per unit volume than deeper lakes (Jeppesen 1998). In lakes of varying depths and nutrient status, fish biomass was negatively correlated with mean depth, with a 10-fold decrease in depth resulting in a 48-fold increase in fish biomass. It is possible that this was due to the different nutrient states of the lakes analysed (Jeppesen *et al.* 1997). There is a positive relationship between the fish biomass and the phosphorus concentration in shallow lakes. Also, freshwater lakes tend to have a higher fish biomass than comparable brackish lakes (Jeppesen *et al.* 1994).

Manipulation of fish communities

Fish communities can affect water quality and they are one of the easier variables to control when attempting lake restoration. Fish manipulations often provide tangible results to lake managers (Lammens 1999) and have been successful in restoring New Zealand lakes (e.g., Rowe and Champion 1994). Overseas, biomanipulations have caused marked reductions in nutrient concentrations and problematic algal populations (e.g., Meijer *et al.* 1994; Horpilla *et al.* 1998), and they are generally effective in lake restoration, especially with small, shallow lakes (Drenner & Hambright 1999; Jeppesen *et al.* 1990). There is a higher chance of success in lakes < 0.04 km^2 than in larger lakes (Reynolds 1994). For instance repeated, whole-lake fish removal has been shown to be effective in improving the water quality of shallow lakes (Kasprzak *et al.* 2002; Van de Bund & Van Donk 2002). However, they are not always successful in deeper, stratified lakes (e.g., Kasprzak 2002).

Benthivorous fish

The effects that fish have on water quality are particularly marked when benthivorous fish are dominant. The fish communities of shallow turbid lakes are often dominated by benthivorous fish (Scheffer 1998). These fish have been shown to have a detrimental effect on shallow lake ecosystems through sediment resuspension (Breukelaar 1994), nutrient recycling through excretion (Scheffer 1998), and macrophyte removal (Williams *et al.* 2002).

In small, very shallow lakes there has been a linear relationship recorded between the biomass of some species of benthivorous fish and the levels of suspended sediment and a positive correlation between fish biomass and chlorophyll *a*, total phosphorus, total nitrogen and silicate (Breukelaar 1994). Other benthivorous fish, such as brown bullhead catfish, have been shown to significantly increase the phosphorus loading to a water body under laboratory conditions (Keen & Gagliardi 1981). In some cases, benthivorous fish populations have caused dramatic and sudden decreases in water quality as their prey populations collapse and the amount of time actively foraging in the bottom sediment increases (Zambrano *et al.* 2001).

The removal of benthivorous fish has been shown to significantly reduce the volume of suspended sediment in lakes (Barton *et al.* 2000) and significantly reduce nutrient and chlorophyll *a* concentrations (Moss *et al.* 2002).

The general ecology of brown bullhead catfish in the Waikato (e.g., Patchell 1977; Wise 1990; Barnes 1996; Bannon 2002) and the life histories, diet, and behaviours of the shortfinned eel in New Zealand are well known (e.g., Ryan 1984; Sagar & Glova 1998; Jellyman *et al.* 1999; Glova & Jellyman 2000; Jellyman *et al.* 2002). However, there has been limited work on the ecology of these fish in Waikato shallow lakes.

Chapter aims

The aims of this chapter are:

1. To estimate the relative abundance of the benthivorous fish populations of three small, shallow, hypertrophic, urban lakes in Hamilton.
2. To determine if there is a difference in fish communities between the three lakes.
3. To determine aspects of the general ecology of the brown bullhead catfish (*Ameiurus nebulosus*) and shortfinned eel (*Anguilla australis*) in the lakes.
4. To relate any differences in fish abundance to lake morphology, ecological conditions, and fish access.

METHODS

Site description

As per Chapter 2

Background to lakes

As per Chapter 2

Fish Sampling

The benthivorous fish in the campus lakes were sampled with replacement seven times at approximately three monthly intervals every 3 months between July 2001 and October 2002. Fish were netted twice in each lake on two consecutive days in December. Fish were captured with fyke nets; 10 nets were set per lake on each sampling occasion. The same net locations were used for each sampling period (Figure 4.1). Coarse nets (25-mm mesh size) were alternated with fine mesh nets (5-mm mesh size) around the perimeter of the lakes. All nets were baited with permeable containers holding approximately 10 g of trout pellets and were set 1-2 m from the bank, running perpendicular out from the shore, with the wing at the near shore end. Nets were set mid to late afternoon and retrieved the following morning, approximately 18 - 22 hours later. Nocturnal nettings were performed as longfinned and shortfinned eels are nocturnally active (Sagar & Glova 1998; Glova & Jellyman 2000), as are many species of catfish (Bolliet *et al* 2001; Pohlmann *et al* 2001), including brown bullhead catfish in Hamilton (Wise 1990). Captured fish were anaesthetised using small volumes of a benzocaine solution in approximately 20 litres of water. Body length and wet weight were measured. Catfish above 200 mm and eels above 500 mm were tagged using a colour-coded anchor T-bar tag. The fish were allowed to recover and re-released into the lake at the approximate site of capture. The number of pre-tagged fish caught were recorded each sampling date.

The number of fish that were recaptured was used to estimate the total population size in the lakes using the Petersen mark-recapture method:

$$N = MC/R$$

where N is the total number of fish, M is the number marked, C is the total fish caught and R is the total fish recaptured. The assumptions for this formula are that there was no mortality of the marked fish, no tag loss, random mixing and no multiple catches of the same tagged fish (Ricker 1975).

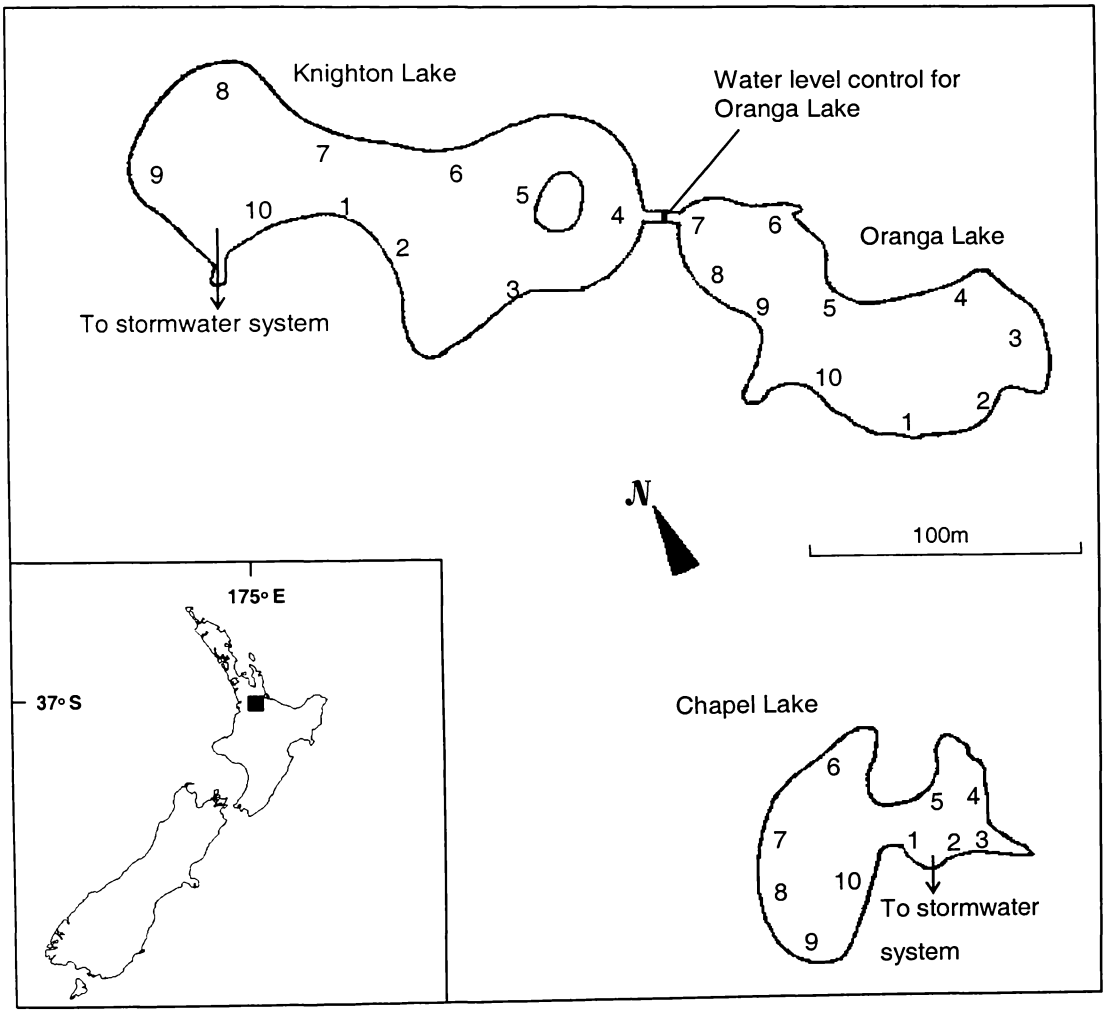


Figure 4.1: Location of fyke nets in the University of Waikato campus lakes

RESULTS

There were 2136 catfish and eels caught from all lakes, covering all sampling occasions. Brown bullhead catfish (*Ameiurus nebulosus*) were numerically dominant, followed by shortfinned eels (*Anguilla australis*). A small number of goldfish (*Carassius auratus*) were caught in Knighton and Oranga lakes and 1 rudd (*Scardinius erythrophthalmus*) was caught in Chapel Lake on one sampling occasion. A number of bullies (*Gobiomorphus cotidianus*) and mosquitofish (*Gambusia affinis*) were also caught. 357 fish were tagged in all lakes. The estimates of each lakes' combined adult catfish and eel populations ranged from 1093 to 26665 fish per lake (Figure 4.3). The highest recapture rate for any one sampling occasion was 7.5% for catfish and 11.1% for shortfinned eels. Recapture rates were usually < 5% (Table 4.1; 4.2).

There was a marked seasonal trend in catch rates in all three lakes. Most fish were caught in summer with declining catch rates as the months became colder. The two shallower lakes had significantly higher catch rates in summer compared to Chapel Lake There were significant differences between lakes on other sampling occasions. There were usually more catfish caught per lake than eels (Figure 4.2).

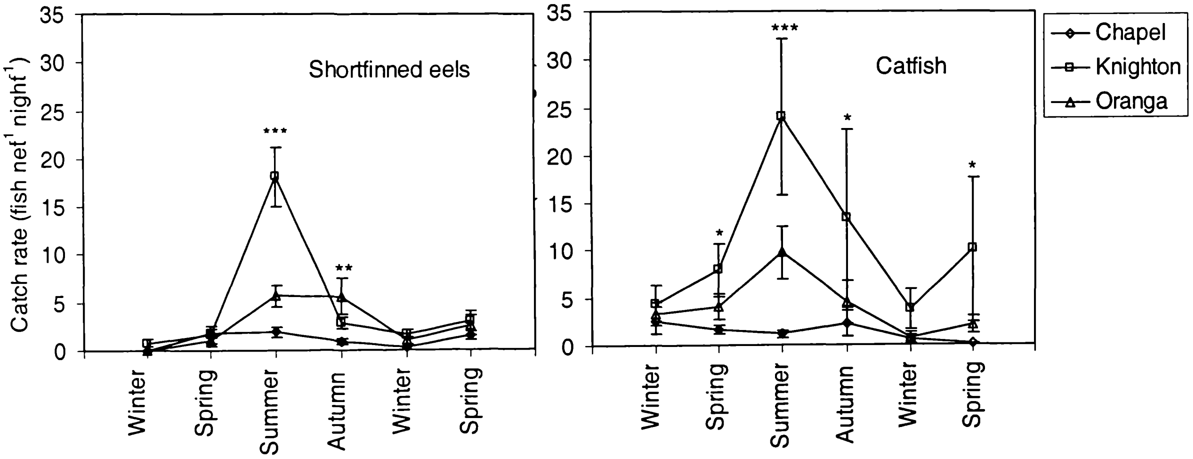


Figure 4.2: Catch rates of shortfinned eels and brown bullhead catfish in the campus lakes 2001 - 2002. Error bars are ± 1 S.E. *** = significant differences between all three lakes; ** = significant differences between Knighton and Chapel lakes only; * = significant differences between Oranga and Chapel lakes only ($P < 0.05$).

Table 4.1: Mark recapture rates for catfish in the campus lakes 2001-2002.

Date	Lake	Fish caught	Fish newly tagged	Cumulative fish tagged	Tagged fish recaptured	Recapture rate (% of all tagged fish)
5/07/2001	Chapel	26	26	26	n/a	n/a
5/07/2001	Oranga	64	14	14	n/a	n/a
20/06/2001	Knighton	53	25	25	n/a	n/a
2/10/2001	Chapel	16	14	40	0	0
1/10/2001	Oranga	40	26	40	0	0
1/10/2001	Knighton	78	32	55	2	8
19/12/2001	Chapel	16	1	41	3	7.5
17/12/2001	Oranga	141	41	81	0	0
17/12/2001	Knighton	318	51	106	3	5.3
20/12/2001	Chapel	4	2	43	2	4.9
18/12/2001	Oranga	50	6	87	0	0
18/12/2001	Knighton	162	22	128	2	1.9
27/03/2002	Chapel	23	0	43	0	0.0
26/03/2002	Oranga	34	0	87	0	0
26/03/2002	Knighton	133	19	147	1	0.8
3/07/2002	Chapel	6	0	43	0	0.0
2/07/2002	Oranga	7	0	87	0	0
2/07/2002	Knighton	37	18	165	2	1.3
2/10/2002	Chapel	1	0	43	0	0.0
1/10/2002	Oranga	21	0	87	1	1.1
1/10/2002	Knighton	100	0	165	1	0.6

Table 4.2: Mark recapture rates for shortfinned eels in the campus lakes 2001-2002.

Date	Lake	Fish caught	Fish newly tagged	Cumulative fish tagged	Tagged fish recaptured	Recapture rate (% of all tagged fish)
5/07/2001	Chapel	1	1	1	n/a	n/a
5/07/2001	Oranga	0	0	0	n/a	n/a
20/06/2001	Knighton	23	20	20	n/a	n/a
2/10/2001	Chapel	17	8	9	0	0
1/10/2001	Oranga	11	5	5	0	0
1/10/2001	Knighton	16	5	25	0	0
19/12/2001	Chapel	22	9	18	1	11.1
17/12/2001	Oranga	76	7	12	0	0
17/12/2001	Knighton	121	23	48	2	8.0
20/12/2001	Chapel	16	6	24	1	4.2
18/12/2001	Oranga	37	1	13	0	0
18/12/2001	Knighton	235	37	85	6	7.1
27/03/2002	Chapel	9	6	30	1	3.3
26/03/2002	Oranga	38	6	19	1	7.7
26/03/2002	Knighton	28	11	96	2	2.1
3/07/2002	Chapel	3	2	32	1	3.1
2/07/2002	Oranga	10	6	25	2	10.5
2/07/2002	Knighton	17	9	105	0	0.0
2/10/2002	Chapel	15	0	32	1	3.1
1/10/2002	Oranga	26	0	25	1	4
1/10/2002	Knighton	30	0	105	0	0.0

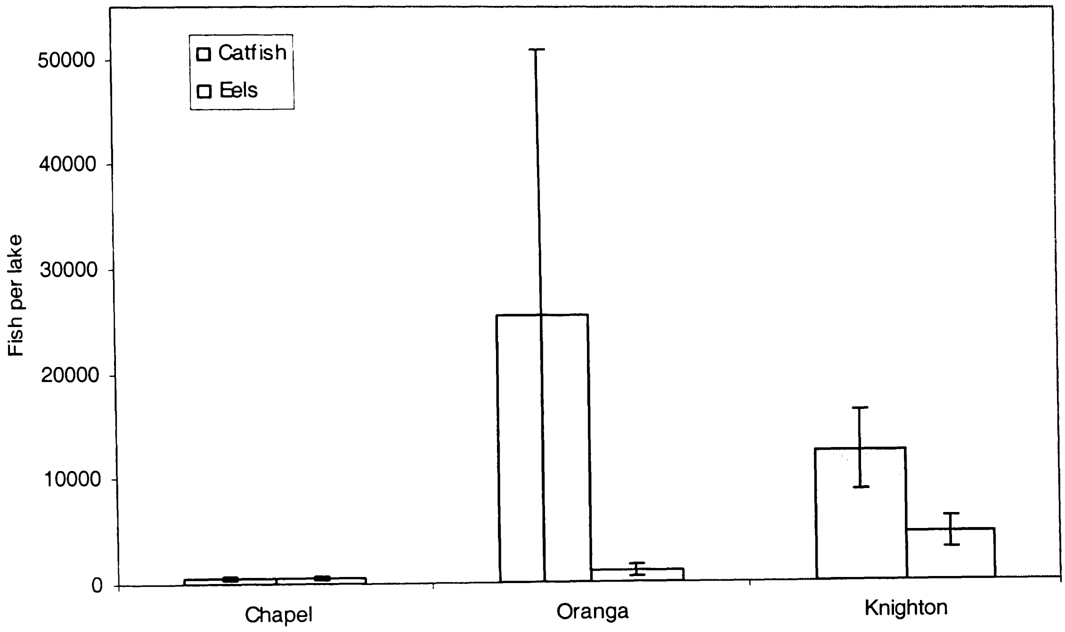


Figure 4.3: Population estimates of catfish > 200 mm and eels > 500 mm in the University of Waikato campus lakes using the Petersen mark-recapture method. Error bars are ± 1 S.D.

The catch rates of the common bully (*Gobiomorphus cotidianus*) were recorded in summer 2001. Bullies were only caught in fine-mesh fyke nets, so only these nets were included in analyses. On average there were 50.6 fish net^{-1} night^{-1} in Chapel Lake. This was significantly higher ($P < 0.001$) than the catch rates in Oranga Lake and Knighton Lake, which had catch rates of 1.41 and 0.84 fish net^{-1} night^{-1} respectively (Figure 4.4).

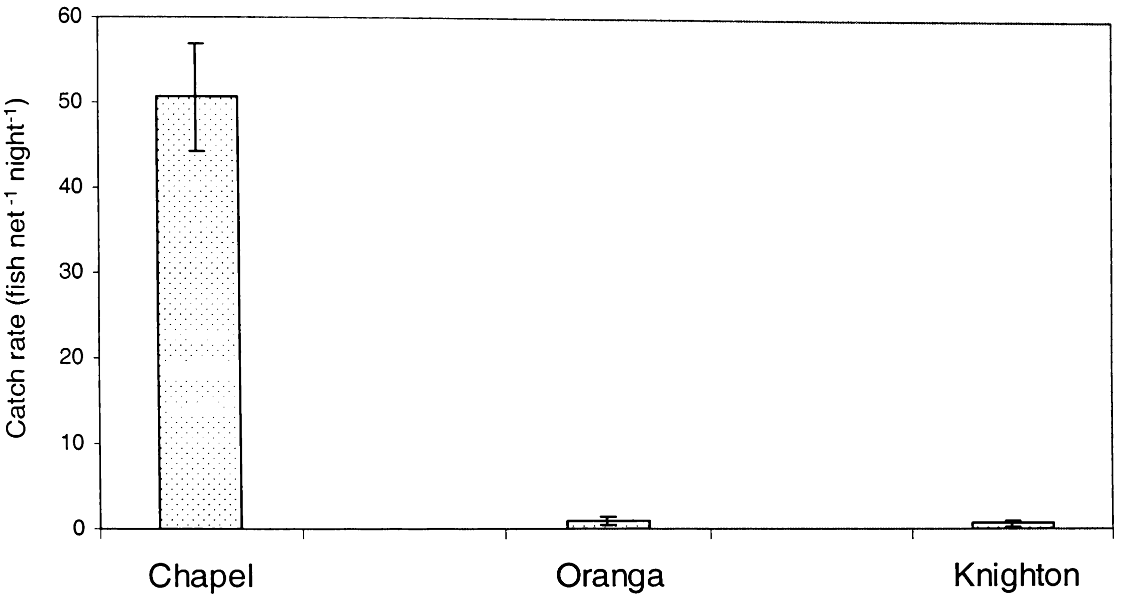


Figure 4.4: Relative catch rates of *Gobiomorphus cotidianus* in the University of Waikato campus lakes in December 2001. Error bars are ± 1 S.E.

Catfish up to 482 mm long (≥ 6 years old (Patchell 1977)) were caught in the lakes. Distinct age classes were evident from length frequency distributions in Knighton and Oranga Lakes but not in Chapel. This may be due to the low catch rates in Chapel. Young of the year were caught in October 2001, but were absent in October 2002. The majority of catfish that were caught in 2001 were within the size class of fish aged 1 (Patchell 1977). The mean fork length for this age class increased over the sampling period. The majority of catfish caught in October 2002 were within the size class of fish aged 2 (Patchell 1977) (Figure 4.5). Over the entire sampling period there were a total of 975, 326, and 84 catfish caught in Knighton, Oranga, and Chapel lakes respectively.

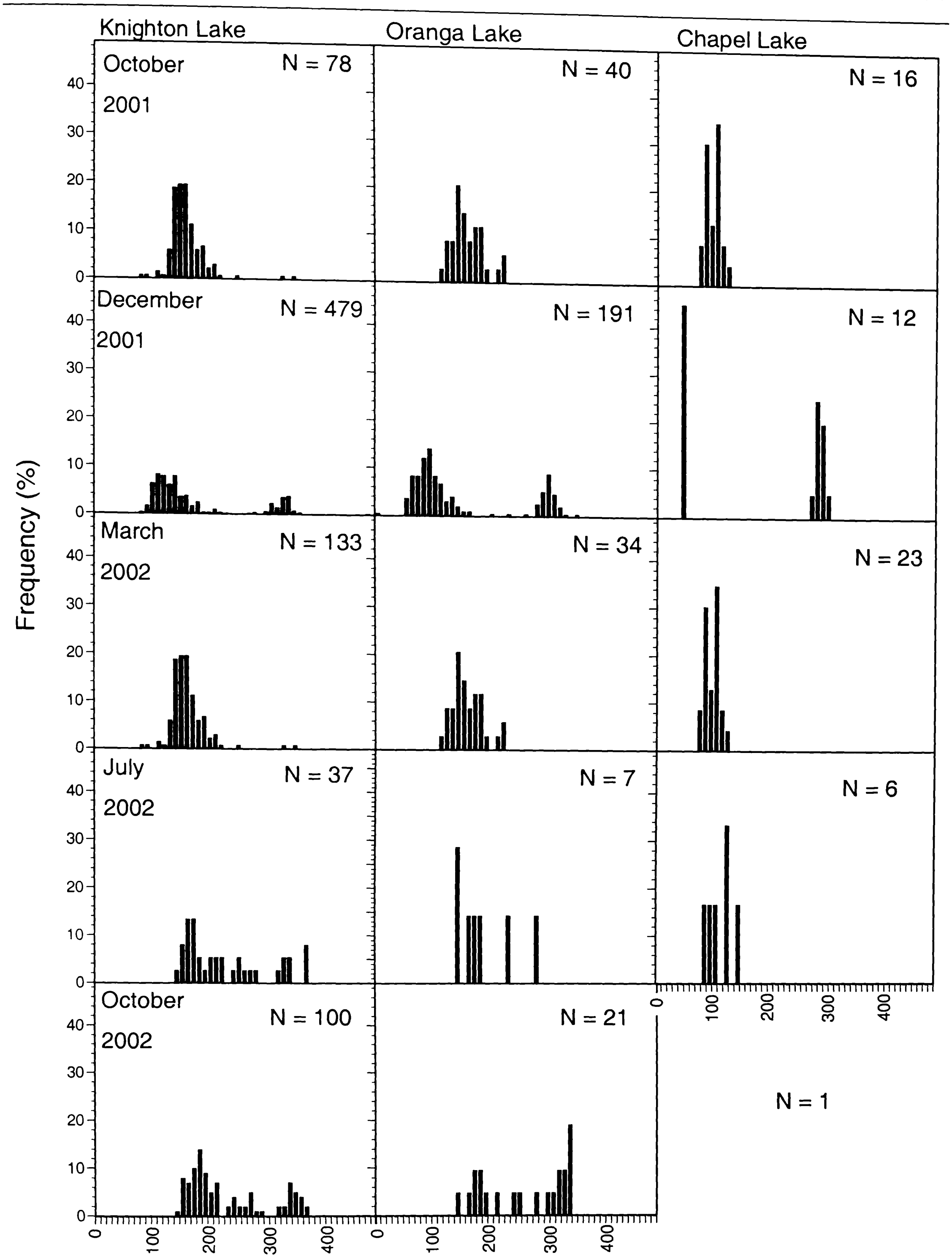


Figure 4.5: Size class distribution of brown bullhead catfish in the University of Waikato campus lakes 2001-2002. N = number of fish caught on each sampling occasion. Too few fish were caught in Chapel Lake in October 2002 to allow meaningful analyses.

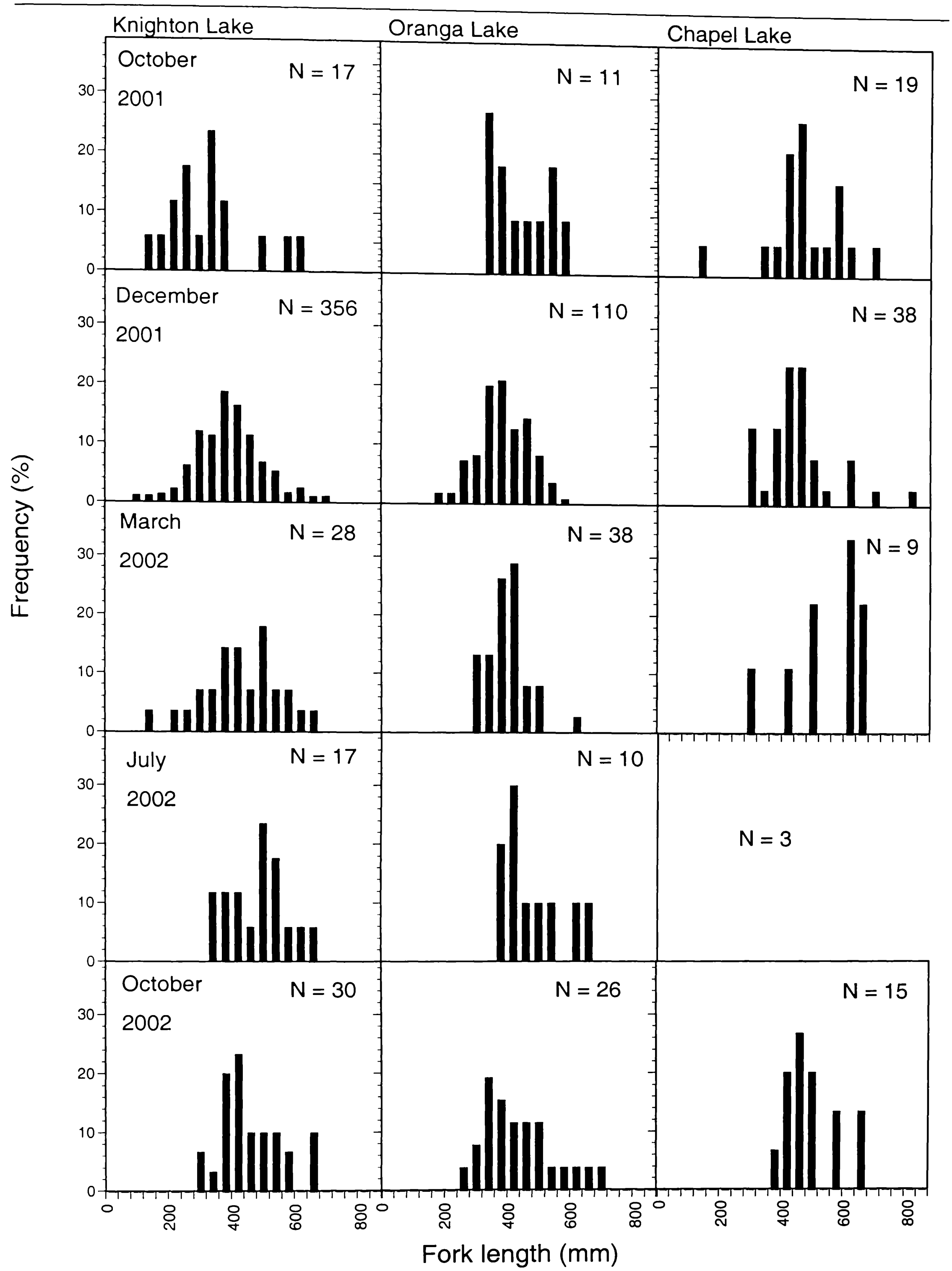


Figure 4.6: Size class distribution of shortfinned eels in the University of Waikato campus lakes between October 2001 and October 2002. N = number of fish caught on each sampling occasion. Too few fish were caught in Chapel Lake in July 2002 to allow meaningful analyses.

Shortfinned eels were caught in all three lakes on all sampling occasions. They ranged in size from 96 mm to 805 mm. There were no obvious age class distinctions in the eel populations. The length frequencies of short finned eels were normally distributed. Small eels were caught in December 2001 in Knighton Lake and Oranga Lake, but not in Chapel Lake (Figure 4.6). Over the entire sampling period there were a total of 471, 195 and 85 shortfinned eels caught in Knighton, Oranga and Chapel lakes respectively.

The weights of the catfish were significantly related to their lengths in all lakes on all sampling occasions ($P < 0.001$; $r^2 \geq 0.99$). The slope of the regression line varied between lakes and between seasons. In Oranga and Knighton lake there was a steep drop in the length-weight regression slope between December 2001 and March 2002 and a gradual increase between all other months. Knighton usually had heavier fish than Oranga Lake. Chapel Lake catfish exhibited a greater range of length-weight relationships than catfish from the other two lakes and experienced a steep drop in relative weight between June and October 2001. Generally there was an increasing trend in relative weights in all lakes over the sampling period (Figure 4.6).

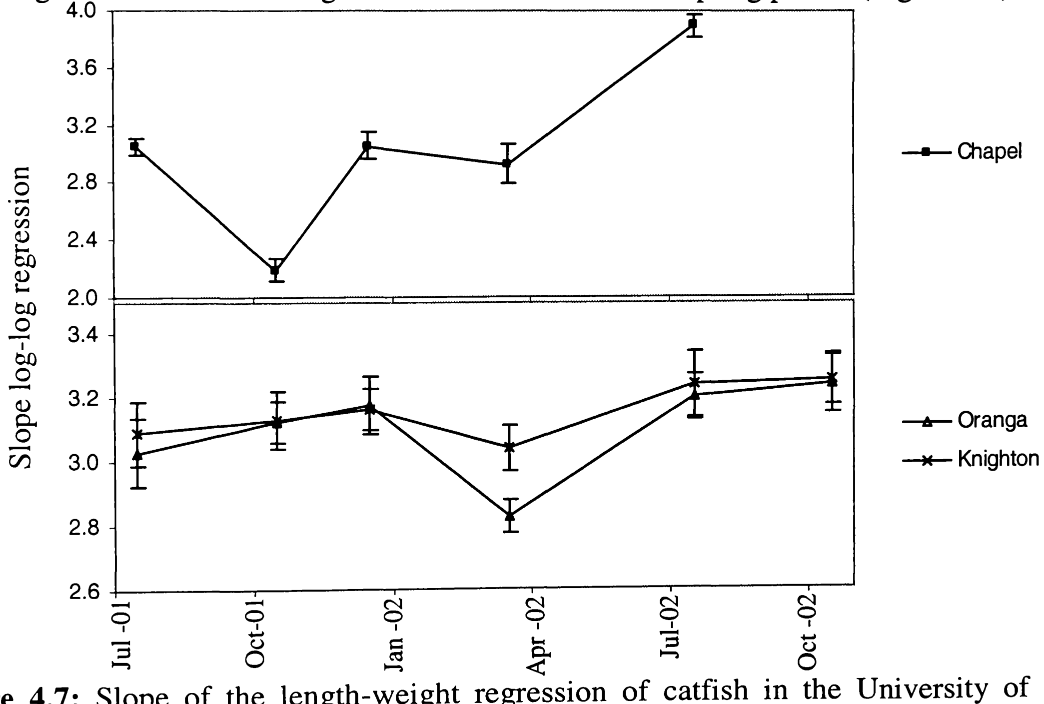


Figure 4.7: Slope of the length-weight regression of catfish in the University of Waikato campus lakes after natural log transformation of the data. Note the different scale for the Chapel Lake samples. Error bars are ±1 S.E.

DISCUSSION

The higher catch rates of catfish and eels in the Oranga and Knighton lakes were likely to primarily be a function of the lakes' habitat and production. These two lakes were very shallow, highly productive, had unstable bottom sediments, and had stands of macrophytes for part of the year, which provided ideal habitat for catfish to live and breed (McDowall 1990). In contrast, Chapel Lake was less productive, had fewer marginal areas, had no macrophytes, and was less suited to large catfish populations.

Macroinvertebrate abundance is strongly influenced by the amount of primary production in lakes (Plante & Downing 1989) and shallow lakes tend to have a higher density of benthic invertebrates than deeper lakes (Jeppesen *et al.* 1997). Oranga and Knighton lakes had significantly higher productivity than Chapel Lake and they also had more light available at the lake bed (Chapter 3). Therefore it is very likely that there was a greater biomass of macroinvertebrates in Oranga and Knighton lakes than in Chapel Lake, although this was not quantified in this study. Catfish are opportunistic feeders, but the majority of their diet is made up of benthic macroinvertebrates (Barnes 1996). Small shortfinned eels (< 300 mm) have similar diet preferences (Sagar & Glova 1998). The higher number of benthic macroinvertebrates may have allowed larger fish populations to become established and in turn caused the higher catch rates in these two lakes. This concurs with other work, where the lower benthic productivity of deeper lakes is thought to limit the effects of benthivorous fish (Zambrano *et al.* 2001).

The abundance of eels in the campus lakes may have also been controlled by the degree of fish access. Oranga and Knighton lakes had direct links to the Hamilton stormwater system and therefore the Waikato River. Although there was a separate stormwater outlet draining Chapel Lake, there may have been barriers within the stormwater system that prevented elver migration to Chapel Lake, but not Oranga and Knighton lakes. The presence of small shortfinned eels in Oranga and Knighton lakes in December 2001 and their absence from Chapel Lake supports this (Figure 4.6). Young shortfinned eels are diadromous (McDowall 1990) and it is possible that the small eels present in Knighton and Oranga lakes accessed the lakes from the

Waikato River, via the stormwater system. The majority of shortfinned elvers (67%) migrate up-river in the months of September and October (Jellyman 1999; Jellyman 2002) and small eels were caught in the campus lakes in December 2001. It is credible to assume that eels were able to navigate into two of the campus lakes and it takes approximately two months to for the peak of elvers to reach the lakes from the coast. Eels may have been able to colonise Chapel Lake through migration overland, migration from the other lakes, or artificial introduction. The absence of small eels and the consistently low catch rates indicate that this is not a regular occurrence.

It is possible that low levels of dissolved oxygen in bottom of the water column (Chapter 2) limited the distribution and feeding range of catfish and eels in the lakes at certain times of the year. However, the lethal oxygen concentration for catfish at 25°C was calculated at 0.95 ppm (Moss and Scott 1961), with lower lethal values recorded as temperatures decrease (Scott and Crossman 1973). Eels can also survive at very low oxygen levels (Dean & Richardson 1999). The oxygen in lakes was never measured below the lethal concentrations for catfish or eels in the study. Also all three lakes experienced periods of low oxygen in the bottom waters (Chapter 2), yet their relative catch rates were significantly different.

The significantly lower catch rates of bullies in the two shallower lakes compared with Chapel Lake may be a result of increased predation in these two lakes. While smaller shortfinned eels (< 300 mm) feed on various invertebrates (Sagar & Glova 1998), shortfinned eels above 500 mm are known to feed almost exclusively on fish (Ryan 1986). It is likely that the high numbers of large shortfinned eels had a significant effect on the Oranga and Knighton lakes' bully populations. The lesser densities of large shortfinned eels in Chapel Lake would not have preyed upon the bully population to the same degree. It is not likely that catfish preyed upon bullies to a large extent. Although fish have been found in the stomachs of many size classes of catfish, they do not make up a significant part of their diet (Barnes 1996). The year-round presence of macrophytes in Chapel Lake may also have influenced the higher number of bullies caught. Bullies can spawn on the stems of aquatic weeds (McDowell 1990) and it is possible the large stands of water lilies (*Nymphea odorata*) and the littoral stands of rushes (*Baumea articulata*) allowed spawning of bullies in Chapel Lake to a greater extent than the other two lakes.

The marked seasonal variation in catfish catch rates was in contrast with Barnes (1996) who recorded high catch rates in early summer but reasonably even catch rates for the rest of year. The lower catch rates as winter progressed were possibly related to the decrease in movement of the catfish due to the decrease in temperature, the onset of breeding and the territorial behaviour of fish (Patchell 1977). Shortfinned eels have also been shown to exhibit little or no feeding activity in winter (Ryan 1986), which is likely to have affected their low catch rates for the two winter sampling occasions.

The average summer 2001 catfish catch rates were 0.03, 0.53 and 1.33 fish net^{-1} h^{-1} in Chapel, Oranga and Knighton lakes respectively These were markedly lower than was found in a similar sampling regime in summer 1996 that caught 1.42 fish net^{-1} h^{-1} in Chapel Lake, 5.12 fish net^{-1} h^{-1} in Oranga Lake and 5.26 fish net^{-1} h^{-1} in Knighton Lake (Willis 1996). The lower catch rates were possibly due to intensive catfish removal in the late 1990s or an indication of poorer conditions for catfish survival and reproduction. The catch rates of catfish in the campus lakes were generally within the ranges reported by Barnes (1996) who recorded catch rates over a year between zero and 92.1 fish net $^{-1}$night^{-1} in Lake Taupo in a range of habitats. However, they were well below the catch rates of over 100 fish net $^{-1}$night^{-1} reported in Waikato farm drains (Bannon 2001). The high numbers of catfish aged 2+ is similar to the dominance of juvenile catfish previously documented in the campus lakes (Willis 1996), in Waikato farm drains (Bannon 2001) and Lake Taupo (Barnes 1996) and similarly the catfish population is likely to be increasing in the campus lakes. This agrees with Johnson's (1994) theory that exploited fish populations will have juvenile fish dominating the length frequency distributions. Conversely, in unexploited populations, the dominance of larger fish will cause a reduction in the food resources and nesting territories available for smaller fish and the larger fish will be slowly and gradually replaced as they die. The catfish populations of the campus lakes seem to still be recovering from the intensive removal of catfish in the late 1990's and occasional recreational fishing that selectively removes larger fish. The presence of distinct age classes and the presence of young of the year fish indicate that catfish are breeding in the campus lakes and have a self-sustaining population.

The largest catfish recorded from the Chapel, Oranga, and Knighton lakes were 373 mm fork length (F.L.), 386 mm F.L. and 482 mm F.L. respectively. These catfish were all larger than any catfish caught in Lake Taupo (359 mm F.L.) (Barnes 1996) or Waikato farm drains, where catfish did not exceed 385 mm F.L. (Bannon 2001) and the largest catfish was longer than any recorded in the Waikato (455 mm F.L.); (Patchell 1977). These data are an indication that the conditions in Oranga and Knighton lakes and to a lesser extent Chapel Lake were conducive to the long-term growth and survival of catfish.

The marked decline in relative weight experienced by the catfish between the December 2001 and the March 2002 sampling is likely to be a result of summer spawning, increased competition from the emerging young of the year and possibly through the decreased foraging time that would result from nest guarding by males (McDowall, 1990). The likely spawning of the catfish occurring between December and March agrees with the schools of young of the year catfish (approximately 10 mm F.L.) seen in the margins of Knighton Lake in January 2003, and the occurence of catfish eggs from Knighton Lake in December 2001. It is slightly later than reported in Lake Taupo (Barnes 1996) where catfish were estimated to spawn between September and December but agrees with spawning in December-January in Waikato farm drains (Bannon 2001). The lower relative weights of the catfish in Oranga Lake compared to the catfish in Knighton Lake are possibly an indication of the lower quality of the habitat in Oranga Lake.

There were a number of shags present on the lakes for many winter months and on several occasions they were observed to predate juvenile catfish (~ 150 mm) (personal observation). A Swedish study on the effect of piscivorous birds on the fish population of a shallow lake found that bird predation played a significant role in loss of fish 100 - 140 mm in length, although this may have been a related to the size classes of fish available in the lake (Winfield 1990). Avian predation may have caused a reduction in numbers of the smaller size classes of catfish in the campus lakes. It is likely that other factors such as disease or recreational angling also affected the numbers of the larger fish.

The population estimates obtained are only approximations of the actual number of fish in the lakes. This is due to the low recapture rates and the possible violation of the assumptions made in the Petersen formula. For example, it is likely that fish mortality and migration occurred in the lakes. If this occurred, the population estimates were over-estimates of the actual abundances. Conversely, there were a large number of smaller fish that were not included in the tagging experiments, which are likely to make the results under-estimates of the total fish populations in the lakes. As there was only one tagged catfish recaptured from Oranga Lake, there is a large margin of error for this population estimate and few conclusions can be drawn about the lake's total catfish population size.

Study Design

The results of the fish mark-recapture experiments and the catch rates allow some comments to be made on possible improvements to the design of this study. The very low winter catch rates in all three lakes, despite intensive netting efforts, meant that there were few fish tagged in the months leading up to summer 2001/02, when the most fish were caught. The subsequent low recapture rates were therefore to be expected. A possible improvement to the fishing regime would involve intensive summer fishing with more fyke nets set per lake per night and several consecutive sampling occasions for each lake. The higher summer catch rates would result in more fish being tagged and higher recapture rates. The value of any winter fishing is limited if data for population estimates are needed.

To maximise fish survival, catfish below 200 mm and eels below 500 mm were not tagged. The dominance of fish in these size classes meant that a large proportion of the fish that were caught were not tagged. A smaller, less invasive method of tagging that allowed the tagging of smaller fish may have been more successful in obtaining better mark-recapture rates.

CONCLUSIONS

The benthivorous fish populations of the campus lakes were dominated by brown bullhead catfish (*Ameiurus nebulosus*) and shortfinned eels (*Anguilla australis*). The conditions in the three lakes were suitable for the long-term growth and reproduction of catfish, although Chapel Lake provides less suitable catfish habitat. The catch rates recorded in this study were less than reported from the same lakes in 1996 and the populations are either below the lakes' carrying capacity or the lakes' carrying capacity has changed. The populations have the potential to increase in spite of losses through disease, predation and recreational angling. The two shallow lakes support larger catfish and eel populations than the slightly deeper one. This is possibly a function of higher productivity in the two shallow lakes. It may also be related to habitat and fish access. The conditions in Chapel Lake appear to be better suited than Oranga or Knighton lakes to populations of bullies. This may be related to predation pressure or the number of spawning sites.

There was a marked seasonal variation in catch rates. This is possibly a result of lower temperatures, territorial behaviour and breeding. Catfish spawning appears to occur between December and March.

Migration by small shortfinned eels was possible into Oranga and Knighton lakes. There is approximately two months between upstream elver migration at the coast and their appearance in the campus lakes. There was no eel migration recorded into Chapel Lake.

Chapter 5

Thesis Conclusions

GENERAL CONCLUSIONS

The greater depth of Chapel Lake appears to protect it from the most extreme effects of hypertrophy experienced by Oranga and Knighton lakes. In spite of similar nutrient regimes, climatic conditions, and physical location, the two shallower lakes had poorer water quality and higher zooplankton and phytoplankton densities compared to the deeper lake. This was especially pronounced in summer. High inorganic suspended sediment concentrations and algal biomass combined to create lower water clarity in Oranga and Knighton lakes than Chapel Lake.

The mixing regimes were significantly different in all three lakes. On average Oranga and Knighton lakes were less stable than Chapel Lake. The stability of the water columns in all three lakes was primarily controlled by diurnal changes in air temperature. The lakes were generally less stable at night than during the day and were less stable in winter than in summer.

Oranga and Knighton lakes had significantly different zooplankton and phytoplankton communities compared to Chapel Lake. The differences in phytoplankton communities were more pronounced than the differences in zooplankton communities. Cyanobacteria and Euglenoids dominated the phytoplankton communities in Oranga and Knighton lakes. Chrysophytes and Chlorophytes were dominant in Chapel Lake. Rotifers and small cladocerans dominated the zooplankton communities in all of the lakes. Oranga and Knighton lakes also experienced marked blooms of phytoplankton and zooplankton in summer, which were not observed in Chapel Lake. The differences between the lakes may be related to the intense mixing that occurred in the two shallower lakes but did not occur to the same degree in Chapel Lake.

All three lakes provided suitable habitat for the long-term growth and survival of brown bullhead catfish and shortfinned eels. However, the increased productivity in the shallower lakes appeared to support larger populations of catfish and eels than in Chapel Lake. Conversely, the relative abundance of common bullies was greater in Chapel Lake than in Oranga and Knighton lakes. This may have been related to predation pressure and spawning sites.

The work presented in this thesis concurs with overseas studies that compared shallow and deep lakes. Dramatic summer blooms of zooplankton and some phytoplankton taxa have been recorded in one shallow lake, which did not occur in a nearby deep lake of similar water chemistry (Agbeti *et al.* 1997). The lakes used in the previous study differed in depth by 35.5 m. When the difference in depth was less pronounced, similar trends have been recorded. Very shallow, highly eutrophic lakes are more effective than moderately shallow or deep lakes at converting similar amounts of phosphorus into phytoplankton biomass. In one study of lakes of similar nutrient concentrations but differing morphologies, total phytoplankton biomass was approximately double in the very shallow lakes when compared to the moderately shallow and deeper lakes. (In this case, very shallow lakes were defined as having a maximum lake depth, z_{max}, that was less than the depth of the theoretical epilimnion predicted by lake area, z_{epi}; moderately shallow lakes had $z_{max} \geq z_{epi}$, and deep lakes have $z_{max} >> z_{epi}$). This was probably due to the influence that the lakes' morphology had on their mixing regime, which created favorable conditions for algal growth. Regular mixing in shallow lakes can allow algal production to approach the physiological maximum (Nixdorf & Denke 1997). Shallow, well-mixed lakes generally have higher summer concentrations of phosphorus in their epilimnion than deeper, stratified lakes (Jeppesen *et al.* 1997). On a smaller scale, very shallow lakes (< 3 m depth) generally have higher phosphorus concentrations in their photic zone compared to slightly deeper (> 3 m depth) lakes of similar nutrient status. Shallow lakes are also more effective at converting high phosphorus concentrations to higher maximum chlorophyll levels than deeper lakes. In study of 142 eutrophic Danish lakes in summer, the algal biomass was inversely related to depth, but not as strongly influenced by the phosphorus concentrations (Scheffer 1998).

IMPLICATIONS FOR LAKE MANAGEMENT

To improve the water quality of Oranga and Knighton Lakes, artificial deepening is the option that will provide the longest lasting improvements in water quality. Deepening will reduce the amount of sediment resuspension, remove the nutrient-rich top layer of sediment, and possibly reduce the size of the benthic fish populations. These factors would combine to reduce the intensity of the summer plankton blooms that the two shallower lakes currently experience. It is also unlikely deeper lakes would experience temperature induced mixing throughout the entire water column that Oranga and Knighton lakes currently experience.

Deepening the lakes is likely to prevent summer blooms of *Potamogeton crispus* as there will be less light available to macrophytes at the bottom of the lakes. Deepening is also likely to also reduce the densities of ducks that inhabit the lakes. Duck numbers reach over 200 ducks lake^{-1} in Oranga and Knighton lakes, while there were significantly lower numbers in Chapel Lake. This was most pronounced in hunting season (Hicks & Bryant unpub. data). Deepening the lakes will reduce the area of lakebed that is available for foraging to the ducks and also remove the potential food source of *Potamogeton crispus*. Bird excretion is known to make up a significant proportion of nutrient inputs to lakes (e.g., Manny *et al.* 1994; Coffey *et al.* 1995; Hobman 2002) and reducing duck numbers will reduce the total nitrogen and phosphorus inputs to the campus lakes. This may contribute to an improvement in water quality.

The lakes should be deepened to a minimum depth of 1.5 m. This will increase the wind speed required for sediment resuspension in the lakes to ~ 6 m s^{-1} (Scheffer 1998) which is well above the average wind speed at the Ruakura climate station in 2001 (Chapter 2). This is also deeper than the critical depth of sediment resuspension given in Yanful & Catalan (2002). The probability of *Potamogeton pectinatus* occurring in eutrophic lakes decreases dramatically below 1 m depth (Scheffer *et al.* 1992) and it is likely that a depth of 1.5 m will severely limit the growth of the *Potamogeton crispus* in the campus lakes.

The lakes were algal-dominated for most of the year. Deepening the lakes is not likely to alter this state as it will reduce the light available to macrophytes at the lake bottom. Also lakes that have very high nutrient concentrations are less prone to switch between algal-dominated and macrophyte-dominated states (Scheffer 1990; Scheffer *et al.* 1993; Moss 1998). However, after deepening it is likely that the total algal biomass will be reduced in the lakes, especially during summer and there will be an improvement in water quality. Macrophyte blooms have been seen as being indicative of poor water quality in comparable urban lakes (Miller 2002), so the general public may even perceive the algal dominance as desirable.

The catfish populations of the lakes appear to be still recovering from intensive removal in the late 1990s. Although this removal has failed to prevent significant algal blooms, it is likely that the absence of benthivorous fish would contribute to better water quality in the lakes. There were still large populations of catfish resident in Oranga and Knighton lakes during this study, which may have influenced water quality. Deepening the lakes may reduce primary production, which would probably reduce the density of benthic macroinvertebrates. This would limit the numbers of catfish.

RECOMMENDATIONS FOR FUTURE WORK

There are some aspects of the physical characteristics and ecology of the campus lakes that were not studied in this thesis. These warrant further research.

Macroinvertebrate communities were not investigated in this study. It is likely that they play a major role in the ecosystems of the lakes, acting as consumers of plankton and as prey for fish. There is work to be done on the macroinvertebrate diversity and abundance in the lakes.

The low taxonomic resolution of some groups of plankton limited the conclusions that could be drawn. There is still work to be done detailing aspects of the zooplankton and phytoplankton diversity in the lakes.

The fishing methods used (fyke nets) targeted the benthic populations of fish. There were sizable populations of pelagic goldfish (*Carassius auratus*) observed in the lakes, which were not caught in this study. Different fishing methods (e.g., gill nets) may be more effective in sampling these populations. Any future studies on the fish populations of the campus lakes need to focus their efforts on intensive summer sampling. It is possible that a higher density of nets per lake would improve catch rates. Low catch rates in the colder months limit the value of sampling during this time.

The sampling frequency in this study ranged between weeks and months. It is possible that there are strong variations in water quality, plankton abundance and nutrient concentrations that occur over a 24-hour cycle. The ecosystem response to the strong diurnal mixing regimes in the lakes warrants further investigation. In particular, vertical and horizontal migration of zooplankton and phytoplankton over short time scales have been recorded in other systems and it would be of interest to study this in the campus lakes. Also, the release of suspended sediment and nutrients could be studied during periods of intense mixing.

Should actions be undertaken in an attempt to improve lake quality, the data presented in this thesis would form the basis of a before and after study. A similar study after restoration work would be of significant scientific value. The amount of existing data and the location and diversity of the campus lakes' ecosystems provide an excellent opportunity to add to the limited information on urban lakes and artificial water bodies. Future studies of the campus lakes should be strongly encouraged.

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APPENDIX 1A

List of phytoplankton taxa identified from the University of Waikato campus lakes.

Cyanobacteria

Anabaena spp.
Lyngbya spp.
Merismopedia spp.
Microcystis spp.

Chlorophyta

Ankistrodesmus spp.
Closterium acutem var. *linea*
Closterium acutem var. *variabile*
Closterium gracile
Closterium sp. c.f. *leibleinii*
Coeleastrum cambricum.
Cosmarium spp.
Kirchinella spp.
Mougeotia spp.
Paedeastrum duplex.
Quadrigula spp.
Staurastrum spp.
Schenedesmus spp.
Schenedesmus obtusus.
Scenedesmus acuminatus
Spirogyra spp.
Tetraspora spp.
Ulothrix spp.

Bacillariophyta

Aulacoseira granulata
Aulocoseira granulata var. *angustissima*
Cyclotella spp.
Cymbella spp.
Epithemia spp.
Fragilaria spp.
Gomphonema spp.
Nitzchia spp.
Pinnularia spp.

Chrysophyta

Dinobryon sertularia
Synura spp.

Euglenophyta

Euglena sanguinea
Euglena spp.
Phacus spp.
Tracelomonas volvolcina
Trachelomonas hispida var *coronata*
Trachelomonas dubia
Trachelomonas spp.

Pyrrhophyta

Cerratium spp.
Perdinium spp.

APPENDIX 1B

List of zooplankton taxa identified from the University of Waikato campus lakes.

Cladocera

Bosmina spp
Chydoridae c.f. *Biaptura* spp.
Chydoridae c.f. *Chydorus* spp.
Daphnia sp. c.f. *pulex*

Rotifera

Asplanchna spp.
Brachionus quadridentatus
Brachionus angularis
Brachionus spp.
Filinia spp.
Hexarthra spp.
Keratella tecta
Keratella tropica
Keratella procurva
Keratella spp.
Lophocanthous spp.
Lecane flexilis
Lecane spp.
Polyarthra spp.
Trichocerca spp.

Copepoda

Calanoid
Cyclopoid