Lake Rotokakahi: The kakahi *(Hyridella menziesi)* in a general framework of lake health.

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Abstract

Lake Rotokakahi is a mesotrophic lake located within the Rotorua Lakes District, North Island, New Zealand. Under the legal guardianship of the Tuhourangi and Ngati Tumatawera tribes of Te Arawa it has remained closed to the public since 1948.

Lake Rotokakahi was last monitored regularly in 1996 under the Environment Bay of Plenty (EBOP) water quality monitoring programme with only the lake outlet (Te Wairoa Stream) being monitored since that time. Water quality data collected up to 1996 suggests that there may be degradation of water quality in the lake, as indicated by declining levels of dissolved oxygen in the bottom waters.

Lake Rotokakahi steeped in historical significance, as well as having major cultural and recreational values was well known for its abundant resources and as the name suggests, particularly for its massive supply of the freshwater mussel or kakahi (*Hyridella menziesi*). Freshwater mussel species worldwide are in decline however little is known on factors controlling kakahi abundance and distribution.

The overarching objective of this thesis is update water quality data last monitored in Lake Rotokakahi in 1996 while also identifying key environmental variables thought to influence kakahi populations. This objective is underpinned by a number of aims that include:

- Establishment of a 12-13 month water quality programme within Lake Rotokakahi so that data is obtained for comparisons with previous water quality data (pre-1997) to allow assessment of whether there have been water quality changes in the lake.
- Conduct a population survey of the resident kakahi population examining possible environmental factors influencing their populations.
- Present an oral history of Lake Rotokakahi focusing on its historical significance, water quality, and collection of kairoto (food collected form lakes).

from the 18 September 2006 to 14 September 2007 monthly water sampling was carried out at a mid lake station, the lake outflow and inflow for measures of nutrients, phytoplankton, zooplankton and chlorophyll a. Vertical profiles of temperature dissolved oxygen and chlorophyll fluorescence were also taken on various part of Lake Rotokakahi.

A lengthened period of anoxia in the bottom waters during thermal stratification was recorded with increased levels of chlorophyll *a* in winter and reduced secchi disk depth indicating an increase in phytoplankton biomass. Nutrient concentrations remain moderate relative to historical data. On going water quality monitoring on Lake Rotokakahi is recommended to further evaluate the extent of which water quality change is occurring. This will provide a better understanding of how Lake Rotokakahi can be best managed to further preserve the lake.

On 1 March and 20 April kakahi were sampled at five sites. Large kakahi densities were distributed predominantly in depths above the hypolimnion. Chlorophyll *a* fluorescence and dissolved oxygen were found to be the best correlates for kakahi density and biomass respectively. Low dissolved oxygen concentrations in the hypolimnion are thought to restrict kakahi distributions to above the thermocline in periods of hypolimnetic anoxia.

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Chapter 1

Introduction

1.1 Introduction

1.1.1 Eutrophication

Eutrophication of freshwaters is an increasingly global environmental issue (Lee *et al,* 1978; Smith, 2003; Bronmark and Hansson, 2002; Wetzel, 2001; Rast and Thornton, 1996; Sondergaard and Jeppesen, 2007). It is a natural process, occurring as lakes become progressively shallower and more productive over time, eventually filling with sediment and organic matter to become dry land. Natural eutrophication usually occurs over geological time scales however, it is made much more rapid as a result of impacts from human activities (i.e. cultural eutrophication). Excessive nutrient inputs to lakes, particularly phosphorus and nitrogen, can cause nuisance blooms and surface scums of algae, deoxygenation of bottom waters and in more severe cases, fish kills and deoxygenation of the entire water column (Hamilton, 2007). During the 1950s and 1960s many lakes in Northern Hemisphere urban and agricultural areas experienced intense algal blooms, fish kill events and disappearance of submerged macrophytes. Scientists later proposed that phosphorus used in detergents was largely responsible for these problems, and in many cases phased out this source (Schindler, 1974; Bronmark and Hansson, 2002).

Nutrient loading to a lake originates as either a point source, for example direct sewage and septic tank input, or a diffuse non-point source such as fertilizer applications to farmland. No matter whether it is non-point source or point source, both contribute to the nutrient load that either flows out of the lake or accumulates within the bottom sediments. During seasonal anoxic periods, nutrients in the bottom sediments are released, further adding to the nutrient load to the lake. It is the non point sources of pollution to lakes that are most problematic as eliminating this source is somewhat difficult to pinpoint.

Nutrients can also seep into aquifers where they can remain for many of years before flowing into a lake. For example nutrient enriched water from various springs and ground water wells within the western Rotorua and northern Okareka catchments were age-dated, with the majority of samples having mean residence times of over 60 years (Institute of Geological and Nuclear Sciences Ltd, 2004). Similarly in the Lake Taupo catchment the average age of groundwater is between 20-75 years in different stream inflows (Hadfied et al, 2001). Extensive pastoral land developments within the Lake Taupo catchment have contributed significant loads of nitrogen to groundwater within the past 35-40 years causing most of the recent water quality degradation in the lake (Vant & Smith, 2002). Therefore water quality deterioration is often the result of landuse activities within the lake catchment that have occurred many years prior. The Rotorua lakes have been undergoing water quality degradation for the past 30-40 years due to increased nutrient loads of nitrogen and phosphorus from pastoral farming, community sewage inputs and internal loading from the bottom sediments of lakes (Parliamentary Commissioner for the Environment, 2006). This has resulted in eutrophication and increased occurrence of nuisance cyanobacterial blooms (Vincent, 1984; Wilding, 2000; Dell, 2004; Scholes, 2004).

Managing lakes that have become degraded through human interference is not easy. Methods used to influence catchment nutrient loads are generally long-term, often expensive and require and involve catchment land use modification and regulation (Hamilton, 2005), that ultimately impact on local communities.

Measures used to reduce nutrient inputs focus predominantly on diversion of nutrient-enriched inflows (Moss, 1998), natural wetlands (Robertson et al 2000), and sewage reticulation, while in-lake measures include application of water column anti-flocculants (MacIntosh, 2007), lake bottom sediment capping to prevent internal nutrient release events (Özkundakci, 2006) and biomanipulation measures such as fish removal (Gulati & van Donk, 2002) with bivalve filter feeders having potential also (Ogilvie & Mitchell, 1995; James et al 1998; Soto & Mena, 1999; Phillips, 2006).

Therefore as the human population grows and with intensification of landuse, freshwaters are becoming increasingly degraded and impacted upon, threatening environmental, historical, recreational and economic values of freshwater bodies. This is particularly true for Lake Taupo and Rotorua Lakes areas in the North Island of New Zealand where lakes provide important recreational, cultural and economic assets but are becoming increasingly eutrophied.

1.2 The role of filter feeding shellfish in regulating water quality

Freshwater bivalve species feed by filtering the water column of suspended algae and detritus. In dense numbers they have been known to effectively reduce phytoplankton densities, leading to improvements in water clarity and have therefore been recognized as a potential biomanipulation tool for lake restoration (Phillips, 2006; Ogilvie and Mitchell, 1995; White, 2000; MacIsaac *et al.*, 1992). In a study by Soto and Mena (1999) on a bio-control for salmon farming eutrophication in Lake Llanquihue (Chile), the freshwater mussel *Diplodon chilensis* (Hyriidae) reduced chlorophyll *a* concentrations by two orders of magnitude (from 300 to 3 µg 1^{-1}). Similar results were found by Hwang *et al.* (2004) who recorded high consumption rates of phytoplankton by *Corbicula leana*, affecting planktonic and benthic food web structure and function due to preferential feeding on small seston and through nutrient recycling. Hwang *et al.* (2004) recommend that control of mussel biomass might therefore be an effective tool for management of water quality in shallow eutrophic lakes and reservoirs in Korea.

The zebra mussel (*Dreissena polymorpha*) is common in European lakes and can reach very high densities. It can have a positive effect on water clarity although it is a highly invasive encrusting species, causing many problems man-made structures and lake ecosystems (Griffiths *et al*., 1991; Reeders *et al.,* 1993; Madenjian, 1995). In Lake Erie, Canada, densities up to 700,000 m^2 have been recorded with an associated increase in water clarity attributable to the high filtering rate and their ability to remove over 90 % of organic matter suspended in

the water column (Reeders and Bij De Vaate 1990; Cooley, 1991, Dermott *et al.,* 1993; MacIsaac, 1996).

Similarly more naturally occurring filter feeders can also significantly impact on water quality by reducing phytoplankton biomass. Ogilvie and Mitchell (1995) report that the freshwater mussel (*Hyridella menziesi*) population in Lake Tuakitoto, South Otago, New Zealand (area of 118ha and mean summer depth 0.7 m) filters a volume of water equal to that of the lake once every 32 hours.

Although the above positive effects on water quality have been recorded by freshwater mussels there is also increasing evidence to suggest that they can to alter nutrient ratios favouring nuisance algae and promoting phytoplankton growth by supplying high amounts of recycled nutrients to the overlying water column. Results from the study of Asmus and Amus (1991) revealed that *Mytilus edulis* beds reduced the standing stock of phytoplankton biomass in the overlying water column. However ammonia released from the beds, particularly in summer when phytoplankton are nitrogen-limited, could induce higher phytoplankton primary production despite filtering of the water column by the mussel beds.

Nuisance freshwater algae such as cyanobacteria can form blooms and produce toxins that are harmful to humans and animals (Hamil, 2001). Surface scums can degrade the aesthetic values of lakes and also cause taste and odour problems in drinking water (Bykova *et al.*, 2006). There is increasing evidence to suggest a relationship linking the presence of freshwater mussels with increases in cyanobacterial bloom formation. One explanation for this relationship is that mussels can selectively feed on algae, favouring green algae removal and rejecting nuisance algae such as *Microcystis* sp., resulting in dominance of *Microcystis* (Heath *et al*., 1995; Vanderploeg *et al*. 2001). Similarly Bykova *et al.* (2006) show that for a lake invaded by zebra mussels, the resulting alteration of nutrient ratios promoted a re-emergence of *Microcystis* blooms.

1.3 The freshwater mussel-kakahi (*Hyridella menziesi*)

The New Zealand freshwater mussel or kakahi (*Hyridella menziesi*) is part of the Unionacea: Hyriidae family of freshwater mussels commonly found throughout Australasia (McMicheal, 1957, Walker, 1981). *Hyridella menziesi* is widely distributed throughout New Zealand in a variety of habitats ranging from small flowing streams to large rivers and lakes (Phillps, 2006). There are three other species found also in New Zealand; *Hyridella aucklandica, Cucumrunio websteri and Cucumerunio websteri* delli, however, these species not as widespread as *H. menziesi*.

Kakahi are commonly found within the littoral zone of lakes but can live down to depths of 30 m in oligotrophic conditions and are known to dominant macroinvertebrate community assemblages, forming dense mussel beds in excess of 600 individuals per m^{-2} (James, 1985, 1987, Weatherhead & James, 2001). Adults generally range in length from 25 to > 100 mm and are long-lived species, sometimes reaching ages >50 years (Grimmond, 1968).

Early life history of unionid mussels (freshwater mussels) includes a parasitic life stage where juveniles attach to a fish host for dispersal (Atkins, 1979; Phillips, 2006). Spawning occurs in summer with the male releasing sperm into the water column. Eggs kept within the female become fertilised as water is taken in by the female. Larvae are subsequently brooded in the mantle cavity of the female where they develop into glochidial larvae approximately 3 mm long and are released in spring (Phillips, 2006). The glochidia then attach to the fish host usually on the pectoral fins, head and gills, using a larval tooth (Atkins, 1979). Kakahi are known to commonly attach to native fish such as eels (*Anguila* sp.), koaro (*Galaxias brevipennis)* and giant and common bullies (*Gobiomorphis gobioides* and *G. cotidianus*). This parasitic life stage along with general life cycle of the kakahi are relatively unstudied in New Zealand but could be underpinning observations of declines in kakahi populations. Due to reductions in native fish species in New Zeland, Mc Dowall (2002) states that declines in kakahi could be attributable to losses in native fish that host glochidial larvae.

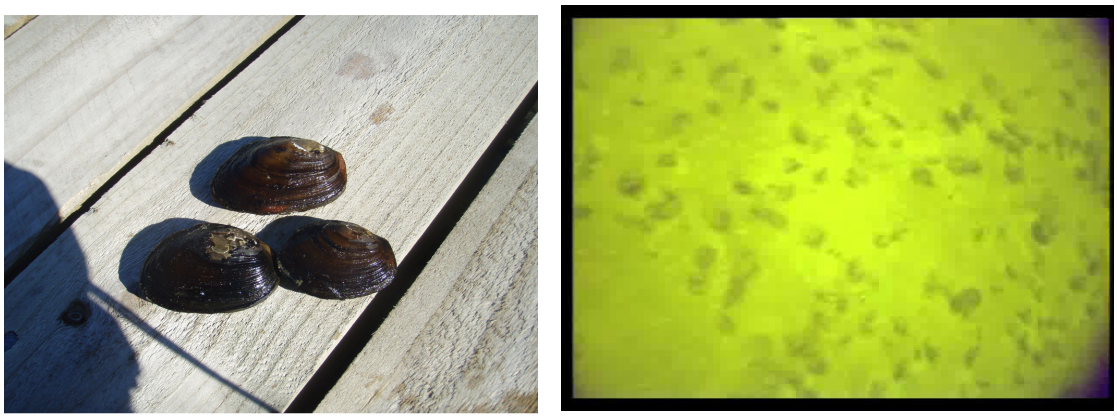

Figure 1.1: (a) Kakahi (Hyridella menziesi) (b) Snapshot of kakahi distribution in Lake Rotokakahi, using an underwater viewing lens (splashcam)

1.4 Environmental factors influencing kakahi growth and distribution

There is very little scientific information published on *H. menziesi* within lakes, and certain knowledge gaps remain in aspects of its life cycle, physiology and factors controlling its abundance and distribution. The use of freshwater mussels as a bioindicator species in environmental studies has been investigated in previous studies (Sanki et al., 2003 Roper & Hickey, 1995; Hickey et al., 1995). Similarly microcystin accumulation in kakahi tissue arising from high levels of cyanobacteria has been quantified in some of the Rotorua lakes (Wood et al., 2006). Kakahi shell abnormalities have been linked to the presence of a chironimid parasite (Forsyth, 1979; Forsyth & McCallum, 1978). James (1985) has studied the environmental factors that regulate distribution, biomass and production of kakahi in Lake Taupo New Zealand. He found through a series of multiple regression analyses that percentage of clean sand and lake bed slope were key environmental variables positively associated with kakahi density. Also, kakahi were found to be most prominent at depths below the macrophyte zone and where sediments are fine and detritus-rich (Weatherhead & James, 2001). However no data exists for potential impacts on kakahi due to a transition to decreased water quality, specifically the sort of declines that have occurred in the Rotorua lakes. It is known that the kakahi can dominate macroinvertebrate biomass in New Zealand lakes, yet there has been only one quantitative study made on the distribution and abundance of kakahi in Lake Taupo (James, 1985).

1.5 Lake Rotokakahi

1.5.1 Historical background

The Iwi of Te Arawa has resided in the Rotorua area for more than 600 years and the lakes of the region were taonga (treasures) for Te Arawa and the foundation of their identity, cultural integrity, wairua, tikanga and kawa. With increasing European settlement around the lakes in the late 1800s, forests were milled, agriculture was established and urban settlements were developed. Numerous conflicts developed between the new settlers and Te Arawa, whose capacity to provide kairoto (food from lakes) from the lakes and koha (gifts) for hospitality was diminished with deteriorating water quality and introductions of exotic species. This culminated in 1922 with the Crown taking ownership of the lakes from Te Arawa in return for a fixed annuity of £6000 paid from the Crown. The only major lake exempt from this arrangement was Rotokakahi (Green Lake). Over the tenure of Crown ownership the Rotorua lakes have become progressively degraded. As a consequence the Te Arawa people have progressively become alienated from the lakes, their taonga (tresures) have been eroded and their role as kaitiaki (gaudians) has been negated. Only in 2004 did the Crown finally agree to offer Te Arawa Maori Trust Board a settlement package for grievances in relation to the lakes.

Lake Rotokakahi is a lake of particular importance to the Te Arawa people and is steeped in historical significance, as well as having major cultural and recreational values. It is currently closed to the public and under legal guardianship of Ngati Tumatawera and Ngati Wahiao/Tuhourangi. A particular concern to the kaitiaki of this lake is its present health, which may be declining, though it has not been studied intensively. Lake Rotokakahi was last monitored regularly in 1996 under the Environment Bay of Plenty (EBOP) water quality monitoring programme, with only the lake outlet (Te Wairoa Stream) being monitored since that time. Baseline monitoring of Lake Rotokakahi was conducted by Jolly (1967) followed by Mc Coll (1972) who determined bottom waters as anoxic for at least 3 months of the year subsequently classing Lake Rotokakahi as mestrophic based on hypolimnion deoxygenation. Data collected by EBOP up to 1996 suggests that there may be degradation of water quality in the lake, as indicated by declining

levels of dissolved oxygen in the bottom waters (EBOP 2002). Lake water quality indicated through the trophic level index (TLI) also suggests water quality has degraded from 1990. The Proposed Regional Water and Land Plan (PRWLP) TLI target objective has been exceeded by 0.2 TLI units for two years and as a result a risk assessment has been completed and the lake is ranked fifth out of the seven lakes to undergo prioritization of land management options (EBOP, 2007).

Lake Rotokakahi was well known for its abundant resources and as the name suggests, particularly for its massive supply of the freshwater mussel or kakahi (*Hyridella menziesi*). The kakahi formed a major component of the Maori people's diet and was especially important to babies who could suck the kakahi from its shell after it was cooked. It was once said "ko te kakahi te whaea o nga tamariki", which translates to "the kakahi is the mother of the children", representing its importance as an appropriate food source for babies and young infants who could easily suckle the flesh of the kakahi similar to taking milk from its mother (Hiroa 1921). Kakahi shells were also used as scraping and cutting tool and for ornamental purposes (Hiroa, 1921).

Environmental factors regulating kakahi populations in lakes may ultimately be very important for water quality and this understanding could be transferred into the Maori community so that they can be pro-active in managing kakahi and actively participating in determining the future of the Rotorua lakes. This thesis presents a preliminary attempt to link environmental factors to kakahi abundance in Lake Rotokakahi and to provide Iwi with the background information that might be useful in helping them to manage, and enhance kaitiakitanga (Gaurdianship) over both the kakahi and the Lake Rotokakahi environment.

1.6 Aims and objectives

The overarching objective of this thesis is update water quality last monitored in Lake Rotokakahi in 1996 while also identifying key environmental variables influencing kakahi populations. These objectives are underpinned by a number of aims that include:

- Establishment of a one-year water quality programme within Lake Rotokakahi so that data is obtained for comparisons with previous water quality data (pre-1997) to allow assessment of whether there have been water quality changes in the lake.
- Conducting a population survey of the resident kakahi population examining possible environmental factors influencing their populations.
- Presenting an oral history of Lake Rotokakahi focusing on the its historical significance, past water quality conditions and collection of kairoto (food collected from the lake).

Chapter 2

Water Quality Methods

2.1 Study site

Lake Rotokakahi is located within the Rotorua lakes District located within the Bay of Plenty, North Island New Zealand. (Fig. 2.1) Fourteen lakes comprise the Rotorua lakes, all of volcanic origin due to their position within a highly active geothermal zone - the Taupo Volcanic Zone. Most lakes have formed through subsidence, infilling of calderas and damning of river valleys. In geological time Lake Rotokakahi is a relatively young lake, forming 13,300 years ago as a result of a damned river valley following various volcanic eruptions (Healy, 1963). The lake is considered mesotrophic (McColl 1972; Environment Bay of Plenty, 2006) has an area of 4.4 km^2 and catchment area of 19.7 km^2 comprised mainly of exotic forest (57.1 %) (Environment Bay of Plenty, 2006) although current logging operations within the lake catchment would have reduced total exotic forest land cover. It is fed by a small stream and groundwater, with the lake outflow (Te

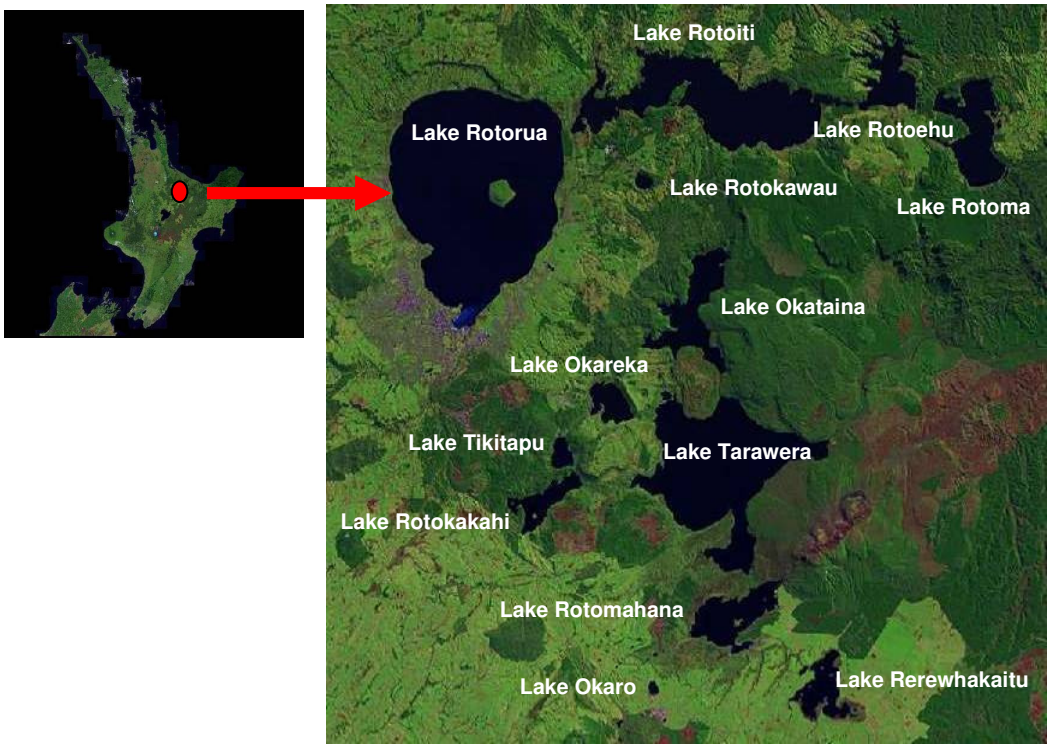

Figure 2.1. The location of Lake Rotokakahi amongst the Rotorua Lakes

Wairoa Stream) draining the lake at the north eastern end which eventually flows into Lake Tarawera (Environment Bay of Planty, 2006). Table 2.1 and 2.2 summarise Lake Rotokakahi morphology and catchment land cover respectively

Table 2.1. Lake Rotokakahi morphology (Environment Bay of Plenty, 2007)

Lake area (km2)	Catchment area (km2)	Max depth (m)	Mean depth (m)	Long axis (km)	Altitude (m above msl)
4.6	19.7	32	17.5	4.3	394

Table 2.2 Lake Rotokakahi Catchment land cover (Environment Bay of Plenty, 2007)

Catchment land cover (%)			
Pasture	Indigenous forest/scrub	Urban	Exotic forest
26.3	16.6	0	57.1

2.2 Water quality data collection

2.2.1 Field water sampling

Water quality monitoring was conducted monthly from 18 September 2006 to 14 September 2007. Water samples were collected from a mid lake sampling station and from the lake inflow and outflow (Te Wairoa Stream) (Fig. 2.2). At the mid lake sampling station samples were taken at four selected depths through the water column including the surface (0 m), surface-integrated (I.S.; 0-10 m), 15 m and 28 m depths for subsequent analysis of dissolved and total nutrients and chlorophyll *a*. The latter two water samples were an attempt to obtain thermocline and near-bottom-water samples. Surface integrated samples, were collected using a 10 m tube sampler, while surface samples were collected by immersing a sample bottle just below the surface. A Schindler-Patellas trap was used to collect samples at 15 and 27 m. From each of these depths unfiltered water samples were taken for total nitrogen and total phosphorus analysis while water samples taken for dissolved nutrients and chlorophyll *a* were filtered in the field through a 60 mL plastic syringe and Swinnex glass fibre filter. All samples were kept on ice while in the field and then taken back to the laboratory and stored frozen at -20°C

until analysis. Water samples were analysed for total nutrients (total phosphorus TP and total nitrogen TN), dissolved nutrients (soluble reactive phosphorus (PO_4-P), ammonium (NH_4-N), nitrate + nitrite (NO_3–N + NO_2-N), chlorophyll *a*, phytoplankton, and zooplankton.

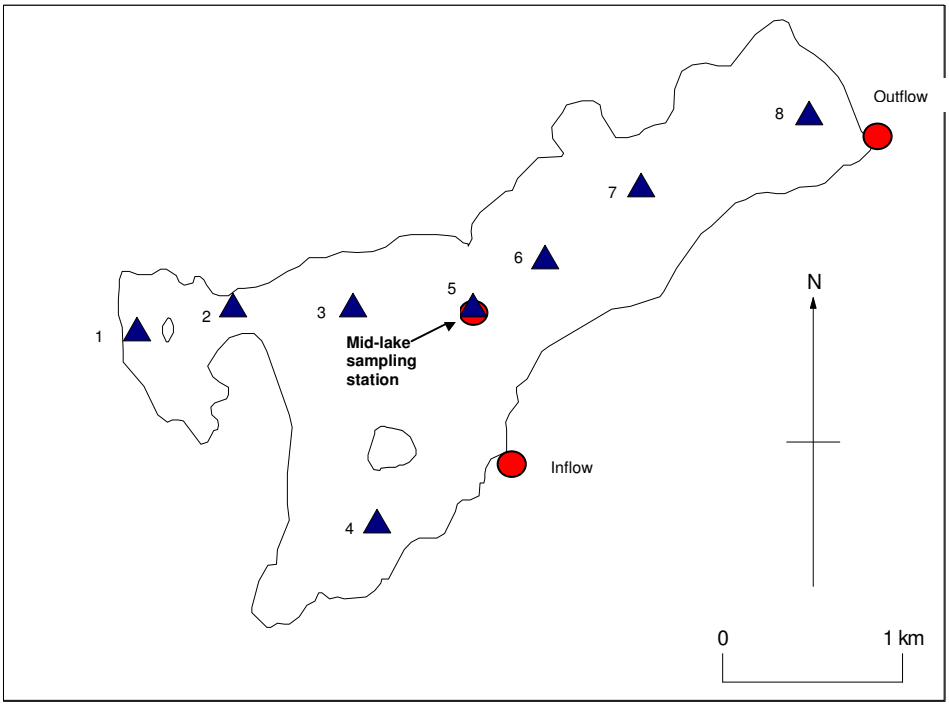

Figure.2.2 Water sampling and CTD cast sites. ● = water sampling location, ▲= CTD cast site.

2.2.2 Conductivity-Temperature-Depth (CTD) Profiling

A Conductivity-Temperature-Depth Probe (CTD) (SBE 19 and SEACAT Profiler, Washington) was used to measure conductivity and temperature variations with depth. Additional sensors for dissolved oxygen (Sea-Bird Electronics Inc.) and chlorophyll *a* fluorescence (Chelsea Instruments Ltd) allowed for profiles to be taken for these variables. CTD profiles were taken at monthly intervals at eight stations on the same day as water sampling. The location of each station was chosen to form a transect along the longest lake axis (Fig 2.2, Table 2.3) in order to construct profiles of the various measurements along the entire lake length using the graphics programme Ocean Data Viewer. The measurements included temperature, dissolved oxygen and chlorophyll *a* fluorescence with depth.

Table 2.3. GPS coordinates of CTD cast locations 1-8

CTD Site	Depth (m)	GPS co-ordinates
1.	10	38° 13' 01. 198 S 176° 17' 56. 457 E
2.	24	38° 13' 03. 731 S 176° 18' 17. 587 E
3.	30	38° 13' 02. 330 S 176° 18'42. 187 E
4.	23	38° 13' 39. 819 S 176° 18' 48. 065 E
5.	29	38° 13' 01.198 S 176° 19' 05.714 E
6.	22	38° 12' 50. 177 S 176° 19' 22. 947 E
7.	22.5	38° 12' 39. 325 S 176 18 51. 663 E
8.	15.7	38 12 26. 433 E 176 20 29. 105 E

2.3 Water Quality Data processing

2.3.1 Light attenuation coefficient (K_d) and euphotic depth (z_{eu})

The light attenuation coefficient was determined using photosynthetically active radiation (PAR) data collected by the CTD. PAR data was subsequently $\log_{10}$ transformed and entered into the following equation:

$$K_d = \frac{\ln(I(0)) - \ln(I(z))}{z}$$

Where:

I(0) = PAR at the surface and the depth where PAR = 0

I(z) = PAR at depth where $z = 0$

z = depth where *I(z)* was taken

The euphotic zone depth was estimated as 2.5 x the monthly secchi disk depth reading.

2.3.2 Chlorophyll a analysis

Samples for chlorophyll *a* analysis were collected in the field by flushing 100 mL of water sample from each depth (0 m, I.S, 15 m, 28 m) through a 0.45 µm glass fibre filter using a 60 mL syringe and attached filter tip. Filter papers were wrapped in aluminum foil and placed on ice until return to the laboratory where filters were deep-frozen. To extract phytoplankton pigments and measure chlorophyll *a* the method of Arar and Collins (1997) was followed. Each filter paper was thawed, ground to a slurry in a 90% acetone solution buffered with magnesium carbonate and left to steep (in darkness, at 4 ^{o}C) for a minimum of 2 hours. After the steeping period samples were centrifuged for 10 minutes at 3300 rpm, left to come to room temperature and then measured for fluorescence using a 10-AU fluorometer (Turner Designs) calibrated for chlorophyll *a* analysis. The following equations were used to determine the chlorophyll a concentration within the sample extract and whole water sample:

$$C_{E,c}=F_s\,(r/r\text{-}1)(R_b\text{-}R_a)$$

Where:

$C_{E,c}$= chlorophyll a concentration within the sample extract [µg/L]

F_s = response factor for the sensitivity setting

R = before to after acidification ratio

R_b = fluorescence of the sample before acidification

R_a= fluorescence of the sample extract after acidification

$$C_{s,c} = \frac{C_{E,c}.EV.DF}{SV}$$

Where:

$C_{s,c}$ = chlorophyll a concentration within the whole sample [µg/L]

EV = extract volume

DF = dilution factor

SV = sample volume

2.3.3 Nutrient analysis

Filterable nutrients (soluble reactive phosphorus [PO_4-P], ammonium [NH_4-N], nitrate + nitrite [NO_3–N + NO_2-N]) and total nitrogen and phosphorus were

analysed using a Lachat QuikChem Flow Injection Analyser (FIA+ 8000 Series, Zellweger Analytics, Inc). Total phosphorus, TP, and total nitrogen, TN, samples were first digested using the combined persulphate TN/TP digestion method (Ebina *et al*., 1983) before analysis on the FIA. PO_4-P was analyzed using the Lachat QuickChem Method 10-115-01-1-A (Diamond, 2000) while NO_3-N, NO_2-N and NOx (NO_3+ NO_2) analyzed using the Lachat QuickChem Method 10-107-04-1-A (Wendt, 2000). During this analysis nitrate within the water sample is reduced to nitrite after passage through a cadmium column, therefore NO_3 was determined by subtracting NO_2 from the NOx concentration. Ammonium (NH_4-N) was analyzed using the Lachat QuickChem Method 10-107-06-2-C (Prokopy, 1992).

To avoid contamination only milli Q water of >16 MΩ resistance was used in preparing reagents and stock standard solutions and was obtained fresh daily. Stock standard solutions were made from analytical reagent-grade chemicals, pre-dried at 105 ℃ for one hour and stored at 4 ℃. Calibration standards were made from diluted stock standard solution and prepared daily in order to calibrate the FIA before each run of water samples. A set of check standards including a milli Q water blank were repeated every 30 samples to check for signs of contamination.

2.3.4 Phytoplankton and zooplankton counting and identification

Phytoplankton were collected from the water samples taken at 0 and 15 m and preserved in Lugols iodine. From each sample, cell counts were made and key species identified using the expertise of some University associates and identification reference books from Otago Regional Council, (2000), Entwisle (1997) and Prescott (1978). Phytoplankton counting and identification were carried out following the University of Waikato protocol for phytoplankton analyses (Paul *et al.* 2007). Depending on the density of the phytoplankton sample, 1-10 ml of sample was subsampled into Utermöhl chambers (Utermöhl, 1958) where they were settled and viewed with an inverted microscope (CKX41, Olympus, Wellington, New Zealand). A count of 1 to 1.5 planktonic units per field of view at 100x magnification was aimed for, and if cell densities were much

higher or lower than this, the subsample was adjusted by dilution or concentration to achieve this density. Colonial and filamentous algae in dense aggregations were counted along a transect at 200x or 400x magnification to reduce the time taken per sample.

Plankton cell counts per ml were calculated as:

$$N = C f (A/ba V)$$

Where:

N = number of algal cells per ml in original water sample

C = total number of algal cells counted in all transects

A = total area of bottom of the settling chamber (mm^2)

a = total area of transect (mm^2)

b = number of transects counted

f = dilution or concentration factor

V = volume of lake water that was settled (mL).

Zooplankton counting and identification were conducted using a dissecting microscope and gridded counting tray. Zooplankton water samples were drained through a 40µm sieve and a 5ml subsample taken. In each subsample the first 300 species encountered were counted and identified. If 300 individual zooplankton were not encountered within the first subsample then more subsamples were made. Zooplankton densities per litre and relative abundances of species were calculated

2.4 DYRESM-CAEDYM model input variable files

The one dimensional water quality model DYRESM-CAEDYM combines a process based hydrodynamic model with numerical descriptions of in-lake biological and chemical processes to predict water quality trends in lakes and reservoirs (Hamilton and Schladow, 1997). Once calibrated against measured field data depth profiles of physical, chemical and biological variables such as

temperature, oxygen, bottom nutrient fluxes and phytoplankton production can be visualised over daily time intervals using DYRESM-CAEDYM. Essentially in lake temperature and oxygen levels will be compared against those collected in the field.

Pre-constructed input files required by the DYRESM-CAEDYM water quality model were prepared and formatted specific to Lake Rotokakahi for the period of this water quality monitoring. Input files were daily data values including water inflows and outflows, lake morphology, meteorological conditions and for comparisons with simulation output, measured biological and chemical conditions such as phytoplankton chlorophyll *a* and nutrient concentrations. However for this study only temperature and dissolved oxygen were simulated by the model. Input file data for comparison were sourced from field measurements collected monthly.

2.4.1 Meterological data

Base meteorological data required to calculate specific meteorological variables for the model DYRSEM-CAEDYM were collected from the Rotorua Airport situated approximately 9 km from Lake Rotokakahi. Base meteorological data included hourly measures of rainfall, wind speed wet and dry bulb air temperature daily total solar radiation and mean sea level pressure (MSLP) that were later averaged into daily values required by the model. Daily wet and dry bulb temperature and MSLP were used to calculate water vapour pressure while daily total solar radiation was used as a basis to calculate daily shortwave radiation and cloud cover.

2.4.2 Lake inflow and outflow data

The lake inflow and outflow stream volumes were gauged using a Marsh-McBridy flow meter. Mean flow rate ($m^3\ s^{-1}$) was calculated as the stream width multiplied by mean depth and mean flow ($m\ s^{-1}$). As the inflow is most likely groundwater fed temperature and dissolved oxygen data was kept at a constant value (12.50° C and 10.63 mg L^{-1} respectively) based on mean values of ground water fed inflows to Lake Rotorua. Nutrient data (total nitrogen, total phosphorus,

nitrate, ammonium and phosphate) was collected monthly. Flow volume and nutrient data were adjusted to daily values through a linear interpolation.

2.4.3 Lake morphometry

Lake morphometry including lake hypsography data was provided by Environment Bay of Plenty.

Chapter 3

Kakahi Methods

3.1 Field sampling strategy

A preliminary examination of kakahi distributions and potential site locations was undertaken on the 1st of March using an underwater viewing lens (Splashcam).

Kakahi data and environmental data were collected from five sites in Lake Rotokakahi on the 20th of April 2007 (Fig. 3.1, Table 3.1). These sites were chosen in order to represent the variation within the lake in those variables thought to affect kakahi populations, including lake bed slope, macrophyte coverage, sediment size, water depth and temperature, oxygen and fluorescence levels.

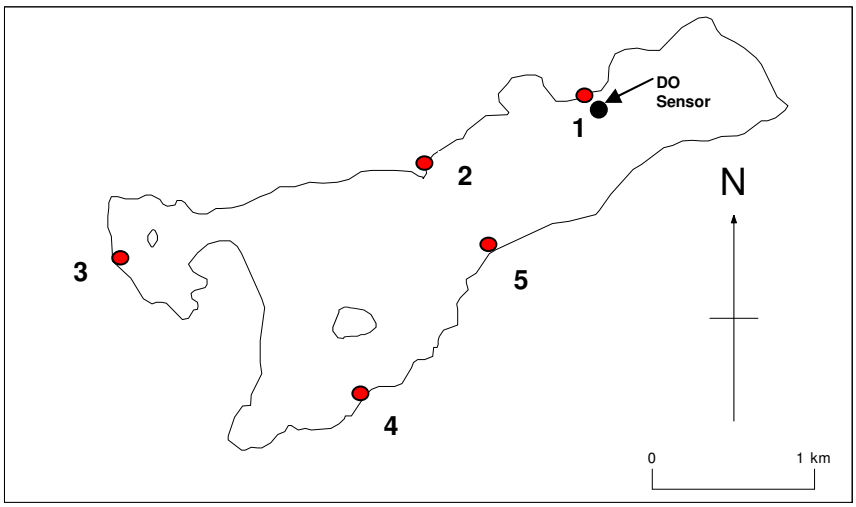

Figure 3.1 Kakahi sampling sites ● in lake Rotokakahi

Table 3.0 GPS location of kakahi sampling transects

Sampling Site	GPS co-ordinates
1	38° 12' 29. 851 S 176° 19' 51. 746 E
2	38° 12' 45. 176 S 176° 19' 14. 993 E
3	38° 13' 10. 116 S 176° 17' 54. 102 E
4	38° 13' 35. 272 S 176° 19' 11. 135 E
5	38° 12' 59.802 S 176° 19' 51.983 E

3.2 Measured variables

3.2.1 Kakahi density and size distribution

The preliminary examination of kakahi distributions showed that numbers varied greatly along the depth gradient with few kakahi deeper than 15 m. In general kakahi numbers at each site were much lower in shallow (1-2 m) and deeper regions (15-20 m) compared with intermediate depths (5-10 m). Therefore at each site, sampling of individual kakahi was made by SCUBA at 1, 5, 10 and 15 m depths along transect lines extending outwards from the lake shore.

To increase sufficient power of statistical analysis, a 0.25 m^2 quadrant was deployed at least five times to quantify kakahi numbers at 1 and 15 m depths while at 5 and 10 m depths the quadrants were deployed at least three times. Divers were dispatched at each depth to randomly place quadrants on the lake bed. All kakahi encountered within each quadrant was collected.

Kakahi density data was collected during both the preliminary study (1st March) and main sampling day (20th April 2007) when biomass and environmental data was also collected. Numbers from each 0.25 m^2 quadrant were averaged at each depth and multiplied by a factor of four to produce densities per 1 m^2.

Once on the boat, mussels were counted and individual length (long axis), width and height (i.e. thickness) of each kakahi shell was measured with callipers before specimens being retuned to the lake.

3.2.2 Kakahi biomass and condition

A sub-sample of 20 kakahi from each depth was kept and taken back to the laboratory and used to calculate biomass and condition indexes following Roper and Hickey (1993). However at 1 and 15 m it was impossible to collect 20 kakahi as numbers were very low at these depths. After each kakahi was shucked, the flesh was oven dried at 60℃ for 48 hours and shells air dried for 48 hours. The following two physical condition indices were calculated as in Roper and Hickey (1994) as a measure of physiological stress.

$$CI_{\text{flesh:shell weight}} = \frac{\text{dry flesh weight (mg)}}{\text{shell weight (g)}}$$

$$CI_{\text{flesh:shell volume}} = \frac{\text{dry flesh weight (mg)}}{\text{shell volume (ml)}}$$

3.2.3 Sediment

At each site and depth a visual assessment of sediment type and composition was recorded and classified into categories. Table 3.1 describes the basic sediment type as being either mud, sand or gravel and consistency classified as firm or soft.

Table 3.1 Visual assessment table of the sediment at each site

Sediment class	Sediment type	Consistency
1	Mud	Firm
2	Sand	Firm
3	Gravel/mud	Soft
4	Mud	Soft
5	Sand	Soft

3.2.4 Lake bottom slope

Along each sampling transect starting near-shore the lake bottom gradient or steepness was quantified from a boat using an echo sonar combined with Global Positioning System chart plotter (Garmin GPSMAP 168 Sounding) and associated software to record GPS co-ordinates and associated depth every few seconds. This data was later used within the Ocean Data View programme to plot bathymetric profiles of each transect where kakahi were sampled. Using these profiles, relative estimates of the lake bottom gradient along transects could be made by dividing the depth where kakahi were sampled by distance from shore.

3.2.5 Macrophytes

Macrophyte data was recorded when conducting initial surveys of potential transect sites. In particular percent weed cover of the lake bottom was estimated and dominant species recorded. The results of LakeSPI weed survey data was also consulted specific to Lake Rotokakahi (Clayton, 2002; Clayton, pers.com, 2007)

3.2.6 Depth

Water depth was measured initially using a depth sounder, then secondly by SCUBA depth gauges when diving at each depth.

3.2.7 Dissolved oxygen, chlorophyll a fluorescence and temperature

Dissolved oxygen chlorophyll a fluorescence and temperature measurements were obtained from the CTD profile data. These data was not collected directly from each kakahi transect site but from the CTD site nearest each transect site and averaged over the monitoring period (13 months) at 1 m 5 m, 10 m and 15 m depths.

An oxygen sensor (Greenspan Technology, Model DO300) was deployed in close proximity to site 1 (38° 12’ 30. 819S, 176° 19’ 51. 983E) (Fig. 3.1) at depths 10 and 15 m from 28 February to 19 March and from 19 March to 1 April respectively to record dissolved oxygen and temperature every 30 minutes. At

both depths the sensor was positioned 1 m off the bottom (i.e. 9 m and 14 m). This was done in order to closely observe possible fluctuations in dissolved oxygen near the sediment - water interface and to better represent dissolved oxygen levels experienced by kakahi in the immediate vicinity. It was noted that times when dissolved oxygen were recorded were not updated by the sensor and the senor stopped recording unexpectedly towards the end of the data set.

3.2.8 Data analysis

Monthly temperature, oxygen and fluorescence data was averaged annually at 1 m, 5 m, 10 m and 15m to allow comparison with kakahi data. Simple linear regression plots were initially used to observe any relationship between kakahi density and biomass and individual environmental data. If data appeared produced a non-linear pattern, a non linear regressional analysis was used. A Pearsons product momentum correlation matrix was then constructed including all variables measured in this study to examine all variable interactions.

Using the statistics programme STATISTICA (V.8) a multiple regression analysis was performed on kakahi density and biomass against all environmental variables. Kakahi condition and biomass data as dependant variables were compared against depth using a one way Analysis of Variance (ANOVA).

Chapter 4

Water Quality Results

4.1 Temporal and spatial variations in temperature, dissolved oxygen and chlorophyll fluorescence

4.1.1 Temperature

Lake Rotokakahi is monomictic, and stratified for approximately eight months annually, from October 2006 to May 2007. Water temperatures ranged from approximately 21.6 to 9.8 ℃ in the epilimnion and 9.8 to 11.0 ℃ in the bottom of the hypolimnion over the annual cycle (Fig. 4.1 a). Turnover occurred at the start of winter, in June, when the water column became isothermal (temperature 12.4 ℃). During the winter mixed period (June-September) temperatures remained between 12.4 and 9.6 ℃ (Fig. 4.1 a). The metalimnion was located between 10 and 18 m depth during the stratified period. During January and February the lake was most strongly stratified, with the metalimnion beginning to deepen in March and continuing to deepen until turnover in June (Fig. 4.2). Evidence of thermocline tilting can be seen in April-March (Fig. 4.2d) shown by an accumulation of warmer water at the north eastern end, which was exposed to prevailing winds from the west of up to 20 knots.

4.1.2 Dissolved oxygen

Dissolved oxygen concentrations changed with a pattern similar to that of seasonal temperature variations through the year (Fig 4.1 b). The hypolimnion was anoxic for four months of the year (February-May), and concentrations during this time were between 0.14 and 0.08 mg L^{-1} while during winter turnover entire lake water was well oxygenated with concentrations in the range 6.2-9.5 mg L^{-1}. The epilimnion was generally well oxygenated (> 7.1 mg L^{-1}) during stratification but with a pattern of lower values during warmer water temperatures, and with signs of reduced levels of 6.0 and 6.1 mg L^{-1} in March and June, respectively (Fig. 4.1b).

4.1.3 Chlorophyll fluorescence

Measurements of chlorophyll fluorescence are given only as relative values but numerical values approximate to concentrations of chlorophyll *a* in units of µg L^{-1}. Fluorescence measurements recorded ranged from 13 µg L^{-1} (September 2006 06 to January 2007) to concentrations up to 127 µg L^{-1} in April, May and August during winter bloom events. Solar quenching of phytoplankton was observed at in the surface waters at depths between 0 and 5 m, most notably in April to May 2006, and August through to September 2007 (Fig. 4.3). These occurrences were notable by progressively lower fluorescence towards the water surface during periods of when there was high solar radiation.

Chlorophyll fluorescence was elevated in the upper 10 m late in the period of thermal stratification, particularly from March to May and during winter mixing from July to August (Fig. 4.3). Fluorescence was generally greatest during late summer and winter and is related to chlorophyll *a* as 0.5722 • chlorophyll *a*.

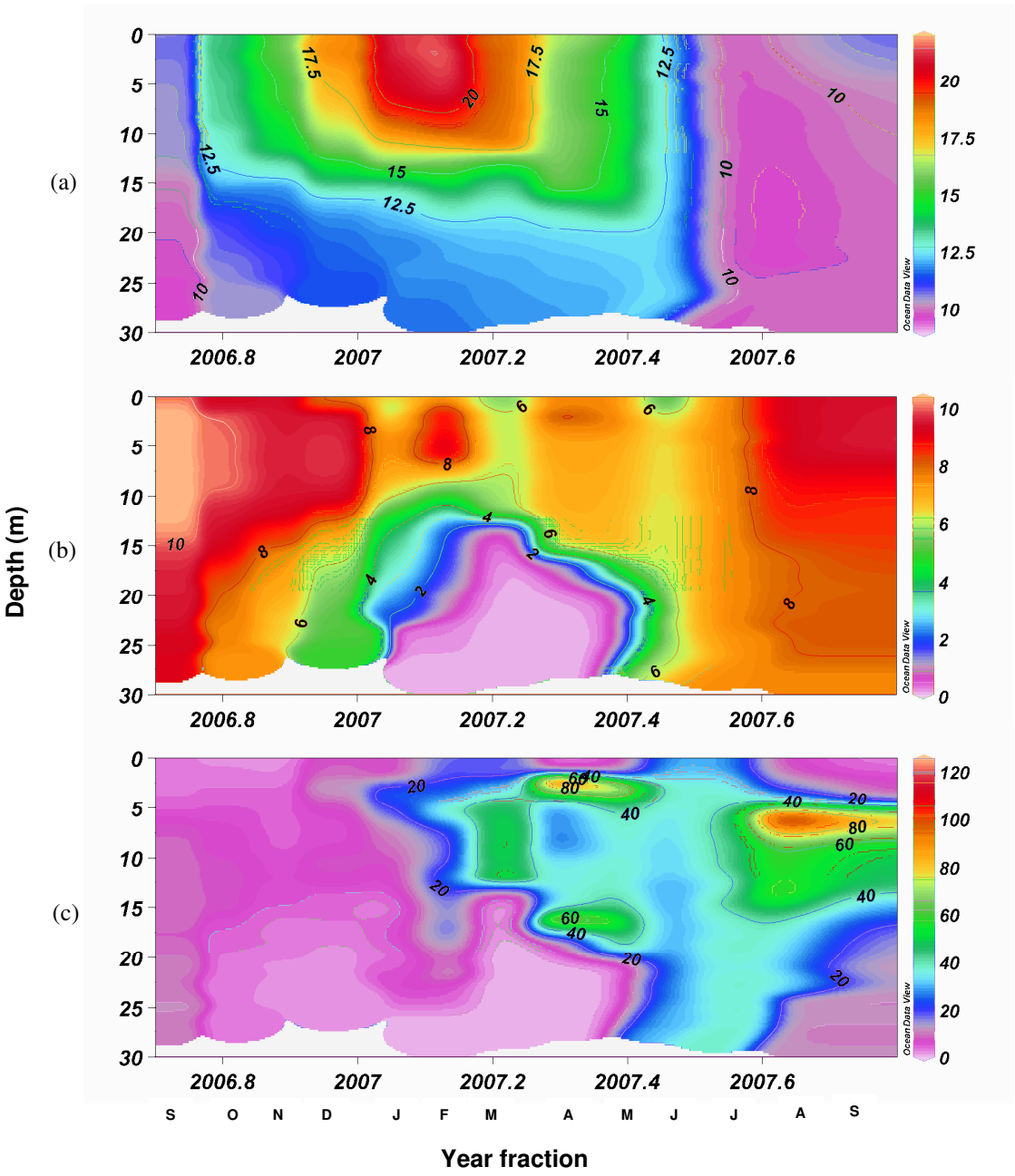

Figure. 4.1 Temperal variation in (a) temperature in ℃, (b) dissolved oxygen in mg L^{-1} and(c) chlorophyll fluorescence over the lake depth for the sampling period of 18/09/06 to 14/09/07. letters indicate month and date when CTD profiles were taken

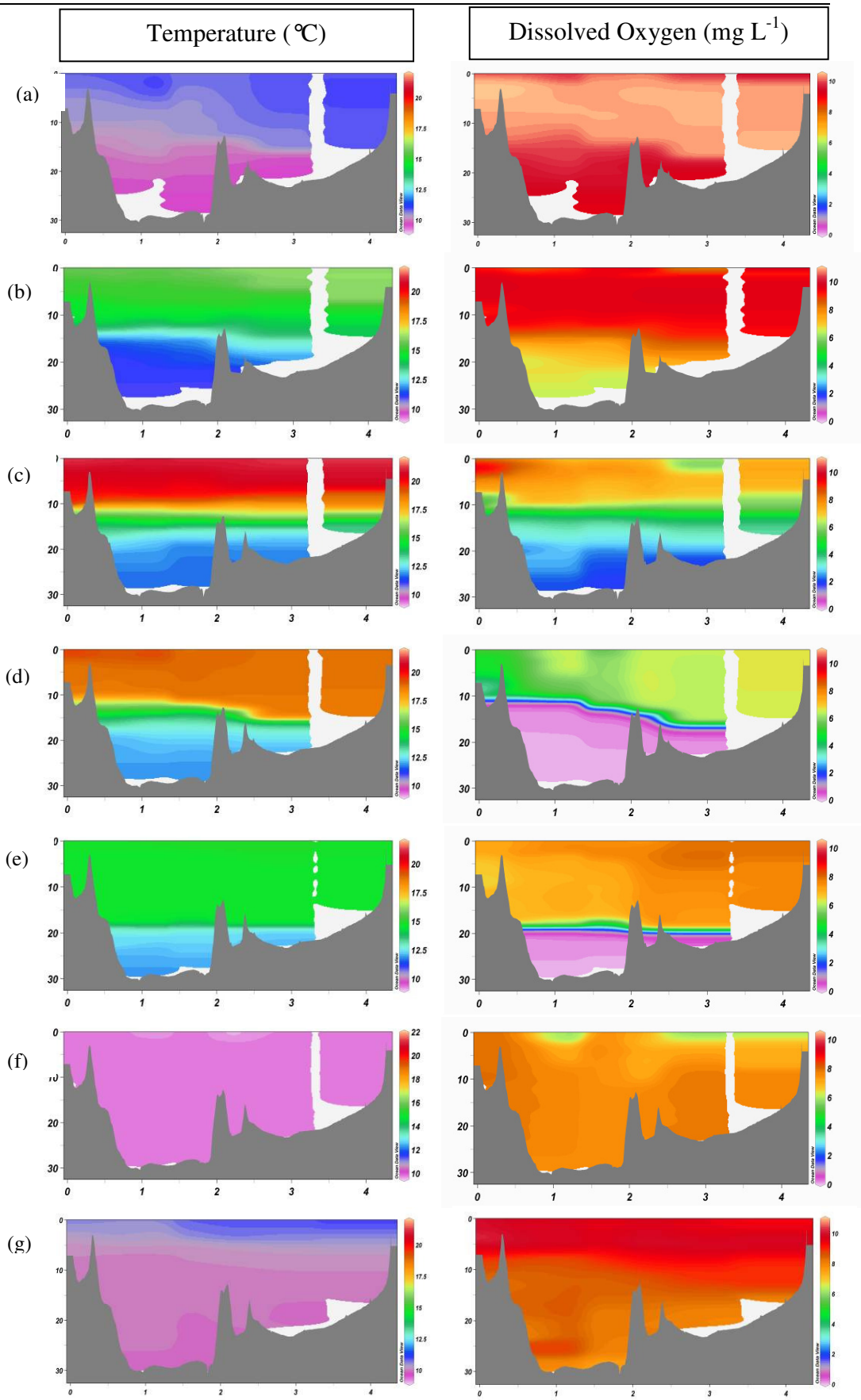

Figure 4.2. Spatial variation in temperature (ºC) and dissolved oxygen (mg L^{-1}) along the CTD transect. x axis = distance (km), y axis = depth (m). Colour bars indicate temperature and dissolved oxygen gradients. (a) = September 06, (b) = November, (c) = January, (d) = March, (e) = May, (f) = July, (g) = September 07

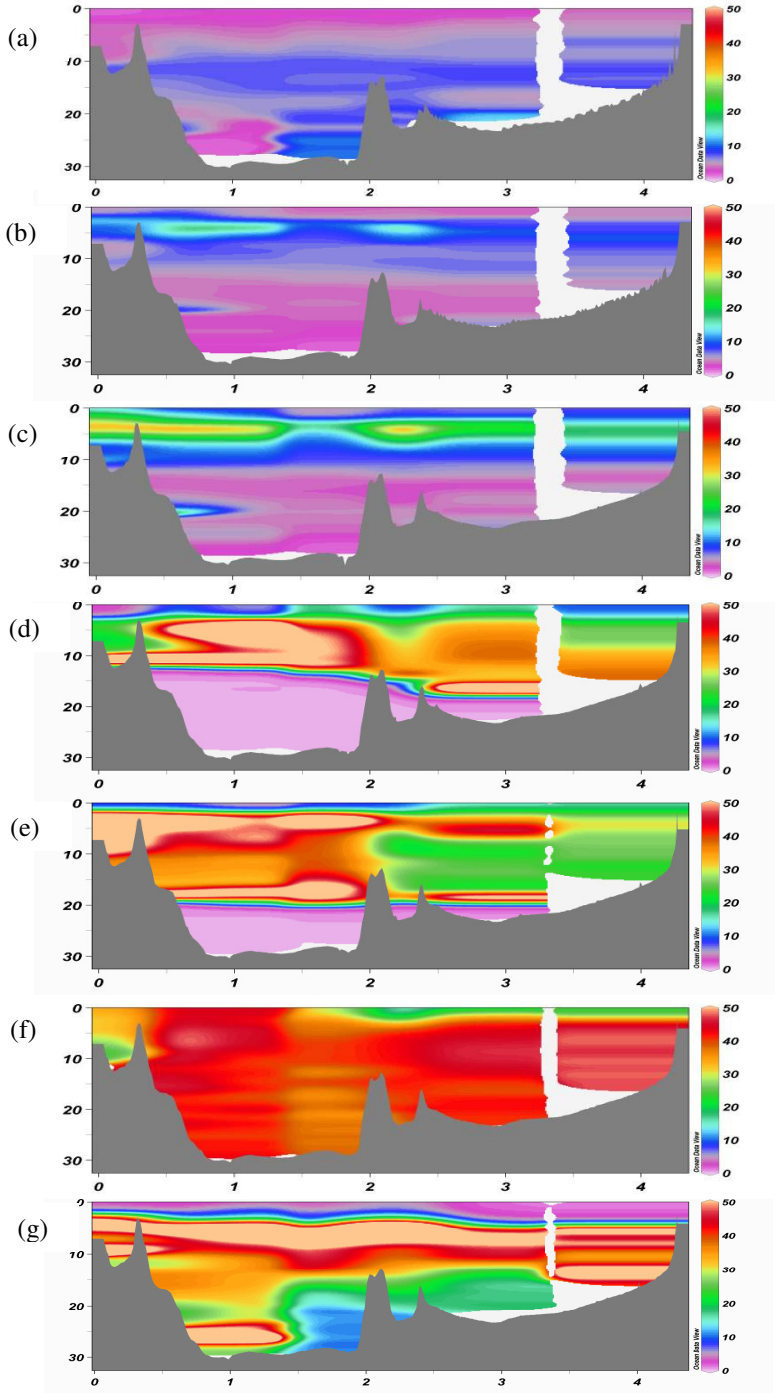

Figure 4.3. Spatial variation in fluorescence (µg L^{-1}) along the CTD transect. x axis = distance (km), y axis = depth (m). Colour bars indicate fluorescence gradients. (a) = September 06, (b) = November, (c) = January, (d) = March, (e) = May, (f) = July, (g) = September 07

4.2 Light attenuation (k_d), Secchi depth and Euphotic depth

The light attenuation coefficient was fairly consistent between stations on each sample day (Fig. 4.4). From September to November 2006 k_d was high (0.26-0.38 m^{-1}) compared with the remaining months of sampling (0.12-0.28 m^{-1}). By December 2006 k_d had decreased rapidly and reached a minimum over the year of sampling. Following December k_d gradually increased until June 2007, before decreasing for the remaining 3 months of sampling.

Secchi disk depths (Fig 4.5) were relatively high, between 4.7 and 6.5 m from September 2006 to January 2007, when chlorophyll *a* levels were mostly lower. From February to July 2007 Secchi depth then decreased to an average of 3.96 m before increasing again from July 2007 as chlorophyll *a* concentrations again decreased.

Euphotic depth (Table 4.1) was higher in spring to early summer averaging 15 m from September 2006 to December then was lower for the remaining period (average = 10.7 m) although did increase to 13.8 m in September 2007.

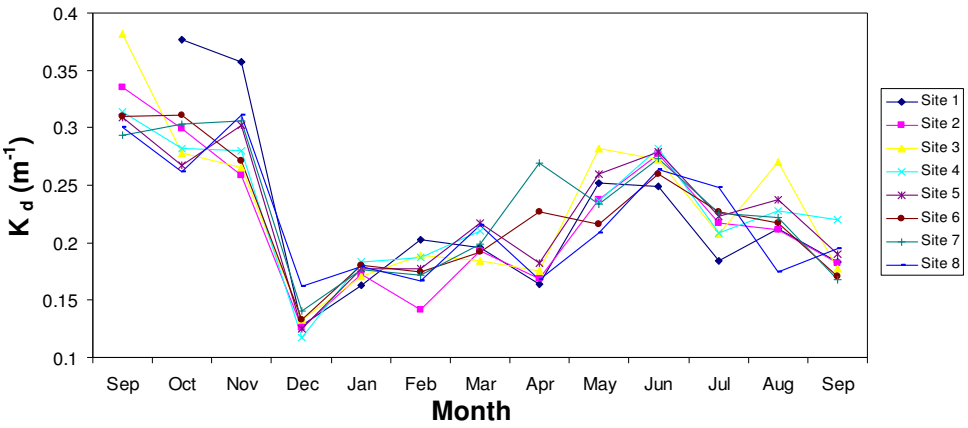

Figure 4.4 Light attenuation coefficient values (k_d) recorded at sites 1-8.

4.3 Chlorophyll *a*

Chlorophyll *a* concentrations, measured as solvent-extracted pigment, were generally fairly constant from September 2006 to January 2007, throughout the water column and in the outflow (Fig. 4.5). During the first 6 months of sampling,

chlorophyll *a* ranged from 0.62 to 1.02 µg L^{-1}. From Februrary 2007 chlorophyll *a* increased and became more variable at all locations, excluding the 27 m depth. At this depth chlorophyll *a* gradually increased from November 2006 to August 2007 and then decreased dramatically by September 2007. Two exceptional increases in chlorophyll *a* occurred within the outflow (76 µg L^{-1}) and at 15 m depth (92 µg L^{-1}) in April and May, respectively (Fig. 4.5). These high values correspond to elevated chlorophyll *a* fluorescence values observed above the thermocline in these months (Fig. 4.3 d and e).

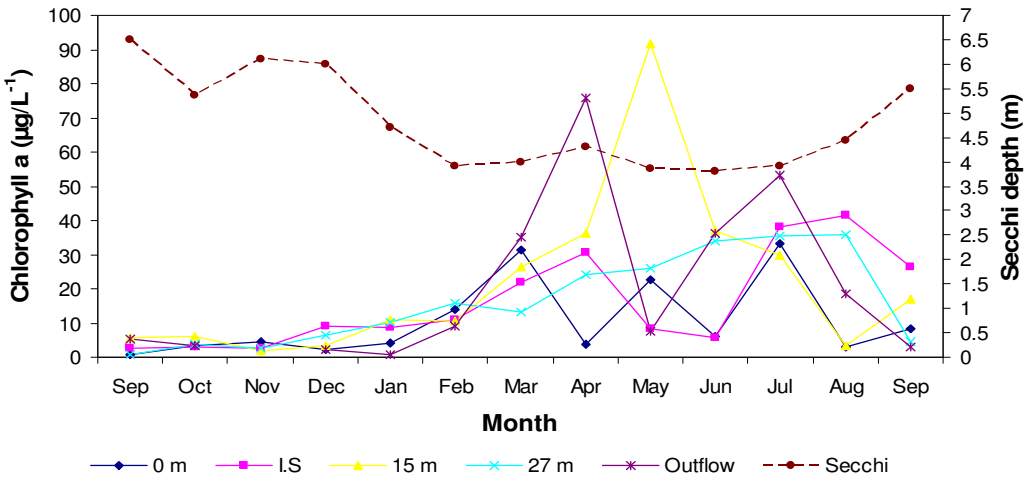

Fig. 4.5 Chlorophyll a and Secchi disk depth measurements taken at 0 m, Integrated (0 - 10 m), 15 m, 27 m and at the outflow

Table 4.1 Comparison of euphotic depth $Z_{eu,}$ light extinction coeffieicent K_d and Secchi disk depth taken monthly at the mid lake sampling station

	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep
Z_{eu} (m)	16.2	13.4	15.25	15.0	11.8	9.8	10	10.8	9.6	9.5	9.8	11.1	13.8
K_d (m^{-1})	0.3	0.3	0.3	0.1	0.2	0.2	0.2	0.2	0.3	0.3	0.2	0.2	0.2
Secchi (m)	6.5	5.4	6.1	6.0	4.7	3.9	4.0	4.3	3.9	3.8	3.9	4.5	5.5

4.4 Phytoplankton and zooplankton abundance

4.4.1 Phytoplankton

The dominant phytoplankton over the entire period of sampling were from the dinoflagellate, diatom, chlorophyte and cyanobacteria phyla. At 0 m chlorophyte species were most abundant from September 2006 to May 2007 (12-75%) followed by diatoms and dinoflagellates (Fig. 4.6 a). Cyanobacteria species were present in the first half of the year reaching 60% abundance in November then reducing to less than 10% in December onwards. From May to September both diatoms and dinoflagellates dominant with diatoms progressively increasing to 90% abundance in September. Chrysophytes and euglenophytes were present in low densities at the beginning and end of the monitoring period.

At 15 m from September 2006 to February 2007 diatoms and cyanobacteria remained dominant, with diatoms reaching 96% abundance in November 2006 (Fig. 4.6 b). From March onwards dinoflagellates were most dominant, with abundances of 46-79%. Chlorophytes were present in lower denisities, between 0.3-25 % abundance.

Dominant species included the chlorophyte *Staurastrum* spp., the diatoms *Fragilaria crotonensi*s., *Aulacoseira granulata*. and *Asterionella formosa*, the dinoflagellate *Ceratium hirundella* the cyanobacteria *Anabaena flos aquae* and *Anabaena spiroides* and the Chrysophyte *Dinobryon cylindricum* at both 0 and 15 m.

Quantitative cell counts were not determined at 0 m as samples as samples were obtained by drag net and therefore were concentrated. Generally cell counts taken at 15 m (Fig. 4.7) range from 844 to < 10 cell ml^{-1}. Diatom and dinoflagellates reached peaks of 2,821 cell ml^{-1} and 1,891 cell ml^{-1} in February and March, respectively, while cyanobacteria were highest in December peaking at 514 $cell^{-1}$.

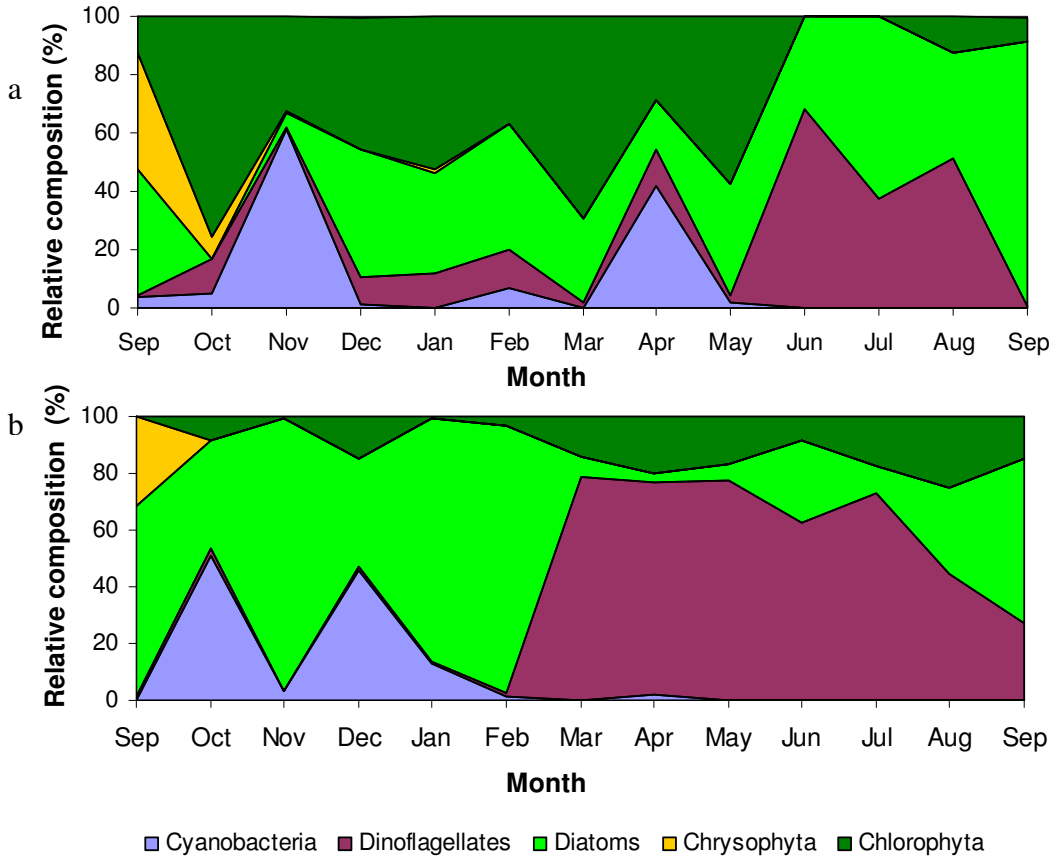

Figure. 4.6 Relative composition of phytoplankton phyla at 0 m (a) and 15 m (b)

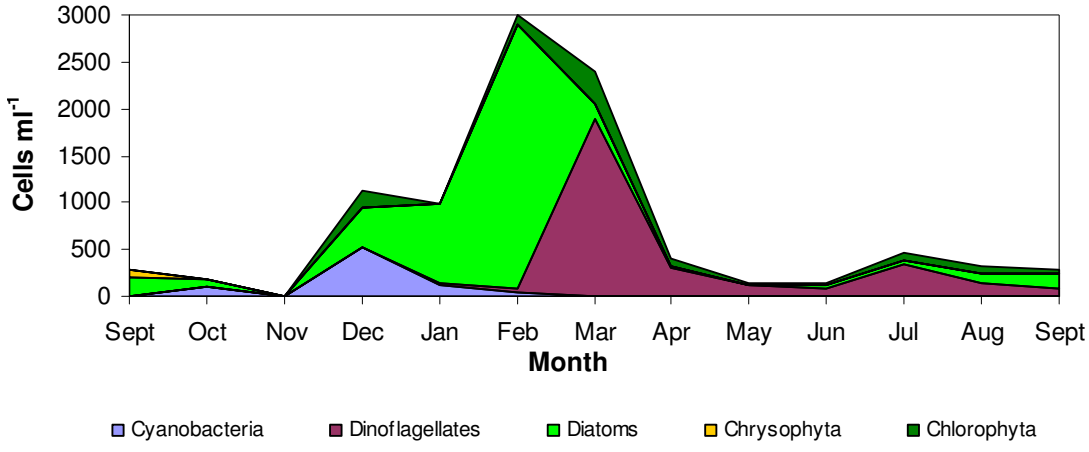

Figure. 4.7 Cell counts of phytoplankton phyla taken at 15 m depth

4.4.2 Zooplankton

Most zooplankton groups showed large monthly fluctuations in relative abundance, ranging from >90% to >5% (Fig. 4.8 a). Rotifer abundances appeared to be negatively related cladocerans and copepods abundances while nauplii remained at low abundances with little monthly variation. From March to September 2007 cladoceran, copepod and rotifer abundances gradually increased. Nauplii roughly followed copepod abundances.

Dominant zooplankton species represented in each group included *Bosmina meridionalis* (Cladocera), *Calamoecia lucasi* (Calanoid copepods), *Keratella* sp. *Filinia* sp., *Trichocerca similes* and *Asplanchna* sp. (Rotifera) and calanoid nauplii.

Densities recorded from 0 to 10 m were very high for rotifers in January and March (>300 L^{-1}), however, underwent large fluctuations between (February and April 2007) when there were very low densities (Fig. 4.8 b). Cladocerans and copepod densities both peaked in February and then had low densities thereafter. Over the entire period of sampling, average annual densiity recorded was 32 zooplankton L^{-1}. Zooplankton relative composition and cell counts were not determined in October.

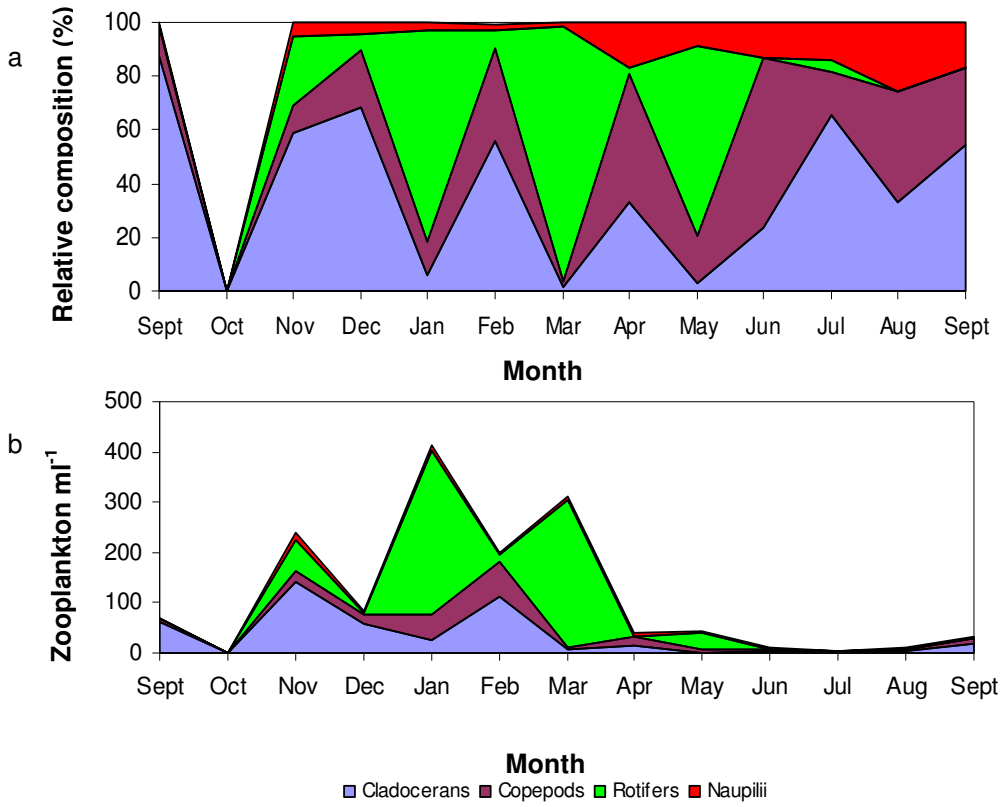

Figure. 4.8 Relative composition (%) (a) and composition (ml^{-1}) (b) of zooplankton phyla within the integrated sample (l^{-1}0 m)

4.5 Nutrients

Concentrations of phosphate were low (0.001-0.010 mg L^{-1}) at most sampling locations during the monitoring period. However samples from 27 m depth and the inflow in particular showed marked increases during February to June 2007 and from April to August 2007, respectively (Fig. 4.9).

Concentrations of total phosphorus (TP) were mostly higher in the inflow (0.05-0.092 mg L^{-1}) compared with other lake site concentrations (0.005-0.066 mg L^{-1}) including the outflow, for which concentrations were mostly lower (Fig 4.9).

Ammonium concentrations at 27 m were higher than most depths including the lake inflow and outflow. Concentrations at 27 m depth peaked in May (0.073 mg L^{-1}) before decreasing to a minimum in July and remaining low thereafter (Fig

4.9). Ammonium at all other sites showed similar trends, ranging from 0.001-0.024 mg L^{-1}. In December, concentrations of ammonium at 0 m, the integrated sample (0-10 m depth), 27 m and the outflow were greater (0.016-0.019 mg L^{-1}) than at any other time during sampling for the respective sampling sites.

From January 2007 nitrate generally decreased except for an increase in August. Similar to other nutrients, nitrate concentrations in the inflow were generally higher than all other sampling sites on the same day of sampling. At 27 m depth nitrate concentrations were greatly reduced relative to ammonium.

Concentrations of total nitrogen gradually increased from September 2006 to March 2007, then tended to plateau at all sites until completion of sampling. One notable feature was a relatively large peak in TN (0.587 mg L^{-1}) in the outflow in August.

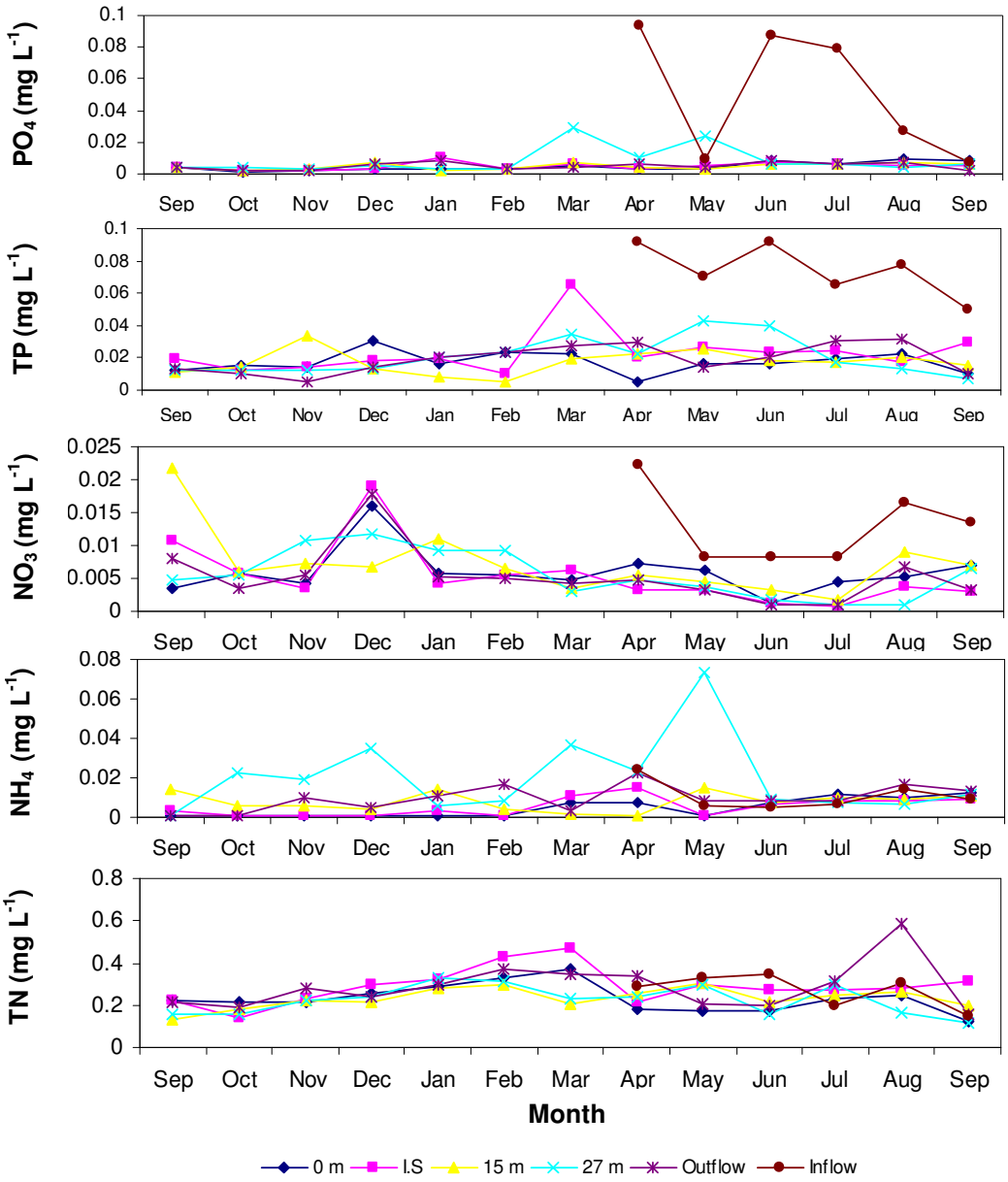

Figure. 4.9 Total and dissolved nutrients from 0 m, integrated (0 - 10 m), 15 m, 27 m, outflow and the inflow

4.6 Modelling using DYRESM-CAEDYM

4.6.1 Temperature

Fig. 4.91 presents a comparison of temperature data simulated with the model DYRESM against measured data from CTD profiles. Temperature was predicted by the model most accurately at the water surface (Table). At a depth of 14 m the temperature simulated with the model were too high during stratification (January-April). In the hypolimnion (26 m) modelled temperature data was generally slightly lower than the field data, however simulations captured the slight increase in temperature from June-July.

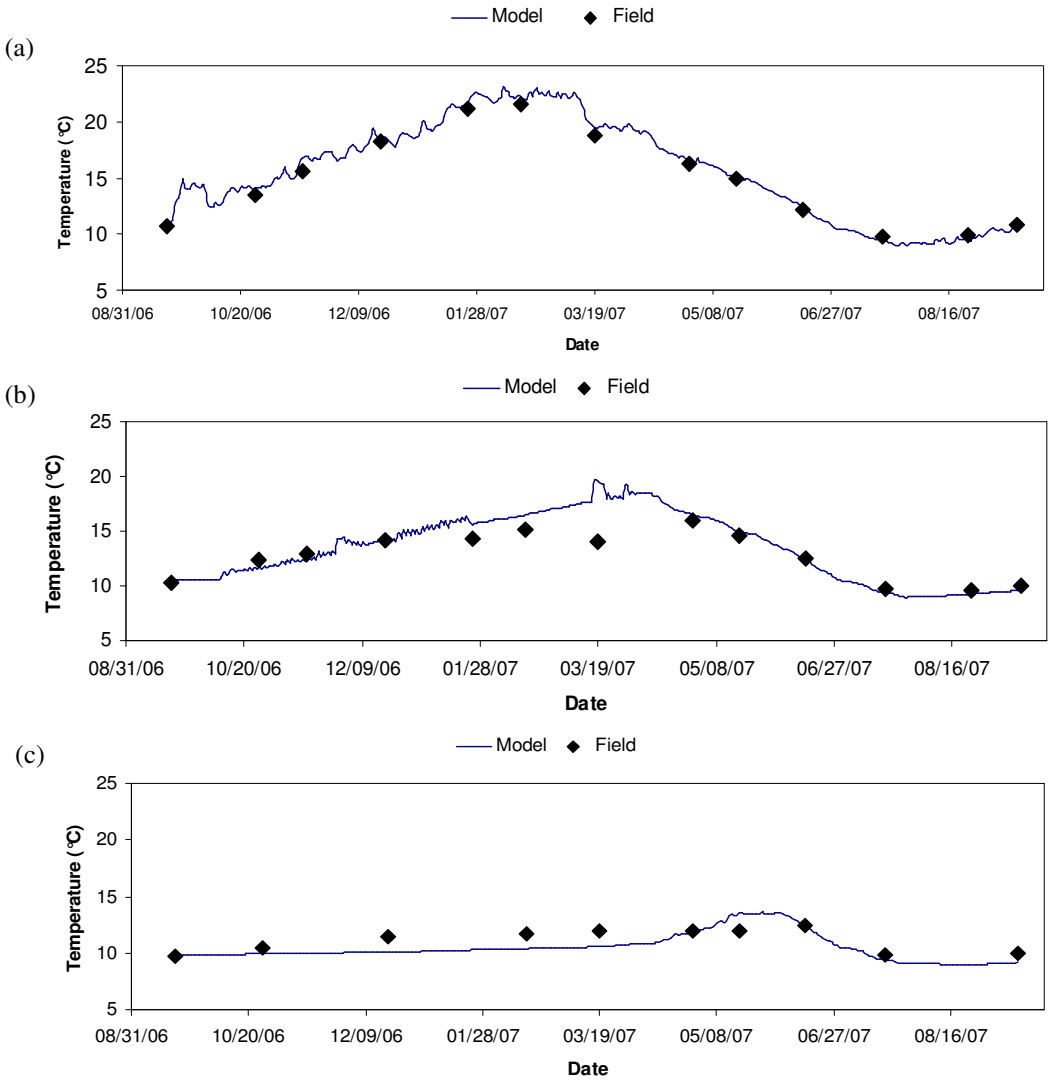

Figure. 4.91 Temperature observed and modelled data at 0 m (a), 14 m (b) and 26 m (c).

4.6.2 Dissolved Oxygen

The model simulations captured the general pattern of variations in dissolved oxygen, which were associated generally with changes in stratification (Fig. 4.92). At the surface the model did not represent well the peaks and troughs in the observed dissolved oxygen concentrations. Simulations of dissolved oxygen tended to be too high throughout the water column during the period of lake mixing. At 14 m depth the model did show the large reduction in dissolved oxygen that occurred from December 2006 to March 2007, however, during the entire monitoring period simulated concentrations were generally too high. At 26 m the low dissolved oxygen concentrations during the period of summer stratification were mostly well predicted, at least until the period of winter mixing (Fig.4 .92 c).

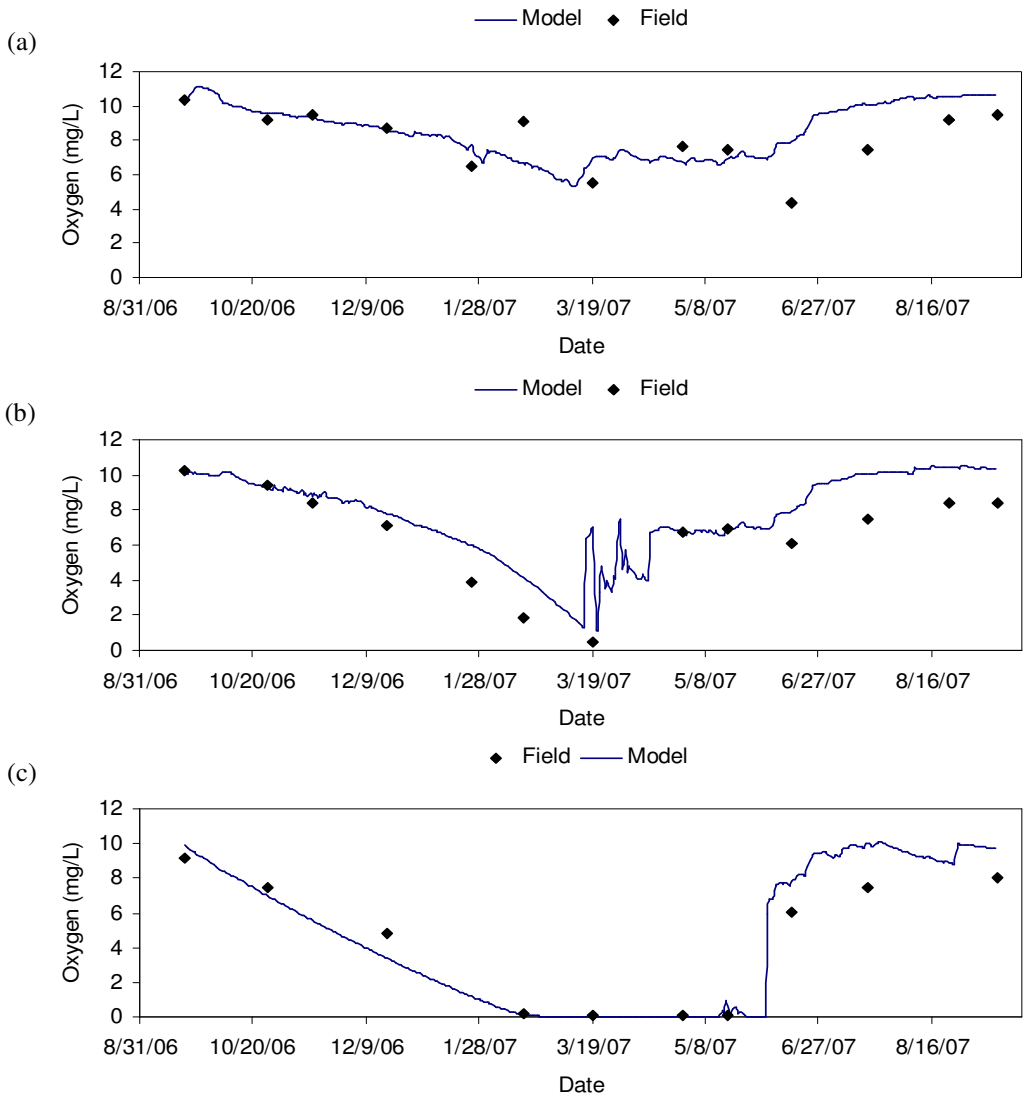

Figure. 4.92 Dissolved oxygen observed and modeled data at 0 m (a), 14 m (b), 26 m (c).

Chapter 5

Kakahi Results

5.1 Population densities

Table 5.1 Site characteristics where kakahi were sampled including mean kakahi density (m^{-2}) with range in parenthesis

Site	Depth (m)	Macrophyte species	Weed coverage (%)	Slope (depth/dist)	Sediment characteristics	Kakahi density
1	1	*Elodea*	10	0.33	Sand/soft	40.8 (28.8-52.8)
	5	*Elodea*	70	0.19	Mud/firm	204.5 (106.1-302.9)
	10	-	0	0.21	Mud/soft	213.3 (102.6-324)
	15	-	0	0.21	Gravel/firm	19.3 (0-38.7)
2	1	Raupo	90	0.50	Mud/soft	3.8 (2.7-5)
	5	*Elodea*	70	0.13	Mud/soft	163.9 (88.3-239.5)
	10	*Elodea*	60	0.16	Gravel/soft	552.8 (540.3-565.3)
	15	-	0	0.16	Mud/firm	32 (7.5-56.5)
3	1	Raupo	100	0.50	Mud/soft	79 (32-126)
	5	Elodea	80	0.16	Mud/soft	386.9 (274.7-499.2)
	10	Pondweed	45	0.19	Mud/firm	232.7 (125.3-340)
	15	-	0	0.15	Mud/firm	34.7 (0-69.3)
4	1	Turf	60	0.50	Sand/firm	16 (0-32)
	5	*Elodea*	100	0.17	Mud/soft	79.8 (70-89.6)
	10	Charophyte	60	0.22	Sand/soft	376.3 (146.7-606)
	15	-	0	0.12	Mud/firm	0.8 (0-1.6)
5	1	Turf	30	0.10	Sand/firm	72 (0-144)
	5	*Elodea*	100	0.25	Mud/Soft	172.8 (41.6-304)
	10	-	0	0.24	Mud/Firm	566 (544-588)
	15	-	0	0.21	Mud/Firm	27.2 (0-54.4)

Total kakahi numbers averaged at each depth were greatest at a depth of 10 m (avg. 388 m^{-2}) but were also quite abundant at 5 m (avg. 202 per m^2) (Table 5.1, Fig. 5.1). A large range in numbers was evident at most depths except 15 m depth (0-69 m^{-2}). At 1 m kakahi numbers were greatly reduced, and ranged from 5-144 m^{-2} (Table 5.1). Visual observations by the underwater viewing lens indicated that there were no live kakahi present at 20 m.

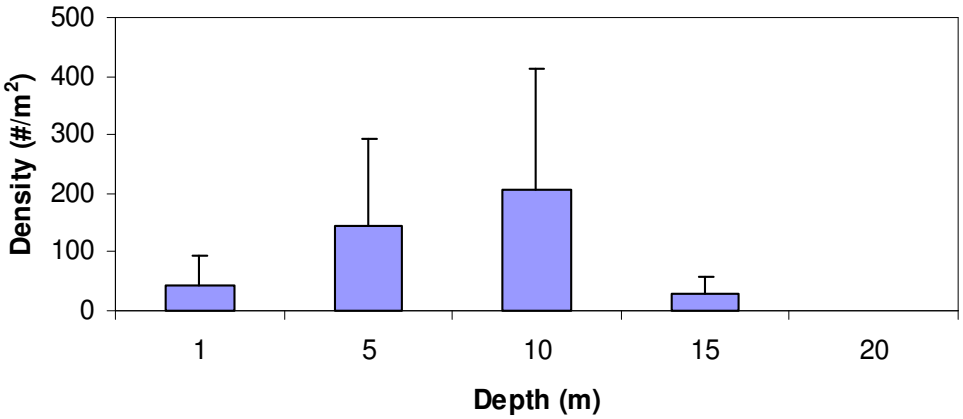

Figure. 5.1 Mean kakahi densities recorded at each depth. Error bars = 1 standard deviation. n =66 (1m), 396 (5 m), 1184 (10 m), 72 (15 m)

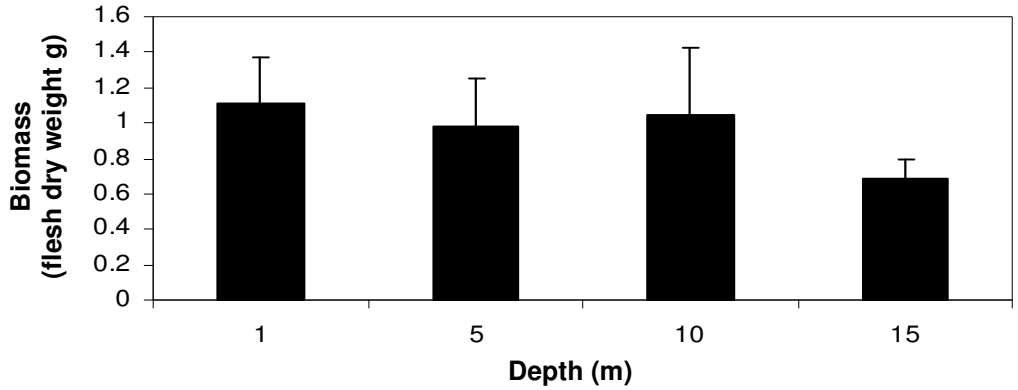

Figure.5.2 Mean biomass recorded at each depth. Error bars = 1 standard deviation. n = 59 (1 m), 97 (5 m), 99 (10 m), 16 (15 m)

5.2 Biomass

The relationship between biomass and depth was better described through a polynomial regression ($R^2 = 0.43$). Average biomass of individual kakahi at 15 m was much lower than at all of the shallower depths, and was 30 % less than that at 10 m depth (Fig. 5.2). However it must be noted that *n* was only equal to 16 at 15 m and standard deviation at 1 -10 m was higher than at other depths.

5.3 Condition

Mussel condition indices give an indication of potential physiological stress on the mussel (Goldberg, 1980). Both mussel condition indices ($CI_{flesh:shell\ volume}$ and $CI_{flesh:shell\ weight}$) showed a slight decrease with depth, although at 10 m $CI_{flesh:shell\ volume}$ in particular did increase (Fig. 5.3). Kakahi were in poorest condition at 15m depth while the kakahi at 1 m and 10 m depths were in best condition. Both $CI_{flesh:shell\ volume}$ and $CI_{flesh:shell\ weight}$ were significantly related to depth ($p > 0.05$).

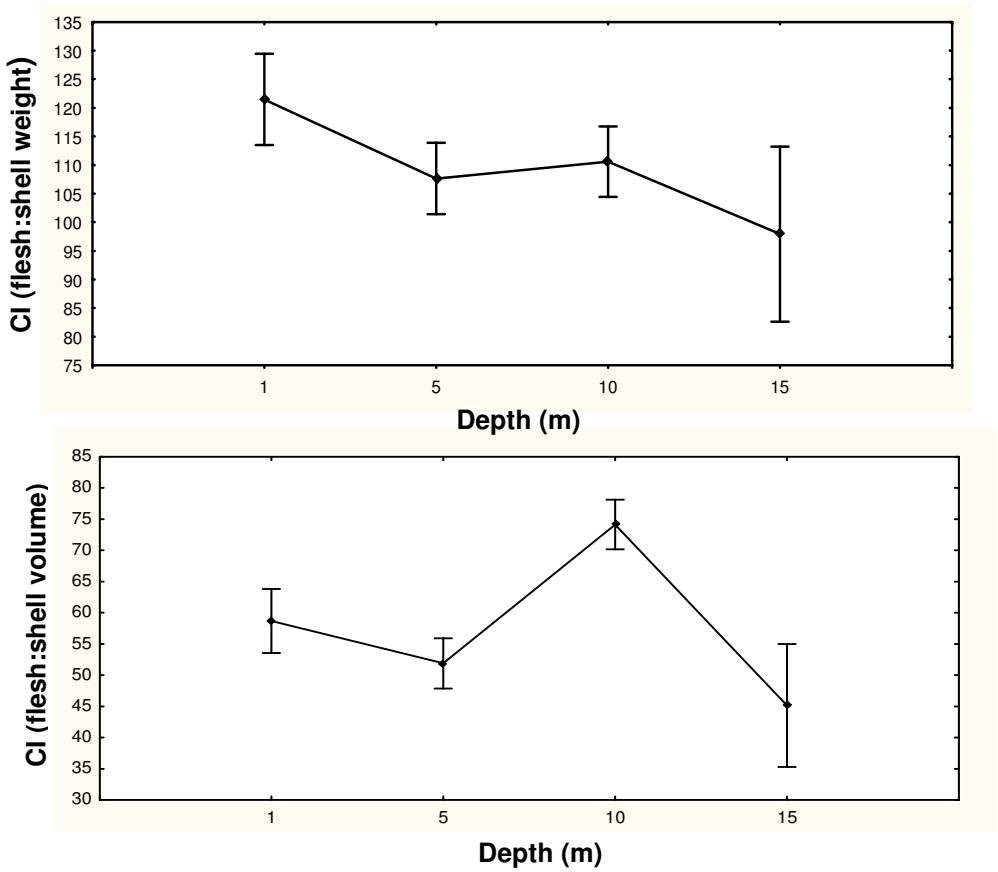

Figure. 5.3 Mean kakahi condition (CI (flesh: shell volume) and CI (flesh: shell weight) at depth. Error bars = ± 0.95% confidence intervals. n = 59 (1m), 97 (5 m), 99 (10 m) and 16 (15 m)

5.4 Size relationships or distribution with depth

Kakahi length to width relationships were highly correlated (R^2=0.74) (Fig. 5.4 a) compared to length/width, width/height relationships (Fig. 5.5). Within Fig. larger individuals seem to be located at 1 m depth ($p < 0.001$). Length/height, width/height relationships show modest correlations with each other R^2 = 0.34 and 0.38 ($p > 0.1$) and with no relationship to depth.

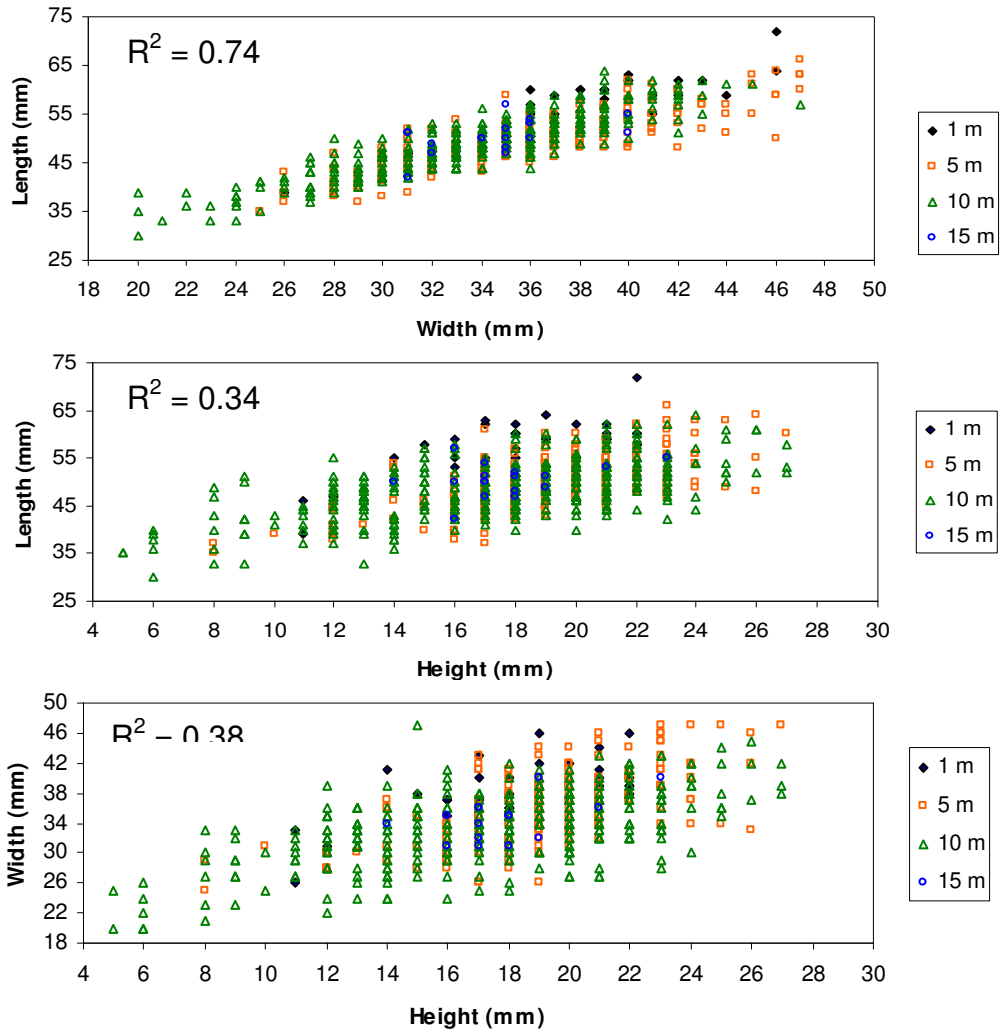

Figure. 5.4 Kakahi size relationships with depth. n = 28 (1 m), 191 (5 m), 362 (10 m), 17 (15 m)

5.5 Density and biomass relationships with measured environmental variables

Table shows a Pearson's product momentum correlation (R) matrix of all variables measured.. Kakahi density was most closely correlated with fluorescence ($R = 0.56$, $p < 0.05$) while biomass was most closely correlated with dissolved oxygen but not significantly ($R = 0.649$, $p > 0.05$), A multiple regression analysis with all variables included as independent variables, revealed that 61 and 60% of all variation in the dependant variables namely density and biomass, could be explained respectively. The variables included weed cover, slope, depth. Sediment type, temperature, chlorophyll fluorescence and dissolved oxygen. As

expected there were high correlations between environmental variables such as temperature and dissolved oxygen, and with water depth ($R > 0.91$).

A plot of kakahi density versus depth indicated a non-linear relationship that was better explained through a polynomial regression ($R^2 = 0.46$, $p < 0.05$) than a linear regression, as most kakahi were found at intermediate depths (5 and 10 m) rather than at shallower (1 m) and deeper sites (15 m).

Table 5.2. Pearson's product momentum correlation matrix of kakahi density, biomass and environmental variables.

	Depth	Density	Biomass	Slope	Macrophytes	Sediment	Temperature
Depth							
Density	0.040						
Biomass	-0.484	0.505					
Slope	-0.550	-0.251	-0.005				
Weed cover	-0.674	0.256	0.399	0.358			
Sediment	-0.509	0.046	0.442	0.299	0.506		
Temperature	-0.946	0.228	0.640	0.434	0.723	0.546	
Fluorescence	0.207	0.561*	0.128	-0.530	0.310	0.101	0.023
Oxygen	-0.911	0.239	0.649*	0.394	0.766	0.528	0.980

**** = variable relationship significant (p < 0.05)***

5.6 Dissolved oxygen and temperature near the sediment-water interface

Sharp fluctuations in dissolved oxygen were recoded near the lake bottom at 9 m depth in approximately 1-2 day cycles ranging from concentrations above 5 mg L^{-1} to those close to 0 mg L^{-1}. These fluctuations in dissolved oxygen became more reduced with time and remain close to 0 mg-1 for approximately 6 days then suddenly rises to below 10 mg L^{-1} which stays constant for 10-12 days. At 14 m concentrations remain low then increase to approximately 5 mg L^{-1} then progressively reducing after which the sensor stops logging data unexpectedly.

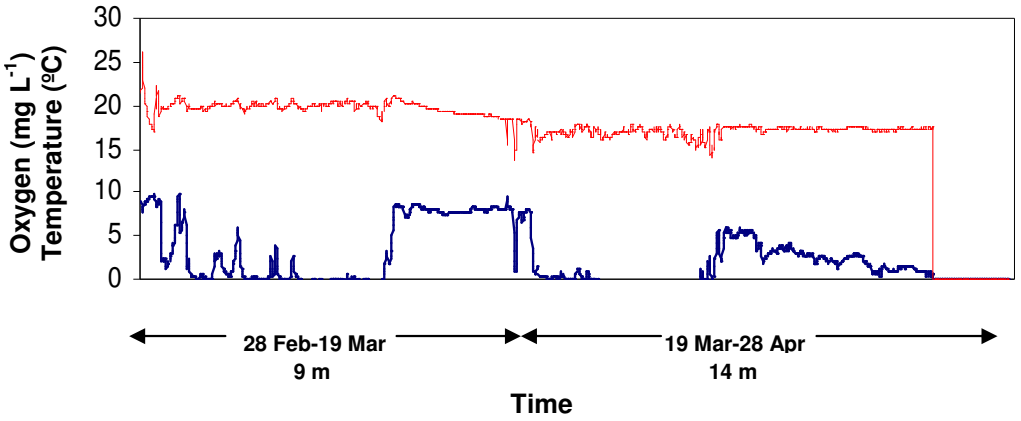

Figure 5.5. Dissolved oxygen and temperature data recorded in 30 minute intervals at 10 and 15 m depth in close proximity to site 1

Chapter 6

Discussion

6.1 Temporal and spatial variations in temperature, dissolved oxygen and fluorescence

All lakes within the Rotorua lakes region excluding Rotorua and Rotoehu are described as warm monomictic and undergo thermal stratification once a year (Hutchinson, 1957; Jolly, 1967; Irwin, 1968). Temporal variations in temperature are largely driven by ambient air temperature and wind speed, with low winds coupled with higher temperatures in summer promoting thermal stratification of deep lakes. Consequently beginning in autumn, temperatures begin to decrease in Lake Rotokakahi, and deeper convective mixing leads to depending of the thermocline, ultimately leading to overturn. As a result, during winter Lake Rotokakahi is isothermal throughout the water column. From 1990-1996 temperature measurements in Lake Rotokakahi showed a rate of increase of 0.09 ℃ yr^{-1} (EBOP, 2003). However seasonal trends in temperature and thermal structure have not changed compared with earlier data collected by Irwin (1968). Simulated data using the model DYRESM-CAEDYM agrees with field measurement and replicates the seasonality of thermal stratification and mixing.

Thermocline titling events are caused by the production of an internal seiche or standing wave along a density gradient within the water column, i.e. at the thermocline (Mortimer, 1953). Formation, structure and frequency of an internal seiche largely depend on basin shape, wind field, density (thermal) stratification and local topography (Mortimer 1993). Internal seiches travel in the direction of the wind, forcing less dense water above the thermocline to the most wind-exposed end of the lake where this water accumulates progressively, increasing the lake level and deepening the thermocline. Consequently the thermocline will be deepest at the end of the lake most exposed to the constant wind direction, and shallower at the sheltered end. Oscillations in the internal seiche may occur,

causing the thermocline to tilt back and fourth with wave cycles lasting sometimes > 48 hours (Lemmin et al, 2005). A wave period of c. 24 hours has previously been found for Lake Rotoiti (Green et al., 1968). In Lake Rotokakahi thermocline tilting is evident in March (Fig. 4.2 d) where the north-east end of the lake is most exposed to west and south-westerly prevailing winds. In Lake Geneva, Switzerland, a warm monomictic lake, internal seiching causing thermocline depth oscillations during the stratified season has been well documented (Bohle & Carbonell, 1986; Lemmin et al, 2005). Variations in dissolved oxygen recoded by the oxygen sensor near the lake bottom at the north eastern end do appear to be caused by thermocline tilting events. At 9 m fluctuations lasting 1-2 days may be a result of seiching causing the thermocline to oscillate. In effect anoxic waters will follow suit rising up and down according to the oscillations of the thermocline.

Dissolved oxygen variation both spatially and temporally are largely controlled by seasonal patterns of temperature, with concentrations varying greatly between the hypolimnion and epilimnion during thermal stratification. As the hypolimnion is largely isolated from the effects of surface aeration during summer, and due to sedimentation of abundant organic matter from the surface waters, it becomes anoxic with organic decomposition occurring in the lake bottom sediments and also the water column (Burns et al, 1996). The anoxic layer is also affected by tilting of the thermocline, becoming deeper at one end and shallower at the opposite end according to the timing of the seiche. The brief increase in dissolved oxygen observed between 0 and 10 m between February and March 2007 could be caused by higher concentrations of algae. Periods of low oxygen extending above the usual position of the thermocline are possibly caused by basin-scale seicheing allowing anoxic water rise with the thermocline and reaching depths more commonly associated with the epilimnion. Modelled data for dissolved oxygen agrees roughly with observed data with the hypolimnion becoming anoxic for approximately 4-5 months of the year. Results of extensive monitoring on Lake Rotokakahi carried during out 1971 to 1972 suggest that the deoxygenation of the bottom waters occurred for at least three months (McColl, 1972). The hypolimnetic oxygen deficit was calculated as 6.59 g m^3 for 182 days during the

same period. Burns (1996) states that the sediment oxygen demand can have a significant effect on hypolimnetic oxygen concentrations in Lake Rotokakahi. Burns et al (1994) compared the onset of oxygen depletion for 1971 and that in 1994 and concluded that at both cases anoxia commenced in March. However in the current data both modelling and field data suggests this was a month earlier in February. Historical data from 1990-1996 provided by EBOP suggests oxygen concentrations within the hypolimnion Have decreased compared with my data, a trend also observed in Lake Tikitapu, Okareka and Okataina Measuring the duration and depletion rates of oxygen in the Rotorua lakes is proven to be a very reliable indicator of trophic condition in stratified lakes accurately representing changes in water quality over the past 4 decades (McColl, 1972; Hamilton, 2006)

6.2 Chlorophyl *a*, fluorescence, algal biomass and secchi disk depth

General patterns of chlorophyll fluorescence are consistent with those for chlorophyll *a*, and algal biomass particularly above the thermocline during late summer and throughout the water column in winter. High fluorescence around to the thermocline region within the epilimnion represents a deep chlorophyll maximum (DCM) consistent with high cell counts at 15 m depth during the same period. Algal species causing this were the diatom *Fragilaria crotonensis* and possibly the dinoflagellate *Ceratuim hirundella*. Chlorophyte, diatom and dinoflagellate biomass was elevated at 15 m depth during January through to March. Increased biomass measured near the thermocline is where there is low tubulance which allows even rapidly settling species to accumulate. Diatom (e.g. *Aulacoseira* sp. *Fragilaria crotonensis) and* dinoflagellate (e.g. *Peridinium* cf. *sydneyense*) DCMs have been well documented in adjacent Lakes Tarawera and Tikitapu respectively (Ryan 2006). For a DCM to form there must be sufficient light, available nutrients and low turbulence at depth for net growth (Ryan 2006). The algal communities forming DCMs must also be tolerant to low light conditions and low temperatures a feature of many diatoms (Dokulil and Teubner, 2003). Some dinoflagellate species are adapted to low light conditions having maxmum photosynthetic rates in low light environments (Reynolds, 1997). The euphotic depth (depth at which light is 1% of that at the surface) recorded during

the period when cell count were highest at 15 m was at only 10 m i.e. the diatoms and dinoflagellates in abundance at 15 m were subject to very low light conditions.

McColl (1972) noted 128% oxygen saturation in the metalinmnion of Lake Rotokakahi. The compensation point (1.2 x Secchi disk depth) included the metalimnion implying sufficient light for algal growth within this zone. During winter high chlorophyll a concentrations were due mostly to motile dinoflagellates which may well have been regulating their position in the water column and are therefore less likely to undergo limitation by light or nutrients (Reynolds 1997). Chlorophyll *a* measurements recorded during summer and winter 2006-7 were much higher compared with historical data (Flint, 1977; Vincent 1983), particularly during winter dominance of dinoflagellates and diatoms. Past data also indicates a continuation of algal growth in winter with chlorophyll a peaking in July and August (McColl, 1972).

Values of Kd do not correspond with seasonal variations in algal biomass so therefore are probably more influenced by suspended inorganic material. Therefore the sudden improvement in light attenuation in December could be a result of a reduction in material, possibly after winter when isothermal conditions had promoted that material to remain in suspension.

Annual averaged secchi disk depths have decreased from 7.9 m in 1970-71 (McColl, 1972) to 6.8 m during 1993-94 (Burns et al., 1997) and are currently 4.8 m (2006-2007). This reduction of secchi disk depth by an increase in algal biomass due to indicated the by a comparably large increase in chlorophyll a average chlorophyll a measured from 1970-71 was 11.5 µg/L-1 while in the current data it is 17.5 $\mu g/L^{-1}$. Suspended inorganic material causing low secchi depths could also be possible however the relationship between k_d and secchi depth is not as obvious. Significant positive correlations between secchi disk depth and $\log_{10}$ total pigment concentrations in Lake Rotokakahi data have also been found (Mc Coll, 1972).

Solar quenching of surface algae is evident most months. During midday when light is at highest levels, algae reduce their fluorescing activity due to an overexposure to sunlight (Ryan 2006). This is seen particularly well in September 2007 (Fig. 4.3g), and is noted as a thin band of low fluorescence (1-2 m deep) across the surface of the lake.

6.3 Phytoplankton and zooplankton composition

In surface waters chlorophytes are common most of the year, as in Lake Rotorua (Burger et al. 2007), but more so during spring and summer. The dominanting chlorophyte species were *staurastrum* sp, which are flagellated, and not likely to be subjected to the same selection pressures as diatoms.

The dominant phytoplankton flora during winter are the dinoflagellates and diatoms. As described by Reynolds (1997), in meso-eutrophic lakes large celled dinoflagellates such as *Ceratuim* spp. and *Peridinium* spp. are typical in nutrient-enriched lakes where they can form large biomass and often succeed cyanobacterial blooms (Fig). The main bloom forming dinoflagellate *Ceratiun hirundinella* can dominate for long periods after free nutrients have been exhausted from the surface waters and their motility means they can exploit resources in strongly stratified lakes which have segregated resources (i.e. light and nutrients; Reynolds 1997). Diatoms can also dominate in winter, partly because they are adapted to low light and increase cellular chlorophyll *a* (Venrick, 1988).

Compared with a previous phytoplankton investigation conducted by Jolly in the 1950s it would appear that the phytoplankton community of Lake Rotokakahi has not changed in composition with *Aulacoseira* sp and *Ceratium* sp still the most common. Flint (1972), showed that relative to other Rotorua lakes phytoplankton species were sparse in Lake Rotokakahi. Dominant species in 1970-71 *Anabaena spiroides* in summer and the diatom *Dinobryon* sp in winter. Burns et al (1997) also states that there had been no real change in the algal community of Lakes Rotokakahi from the 1950s to 1994.

Zooplankton numbers generally follow seasonal trends in phytoplankton abundance, with high numbers are high from spring onwards to the end of summer. Bruce et modelling nutrient recycling by zooplankton in Lake Kinneret, Israel found that zooplankton could consume more than 51 % of the carbon from phytoplankton. However other Rotorua lakes including Lake Rotokakhi have experienced extensive and irregular fluctuations in zooplankton abundance from year to year particularly *Bosmina* sp and *Calamoecia* sp. Rotifers were dominant numerically and in species diversity and included *Keratella* sp, *Asplancha* sp and *Filinia* sp. Duggan, et al (2001) report that rotifers are species rich and often dominate zooplankton community assemblages. Lake Rotokakahi zooplankton community assemblage confirms with that recorded by Chapman et al. (1973) and follows general seasonal cycle of total zooplankton numbers similar to many other North Island lakes (Chapman et al. 1973).

6.4 Nutrients

Lake Rotokakahi catchment comprises of 26% pasture, 16.6% indigenous forest/scrub and 57.1% exotic forest. With a lack of urban development around the lake and restricted access it can be assumed that the majority of anthropogenic sourced nutrients are sourced from the pasture sector of the catchment. No information is readily available on how intensely the farm is used, however the land is steep further promoting nutrient runoff into Lake Rotokakahi. The one lake inflow also in close proximity to the farm may influence higher levels of phosphate and nitrate recorded in the inflow.

As the Lake inflow is probably of groundwater origin elevated levels of phosphate are possibly influenced by natural levels of phosphate associated with groundwater fed streams such as those in the Lake Taupo area (Vant & Smith 2002). Phosphate levels recorded are not particularly high and are comparable to some inflow streams to Lake Tarawera (Hamilton et al, 2006). Phosphate levels in the lake could possibly be further elevated by the farm given its steep topography of the farm and its close proximity to the inflow. Internal loading of phosphate in the lake during stratification is evident with higher concentrations recorded at 27

m as phosphate is released from the bottom sediments. In lake phosphate levels recorded in 1955 at the surface were only in trace concentrations (Jolly, 1955).

Total phosphorus trends are similar to phosphate with high levels in the inflow, and most phosphorus in the inflow in phosphate form. Within the lake TP concentrations tend to be lower and more constant although higher TP from 1-10 m (integrated sample) in May could be associated with phosphorus taken up by the large phytoplankton biomass at a time near the end of the stratified period.

Nitrate levels in the inflow are also distinctively higher than in-lake concentrations however are still relatively low compared to other Rotorua Lakes such as Lake Rotorua (Morgenstern & Gordon 2006) and Lake Tarawera (Hamilton, 2006). Ammonium increases in bottom waters, particularly near the end of stratification when the water column becomes anoxic. This loss of oxygen prevents ammonium oxidation to nitrate. In an extreme case, Priscu et al (1986) studying hypolimnetic nitrogen and phosphorus transformations in Lake Rotoiti found that N_20 reached 8,800% air saturation just prior to the hypolimnion becoming anoxic, contributing 13% of the total dissolved inorganic nitrogen (DIN) pool to the lake. It was estimated that DIN diffusing from the hypolimnion could increase epilimnion DIN by 0.34 mg m^{-3} d^{-1}. Jolly (1955) found nitrate levels in Lake Rotokakahi surface water to be below detection limits and while ammonium was detected at 0.020 µg L^{-1} .

Total nitrogen is still high relative to ammonium and nitrate levels therefore most nitrogen within the lake is in either dissolved organic form (DON) or particulate nitrogen form, with the latter most likely to be algal-related.

6.5 Kakahi density biomass and condition

In this study total kakahi (*Hyridella menziesi*) numbers are greatest at intermediate depths, with mean annual chlorophyll *a* fluorescence across the different depths being the best correlate of kakahi density, while annual mean dissolved oxygen concentrations were the best correlate for kakahi biomass.

Literature focusing on the importance of phytoplankton in the diet of kakahi is limited, however in many cases kakahi and other overseas species have been shown to have significantly reduced phytoplankton biomass (Ogilvie 1993; Ogilvie and Mitchell, 1995; Soto & Mena, 1998; White 2000; Hwang et al., 2004;). Nobes (1980) studied the energetics of kakahi within the Waikato River, Hamilton, and concluded that food supply was an important factor to their growth and survival. Alternatively Post et al., (2002) discovered that freshwater mussels are an accurate integrator of phytoplankton isotope ratios. In general terms benthic biomass can be positively related to phytoplankton stock or trophic status (Forsyth, 1978), however James (1987), studying kakahi in oligotrophic Lake Rotokawau (Rotorua) found that kakahi derive 95% of energy from allochthonous material. The relationship of kakahi to fluorescence suggests that phytoplankton could be an important part of the kakahi diet and ultimately influence their survival. The annual euphotic depth average for Lake Rotokakahi was 11.9 m corresponding approximately to the depth at which kakahi become less abundant. Assuming that most phytoplankton are within the euphotic zone, kakahi may be responding to an increased food supply (i.e. phytoplankton).

Lake Rotokakahi is stratified for approximately 8 months of the year with the hypolimnion remaining anoxic for 4-5 months. James et al. (1998) state that when oxygen levels are below 5 mg L^{-1}, the long term viability of mussel beds of *Hyridella menziesi* will be reduced. Although this study failed to find any relationship between kakahi density and dissolved oxygen levels with depth, it must be noted that for approximately five months dissolved oxygen levels in the hypolimnion of Lake Rotokakahi are well below 5 mg L^{-1} and therefore could strongly restrict kakahi movement into deeper areas. The sudden disappearance of kakahi at depths greater than 15 m was a noticeable feature at most sites and suggests that bottom limits of kakahi distributions are reflected by the upper limit of the anoxic hypolimnion. Closer to the sediment-water interface at shallower depths above the thermocline, brief periods of anoxia lasting 1-2 days were also observed. Forsyth (1978), sampling Lake Rotokakahi for macroinvertebrates bimonthly for one year, found that oxygen depletion excluded animals, including kakahi, from the deepest parts of the lake. Within the seven Rotorua lakes studied by Forsyth, *Hyridella menziesi* was known to move into shallower waters in

response to reduced dissolved oxygen levels during stratification. These brief periods of anoxia above the thermocline are due to seicheing of the thermocline causing Lake Rotokakahi compared with the other Rotorua Lakes (Okataina, Rotoma, Tikitapu, Okareka, Rotokakahi, Ngapouri and Okaro) had the highest estimated annual mean number of mussels although this was only 13 m^{-2}.

In my study I found that kakahi lost condition with depth, which may have been related to the encroaching anoxia from deeper depths. In Lake Bernard, Ontario Ghent et al. (1978) recorded stunted growth of *Anodonta grandis* with increasing depth as clams in deeper water spend a shorter portion of their growing season above the thermocline in oxygenated waters. Similarly mussels (*Alathyria jacksoni*) from the Murray Darling River, Australia, which were exposed to declining dissolved oxygen showed no ability to regulate oxygen consumption, resulting in a loss of muscle tone under extreme hypoxia By contrast mussels naturally exposed to stagnant waters (*Velesunio ambiguous*, formally known as *Hyridella australis*) could regulate oxygen consumption and would close valves, remaining inactive under extreme hypoxia (Sheldon and Walker, 1989). It is unknown weather kakahi in deeper lake regions have the same tolerance to oxygen or regain biomass after the anoxic period. Sampling before and after anoxic conditions would need to be carried out in order to verify this.

Environmental variables measured within my study such as sediment type, lake bed slope and weed cover were not found to be related to kakahi densities, however, some have been in past studies. For example James (1985), investigating the distribution and abundance of kakahi in Lake Taupo, found that kakahi densities were positively correlated with clean sand and angle of lake bed slope. Alternatively, James and Weatherhead (2001) studying macroinvertibrate distributions in 9 oligotrophic lakes in New Zealand found that kakahi were most common in fine silt and sand below macrophyte beds Certain water and bottom sediment chemistry properties have been shown to affect or even totally exclude kakahi from lakes. For example there are presumably no kakahi within adjacent Lake Tikitapu, Rotorua, due to inadequate levels of calcium needed for shell growth (Forsyth, 1978) and it is recognized the absence of kakahi in Lakes

Rotomahana, Rotorua, may possibly be due to major geothermal inputs however this cannot be verified within the literature.

Mussel size relationships (length/width, length/height, width/height) were similar between depths although mussels with larger lengths and widths tended to be located in shallower lake margins, possibly as there were better adapted to exposure to wave turbulence and coarser sands. Within Roper and Hickey (1994) Cvancara (1972) found that wider individuals were greater in shallow water and exposed areas (Hinch and Bailey, 1988). Overall, in Lake Rotokakahi at all depths sampled there was an absence of juvenile kakahi (>20 mm length) a general anomaly shared by other studies. Roper and Hickey (1994) hypothesise that post larval kakahi distribute into separate niches and recruit to the adult population several years later as larger juvenile kakahi. More recently others attribute losses in freshwater mussel recruitment to habitat degradation through anthropogenic impacts such as channel diversion and land clearance (Brim-box et al 2004; Brainwood; 2006; Playford & Walker, 2007). Specifically McDowall (2002) considers the loss of fish host species have brought about declines in kakahi and other unionid mussels.

Chapter 7

Conclusion

The objective of this study was to update water quality data last monitored in Lake Rotokakahi in 1996 while also identifying key environmental variables influencing kakahi populations.

The results of a one year water quality monitoring study on Lake Rotokakahi found an increasing period of anoxia in bottom waters by approximately 1.5-2 months and increased annual mean algal biomass and reduced secchi disk depth. Although nutrient concentrations remained moderate to historical data and the phytoplankton composition had not changed. Algal biomass and duration of anoxia in the hypolimnion during stratification are commonly used to assess trophic state and have been proven reliable methods to accurately indicate lake trophic conditions in the Rotorua lakes for over four decades. Therefore findings of this study conclude that Lake Rotokakahi water quality has declined.

On going water quality monitoring on Lake Rotokakahi is recommended to further evaluate the extent of which water quality changeis occurring. This will provide a better understanding of how Lake Rotokakahi can be best managed to further preserve the lake.

Kakahi numbers in Lake Rotokakahi were very abundant noticeable in depths above the thermocline. There are many environmental factors that influence kakahi densities and distribution in Lake Rotokakahi. Of the variables measured chlorophyll a fluorescence and dissolved oxygen were most important to kakahi density and biomass respectively. Phytoplankton as a food to kakahi may be a important factor in determining kakahi numbers while low dissolved oxygen concentrations in deeper regions could induce a loss of biomass due to an inability to regulate respiration under potentially anoxic conditions. Thermocline seicheing

has the potential to impact on kakahi in waters above the thermocline during times when the thermocline rises exposing kakahi to anoxic bottom waters. The height of the hypolimnionium during stratification in particular could be a major factor limiting kakahi distributions in deeper lake regions during periods of anoxia.

Similar studies on kakahi year are recommended preferably at set times during the year to further quantify kakahi distributions in intermediately deep monomictic lakes where kakahi are potentially exposed to low dissolved oxygen.

Appendix A

Interview Questions

Interview sessions were based on the following questions:

- Do you remember what kai was gathered from around and within the lake?
- How did you/they gather kakahi?
- Where specifically around the lake did you/they collect kakahi?
- Why did you collect them e.g. hui/tangi/personal whanau?
- Were there any particular seasons when you would collect kakahi?
- With reference to the present can you compare numbers of kakahi collected and whereabouts they were abundant?
- Can you explain any tikanga (protocols) regarding restrictions on gathering kairoto?
- How would you/they prepare kakahi for cooking?
- How would you describe the condition of the lake water back then compared with now?
- Can you remember past land-uses around the lake and were they different to the present.
- Do you have any environmental concerns regarding Lake Rotokakahi?

Interview results

Notes from an interview with Don Stafford (Local historian)

The main tribe occupying Lake Rotokakahi, Tuhourangi/Ngati Tumatawera, was originally from the Rotoiti area and descendants of Te Rangipuawhe. After various feuds with neighbouring tribes at Rotoiti the people moved to Lake Rotokakahi where they have occupied the area up to when Europeans arrived. Motutawa Island was the main stronghold of Tuhourangi/Ngati Wahiao who now resides at Whakarewarewa.

During times of European settlement Kaiteriria on the southern lake shore was another prominent occupancy. During the mid 1800s this was the position of the armed constabulary. A group of Te Arawa warriors formed to protect Rotorua from Te Kooti a highly respected warrior in New Zealand and his invading war party during the period of the Maori land wars. Kaiteriria from a protective point of view was excellent, being close to the main Rotorua-Taupo highway and far enough South of Rotorua to prevent any invasion, as it was known that Te Kooti would be traveling from the Urewera Ranges. A road was built along the southern shores of Lake Rotokakahi linking Kaiteriria with Epeha and the main Tarawera-Rotorua road to provide easier passage to Te Wairoa village at Tarawera. Epeha in close vicinity to Kaiteriria was where one of the first Christian missionaries to Rotorua was set up by Thomas Chapman. Here a church was built along with quite a number of houses and a bakery or flour mill. A stone structure, possibly a baker's oven, still remains at Epeha. In 1822 many Ngapuhi warriors were killed on Motutawa Island by the resident people as an act of vengeance for relatives killed by the same Ngapuhi warriors in Te Totara, Thames. In turn, this prompted Hongi Hika, a feared warrior chief of Ngapuhi, to attack Rotorua the following year resulting in the massacre of more than 170 Te Arawa people.

Carp (species unknown) were introduced into the Rotorua Lakes by Morrison however it remains unclear if they were released into Lake Rotokakahi. Their populations flourished in Lake Rotorua and almost became a stable diet of old people at Ohinemutu on the shores of Lake Rotorua. Brown trout (*Salmo trutta*) were released during the 1870s followed by Rainbow trout (*Oncorhynchus mykiss*), which saw the decimation of smaller native fish species such as koaro (*Galaxias brevipinnis*), and kokopu (*Galaxias argenteus*) and also the koura a freshwater crustacean (*Paranephros planifrons*) that were previously present in huge abundances. For example it was noted that koaro nets set overnight by two men from 8.30 pm to 4.00 am at Hamurana, Lake Rotorua, caught over a total of 1 ton of fish.

Farming in Rotorua began in the 1920s as soldiers retuning from the First World War were given farmland to run. The farm (Fords Farm) on the shores of Lake Rotokakahi may have been established in the late 1930s. The Whakarewrewa

State forest that comprises most of the Rotokakahi catchment began in the early 1900s as a means to overcome a predicted shortage in timber by the mid twentieth century.

Notes from an interview with Frank Maika (Tuhourangi, Chairman of the Lake Rotokakahi Board of Control and Te Rangipuawhe Maika (Tuhourangi, Chief)

Lake Rotokakahi was originally closed off to the public in 1948 due to unauthorized digging on a burial ground situated on Motutawa Island. From 1950 onwards Lake Rotokakahi was then only open to members of the Tuhourangi-Ngati Tumatawera tribes.

Extensive kai mära (cultivation areas) of kumara and fern root were situated around the lake margins on Rotokakahi. Kakahi were particularly in dense proportions around each of the islands (Motutawa and Punaruku) with no noticeable decline in numbers. Kakahi were easily obtainable as a food source and were collected in shallow lake margins by hand. Kakahi were cooked by boiling and made into a soup similar to chowder with added salt and bacon for flavour. When eaten raw kakahi had a bitter taste but in the old days were used to give to children who could easily eat them and were sustaining. Koura, toitoi and kokopu were plentiful in Rotokakahi before the introduction of the trout although it is not known exactly when or how trout became introduced. Tuna or eels (*Anguilla dieffenbachia, Anguilla australis)* have never been recorded in Rotokakahi. Tuna were mainly collected from Lake Tarawera where they were common.

There were planned exchanges of kairoto and kaimoana (seafood) between inland and coastal living people. Often people from Rotorua would huahua (preserve) certain kairoto and other kai such as birds when they were in season. The kai was later exchanged with coastal tribes who would provide kaimoana in return. Before the collection of kairoto including kakahi, certain karakia (prayers) were recited. A tapu or sacredness and mauri (life force) were associated with everything, therefore karakia were said out of respect for the environment that was provided for by the gods.

The road that linked Kaiteriria with Te Wairoa village at Tarawera was covered over after the eruption of Tarawera in 1886. However a track which followed the same route was constructed and used to transport Tupäpaku (deceased) from the Tarawera side of Lake Rotokakahi along the lake shore to Motutawa Island for burial. It has been said the Hinemoa (known in the legend of 'Hinemoa and Tutanekai') was buried there as she was a mokopuna (grandchild) of Wahiao therefore had affiliation the people of Lake Rotokakahi.

Environmental concerns for the lake included the leaching of nutrients from the farm, which may be affecting lake water quality. Fencing off of the farm boundary from the lake shore could be used so that stock cannot access the lake. A need to preserve the lake for future generations and placing some sites around Rotokakahi under the historic lands trust was also mentioned during the interview.

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