Long-term variations in fish assemblage, macrophyte community, and water quality in Lake Rotoroa (Hamilton Lake)

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Duncan Robert McKenzie Law

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Abstract

Lake Rotoroa (37º48'S, 175º16'E) is a small, shallow, polymictic lake located on the western side of Hamilton City. The lake covers an area of 0.54 km^2 with a mean depth of 2.4 m, catchment and riparian margins have been significantly modified into a suburban park-like setting. Due to its urban location and recreational value, exotic flora and fauna have been intentionally and unintentionally introduced. This has resulted in fluctuations in water quality and changes in phytoplankton, fish, and macrophyte assemblages over the past 60 years. The overall aim of this thesis is to summarise the fluctuations in water quality and macrophyte community of Lake Rotoroa associated with introduction of exotic species, and to develop a general understanding of the ecosystem response. This study involved collating and analysing available information on fish assemblages, macrophyte community, and water quality in Lake Rotoroa. Data from nine fish surveys undertaken between 1976 and 2012 has been combined. Water quality and macrophyte data was supplied by NIWA, who have undertaken monitoring for Hamilton City Council as part of the national lakes monitoring programme.

Fishing methods have varied from gill, trap, and fyke netting between 1976 and 2001, with boat electrofishing surveys between 2003 and 2012. Lake Rotoroa has a relatively diverse freshwater fish fauna, comprising two native and six exotic fish species. The fish assemblage is now dominated by the native shortfin eel (*Anguilla australis*), European perch (*Perca fluviatilis*), brown bullhead catfish (*Ameiurus nebulosus*), and tench (*Tinca tinca*), with low densities of rudd (*Scardinius erythrophthalmus*) and goldfish (*Carassius auratus*). Fish density and

biomass have varied throughout the survey period, to some extent related to the environmental conditions and macrophyte cover.

Macrophyte coverage and water quality have undergone considerable changes in the last 30 years, with the collapses of macrophytes stimulating decreases in water quality and increased perch abundance. In 1990, the macrophyte community collapsed with an associated release of nutrients into the water column, causing the lake to become supertrophic. Between 1992 and 2010, water quality improved, with a decrease phosphorus concentrations that apparently limited phytoplankton biomass and improved water clarity. This allowed macrophytes to recolonise the lake to 30% lake bed coverage in 2005 and a consequent improvement from supertrophic to a eutrophic state. Since 2009, the macrophyte community has undergone another collapse, with only a few clumps of native charophytes and *Egeria densa* present in 2011. The reduction of macrophytes has been accompanied by a decrease in water clarity. The collapse has been attributed to disturbance by grazing from the herbivorous rudd and foraging benthic feeding fish, although other stresses such as decreased water clarity and microcystins may also have had an influence.

Further research is needed on the selectivity between passive and active fish capture methods used to allow accurate comparisons between the two methods. This will allow for density and biomass estimates to be made for the passive fishing methods previously used and allow greater insight into changes in abundance of fish populations in Lake Rotoroa.

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Chapter 1: Introduction

Lake Rotoroa is situated on the western side of Hamilton City on the North Island of New Zealand (37º48'S, 175º16'E). Its catchment and riparian margins have been significantly modified into a suburban park like setting. Due to its urban location and recreational value, exotic flora and fauna have been intentionally and unintentionally introduced (Clayton and de Winton, 1994a). The consequences of this modification have resulted in fluctuations in water quality and changes in phytoplankton, fish and macrophyte assemblage over the past 60 years (Clayton and de Winton, 1994b).

This study reports on the changes in fish assemblage macrophyte cover, and water quality over the last 36 years in the context of the theory of alternative stable states in shallow lakes, as well as investigating the impact of exotic coarse fish have on the lake ecosystem.

1.1 Shallow lakes

Shallow lakes are defined as having depth that is low enough to allow year-round mixing to a point where it is "non-stratifying" ie. lacks a persistent vertical stratification because of mixing through cooling or wind turbulence (Padisák and Reynolds, 2003). The main point of difference between deep lakes and shallow lakes is that the ecology of shallow lakes is dominated by the intense sediment-water interactions as a result of persistent mixing. This allows a continual recycling of nutrients, increasing lake productivity (Scheffer, 2004). Due to mixing of the hypolimonion and the penetration of light to the sediments aquatic macrophytes play are important role in stabilising shallow lakes (Scheffer, 2004).

New Zealand has approximately 3620 shallow lakes grater then 1 ha with the biggest Lake Ellesmere at 197.81 km² (NIWA, 2007, Scheffer, 2004). Due to the lower volume of shallow lakes compared to deep lakes they respond more rapidly to catchment modification. The ecological functioning of shallow lakes in New Zealand has received less research effort then deep lakes and is less well understood. This leads to difficulties in managing and restoring these lake ecosystems (NIWA, 2007).

1.1.1 Nutrient relationships of shallow lakes

The relationship between sediment and nutrients in shallow lakes is more important than in deep lakes as the sediment is in direct contact with the photic zone throughout the year. Allowing nutrient exchange between the sediment and water column year round due to polymictic mixing of shallow lakes (Rip et al., 2005). Phosphorus and nitrogen are the fundamental two nutrients in lakes that both undergo different processes that influence there availability to biota controlling the status of a lake.

1.1.1.1 Phosphorus

Phosphorus availability is an important factor in determining water quality as it usually acts as the limiting nutrient controlling phytoplankton growth. (Søndergaard et al., 2003). Under aerobic conditions phosphorus is bound to the sediment by iron (Fe). During periods of the year when bottom waters become anoxic (when turbulence is low and microbiological consumption of oxygen is high) Fe(III) is reduced to Fe(II) causing both iron and phosphorus to come into solution and making them available for uptake by phytoplankton (Scheffer, 2004, Søndergaard et al., 2003). Turbulence in shallow lakes is a key factor controlling phosphorus release. An optimal level of turbulence mixes the water column keeping the lake aerobic. When turbulence becomes too great, sediment becomes resuspended in the water column resulting in release of phosphorus (Scheffer, 2004, Søndergaard et al., 2003). Ogilvie and Michell (1998) state as much as 80% of annual average total phosphorus concentrations can be due to sediment resuspension. The frequent mixing of shallow lakes in comparison to deep lakes results in resuspension of phosphorus over the length of summer allowing continued phytoplankton growth, whereas in deep, stratified lakes phosphorus is lost from the epilimnion until lake mixing in autumn (Jeppesen et al., 1997a).

1.1.1.2 Nitrogen

Nitrogen is less frequently a limiting nutrient in lakes (Abell et al., 2010) and differs from phosphorus in three main aspects; (1) nitrogen does not accumulate in sediment that strongly; (2) denitrification can result in a loss of nitrogen to the air; (3) some cyanobacteria are able to fix N_2 as a nutrient (Scheffer, 2004). The decomposition of nitrate in organic material through ammonification by heterotrophic bacteria, leads to a release nitrogen in the form of ammonium (NH_4^+) (Wetzel, 2001). In the top aerobic layer of sediments microbes transform ammonium to nitrate (NO_3^-) in a process called nitrification (Scheffer, 2004). As ammonium and nitrate are not readily bound to sediment particles like phosphorus there is less accumulation in the sediments, they are released into the water column where they are readily available for uptake by phytoplankton and macrophytes (Scheffer, 2004). Nitrogen is removed from most lakes either by permanent burial in sediments, outflow, or by denitrification (Windolf et al., 1996) the processes where nitrate is microbially transformed into nitrogen (N_2) gas (Scheffer, 2004).

1.1.2 Structuring role of submerged macrophytes in shallow lakes

Macrophytes have a key stabilising effect on a lake ecosystem through several key buffer mechanisms. These include providing a refuge for pelagic zooplankton (Jeppesen et al., 1997a), habitat for macroinvertebrates (Diehl and Kornijo, 1997), supressing phytoplankton growth through the release of alleopathic substances, nutrient uptake, enhanced denitrification (Sondergaard and Moss, 1997, van Donk and van de Bund, 2002), reducing turbulence and resuspension, and increasing sedimentation rates (Barko and James, 1997).

Diurnal vertical migration is expressed by pelagic zooplankton as behaviour to avoid visually feeding planktivores fish. Vertical migration can range over a few to 100 m, with the normal pattern of accent in the evening and decent in the early morning (Lampert, 1989). In shallow lakes this migration is limited by depth of the lake, causing pelagic zooplankton to be exposed to predation (Jeppesen et al., 1997a). Macrophytes provide a refuge for pelagic zooplankton during the day, this can contribute to a higher zooplankton abundance and increased grazing pressure on phytoplankton (Jeppesen et al., 1997a, Jeppesen et al., 1997b). Lauridsen et al. (1996) found that vertical migration of pelagic zooplankton from dense plant beds covering only 3% of the lake is enough to increase the grazing potential of zooplankton in open water by 100%. Macrophytes also increase the diversity of habitats and resources for macroinvertebrates while reducing the predation pressure from fish (Eklöv and Diehl, 1994). This effects the survival and growth of macroinvertebrate feeding fish, ultimately feeding back on the population dynamics of fish and other components of the lake food web (Diehl and Kornijo, 1997).

Macrophytes have the ability to store nutrients thus reducing phytoplankton biomass and growth. Van Donk & Gulati (1995) found that rapid growth of *Elodea* in Lake Zwemlust resulted in a limitation of nitrogen in the following summer and it was calculated that 64% of total lake nitrogen and 61% of total lake phosphorus (excluding the sediment pool) were found in the macrophytes.

Macrophytes have been suspected of suppressing phytoplankton growth through the excretion of chemical substances (allelochemicals) (van Donk and van de Bund, 2002). Jasser (1995) found that under experimental conditions, extract from live *Ceratophyllum demersum*, *Myriophyllum spicatum*, *Potamogeton lucens*, *Statiotes aloides*, and *Chara fragilis* resulted in a decline in the biomass and percentage contribution of cyanobacteria to total algal biomass. The effects of allelopathy by macrophytes at an ecosystem level have yet to be tested (van Donk and van de Bund, 2002).

Aquatic macrophytes can reduce sediment resuspension and erosion by reducing or redirecting turbulent water conditions while also acting as sediment traps via the interception of suspended sediments (Carpenter and Lodge, 1986). This has important role regulating sediment related to water quality problems such as reduced water clarity, enhanced nutrient cycling, and high phytoplankton biomass (Barko and James, 1997).

1.1.3 Alternative stable states

Shallow lakes are generally considered to be in one of two alternative stable states; a clear water, macrophyte-dominated state or a turbid state, where phytoplankton dominate with few submerged macrophytes (Scheffer, 2004). A number of mechanisms control the stability of each state these include, wave action, nutrient availability, turbidity, population dynamics of submerged macrophytes, and both piscivorous and benthos/zooplankton-feeding fish (Blindow et al., 1993, Scheffer and Van Nes, 2007). A lakes' trophic state switches when water clarity is reduced to a point where light is inhibited enough to cause significant stresses on the macrophyte beds resulting in a die back (Scheffer and Van Nes, 2007). This causes a large release in nutrients and an increase in resuspension of sediment, further increasing turbidity and preventing reestablishment of macrophytes while promoting phytoplankton growth. Scheffer et al. (1993) illustrated a mechanism leading to a changing trophic state in a simple geographical model (Figure 1.1). This model is based on three assumptions; (1) turbidity increases with the nutrient level due to increased phytoplankton growth; (2) aquatic vegetation reduces turbidity, and (3) vegetation disappears when a critical turbidity is exceeded.

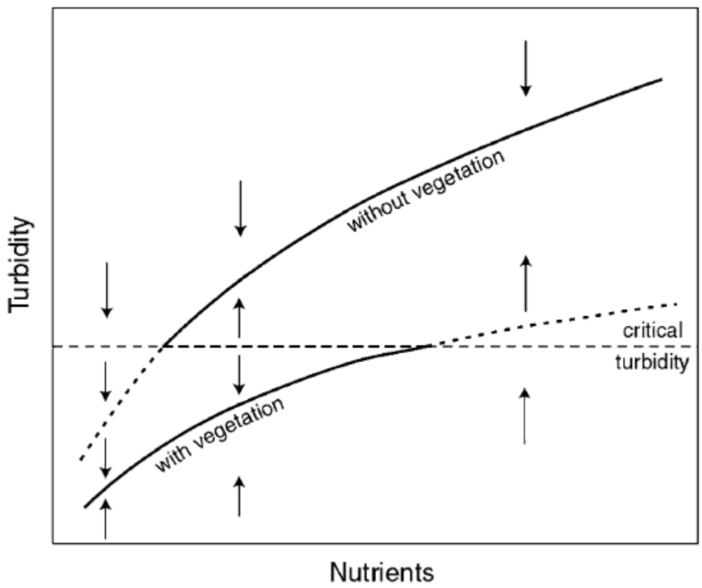

Figure 1.1 Alternative equilibrium turbidities caused by disappearance of submerged vegetation when a critical turbidity is exceeded (from Scheffer et al., 1993)

The Scheffer et al. (1993) model excludes complex environmental factors that can influence the state of shallow lakes. These factors can include seasonality effects, extreme weather conditions and spatial heterogeneity within the lake, and can cause lakes to switch between clear and turbid states over a short period of time (Scheffer and Van Nes, 2007). For example in 1968, Lake Ellesmere, South Island, New Zealand switched from a clear state to a turbid state when a tropical cyclone destroyed extensive submerged macrophyte beds of *Ruppia megacarpa* and *Potamogeton pectinatus*, allowing sediment to be resuspended year round and allowing phytoplankton to dominate (McKinnon and Mitchell, 1994, Williams, 1979). A switch between alternative states generally occurs irregularly, but some lakes show periodic shifts between clear and turbid states (Scheffer and Van Nes, 2007). For example, Lakes Grote Wije and Kleine Wije, Botshol, Netherlands, are two naturally clear shallow lakes that had become turbid due to eutrophication. In an effort to switch the lakes back to the natural state, external phosphorus loading were reduced. After four years both lakes switched back to a clear state but this was not stable and in the 15 years following both lakes switched between alternative states four times (Rip et al., 2005). Periodic switching is predicted to occur when the critical nutrient levels for both lake states are close together and there is a strong relationship between nutrient levels and vegetation (Scheffer and Van Nes, 2007).

1.1.4 Trophic interactions and nutrient conditions

An increase in nutrients through an alteration of land use such as intensification of agriculture (MacLeod and Moller, 2006), can lead to changes in phytoplankton biomass and species composition (Williams and Moss, 2003). Alterations to the ecosystem can also occur through top-down or bottom-up trophic mechanisms. Top-down mechanisms imply that the alteration of a lake ecosystem through changes made at the top trophic level that flow down through the ecosystem to lower trophic levels. Bottom-up mechanisms imply that the processes are dominated by the modification of nutrient availability, which flows up to the higher trophic levels through the ecosystem (Jeppesen et al., 1997a, Scheffer, 2004, Williams and Moss, 2003).

1.1.4.1 Top down modifications

The top-down control is manipulated by modifying the effect of fish predation on determining the size and composition of zooplankton communities, with the flow on effect of increased grazing of phytoplankton (Williams and Moss, 2003, Scheffer, 2004). This is supported by trophic cascade theory, which states that piscivorous fish keep planktivorous fish at a low density, which results in higher abundance of large zooplankton which are efficient at causing significant decline in phytoplankton biomass (Carpenter et al., 1985, Friederichs et al., 2011). In lakes without piscivores, there should be a high phytoplankton due to low zooplankton grazing pressure (Friederichs et al., 2011). For example, in Lake Wolderwijd, Netherlands, the biomass of bream (*Abramis brama*) and roach (*Rutilus rutilus*) was reduced, and pike fingerlings (*Esox lucius*) were introduced. This resulted in a short term improvement in water clarity through reducing grazing predation on *Daphnia galeata*. In the long term this allowed gradual spread of *Chara* meadows, decreasing sediment resuspension with associated improvements in water clarity (Meijer and Hosper, 1997). Danner and Hambright (2002) revised 17 studies of top-down control and found only seven cases had reduced phytoplankton biomass due to manipulation of piscivore density. They also noted that phytoplankton abundance per unit of phosphorus was significantly lower in lakes with piscivores relative to those lakes without, suggesting some type of top-down control.

Effect of top-down control in shallow lakes is limited by depth, through preventing vertical migration of zooplankton and providing alternative benthic food sources for planktivores. Vertical migration by zooplankton is known to be a means of avoiding predation from fish, is limited by depth in shallow lakes (Jeppesen et al., 1997a). Secondly, plankti-benthivorous fish such as rudd (*Scardinius erythrophthalmus*) rely more on benthic feeding in shallow lakes as benthic invertebrate biomass and production at a given total phosphorus level is higher in shallow lakes. This means that fish are less sensitive to variations in zooplankton abundance, allowing fish population density to remain high when zooplankton levels are low (Jeppesen et al., 1997a). Controlling benthivorous fish also influences bottom up processes with fish forging for benthic invertebrates resulting in the sediment resuspension (Havens, 1993). Alternatively, increased grazing zooplankton abundance can result in dominance by large filamentous cyanobacteria that are not easily consumed by zooplankton (Schoenberg and Carlson, 1984).

1.1.4.2 Bottom up interactions

Bottom-up processes are influenced by nutrients in the lake ecosystem. Controlling nutrient loading in lakes limiting phytoplankton growth increasing water clarity and potentially promoting macrophyte development (Scheffer, 2004). Jeppesen et al. (2007) found that by reducing phosphorus loading in 22 lakes resulted in slow recoveries (10-15 years) from eutrophic state. This long delay is due to the internal loading of phosphorus within the lake which requires at least three retention times to wash out 95% of the excess pool of phosphorus in a fully mixed lake (Jeppesen et al., 2005). External nutrient loadings can be reduced through improved waste water treatment (Jeppesen et al., 2007), treatment of agricultural runoff through constructed wetlands (Tanner et al., 2005) and riparian margins (Özkundakci et al., 2010). The reduction of internal loading of phosphorus can be achieved by phosphorus precipitation (Özkundakci et al., 2010) or by the capping of sediments with clay minerals (Berg et al., 2004, Robb et al., 2003). The application aluminium sulphate and zeolite to the small eutrophic Lake Okaro, resulted in a 41% decrease of total phosphorus from the water column between 2004-2005 and 2007-2008 (Özkundakci et al., 2010).

1.2 Aims and objectives

The overall aim of this thesis is to develop a general understanding of the fluctuations in water quality and macrophyte community of Lake Rotoroa associated with introduction of exotic species into the ecosystem, by addressing four key objectives:

1. Review previous studies of ecology and water quality to provide a summary on the current state of Lake Rotoroa.
2. Summarise changes in fish assemblage within Lake Rotoroa.
3. Provide a review of changes in the lake's water quality and nutrients over the past 20 years.
4. Evaluate relationships between lake water quality, fish assemblages and fluctuations in macrophyte cover within Lake Rotoroa.

These objectives were addressed by;

1. Collating available fish surveys in Lake Rotoroa and analysing data for changes in species assemblage catch rates, size and relative abundance.
2. Analysing lake monitoring data (nutrients, Secchi depth, chlorophyll *a*, and total suspended solids), lake's trophic status, and associate changes in key nutrient variables.
3. Identifying possible causes of the decline of macrophytes in Lake Rotoroa.

1.3 Study site – Lake Rotoroa

Lake Rotoroa (37º48'S, 175º16'E) is a small shallow lake located on the western side of Hamilton city, managed by the Hamilton City Council as a park reserve.

Lake Rotoroa covers an area of 0.54 km^2 with a mean depth of 2.4 m, with a northern and southern basin with depths of 5.4 m and 6.5 m, respectively (Clayton and de Winton, 1994b). Over 54% of the lake is less than 2 m deep and 25% less than 1 m deep (Figure 1.2) (Tanner et al., 1990). The lake's catchment is 1.38 km^2 compromised of 40% lake surface, 25% residential housing, and 35% recreational reserve (Tanner et al., 1990). Water enters the lake directly via numerous storm water outlets around the lake margin, overland flow, or through ground water seepage. The outflow is maintained artificially via a weir on the western side of the lake, which flows into the Waitewhiriwhiri Stream via a piped drain (Hamilton City Council, 2010). Lake Rotoroa has a volume of 1.3×10^6 m^3 with an estimated residence time of 2.4 years (Dugdale et al., 2006).

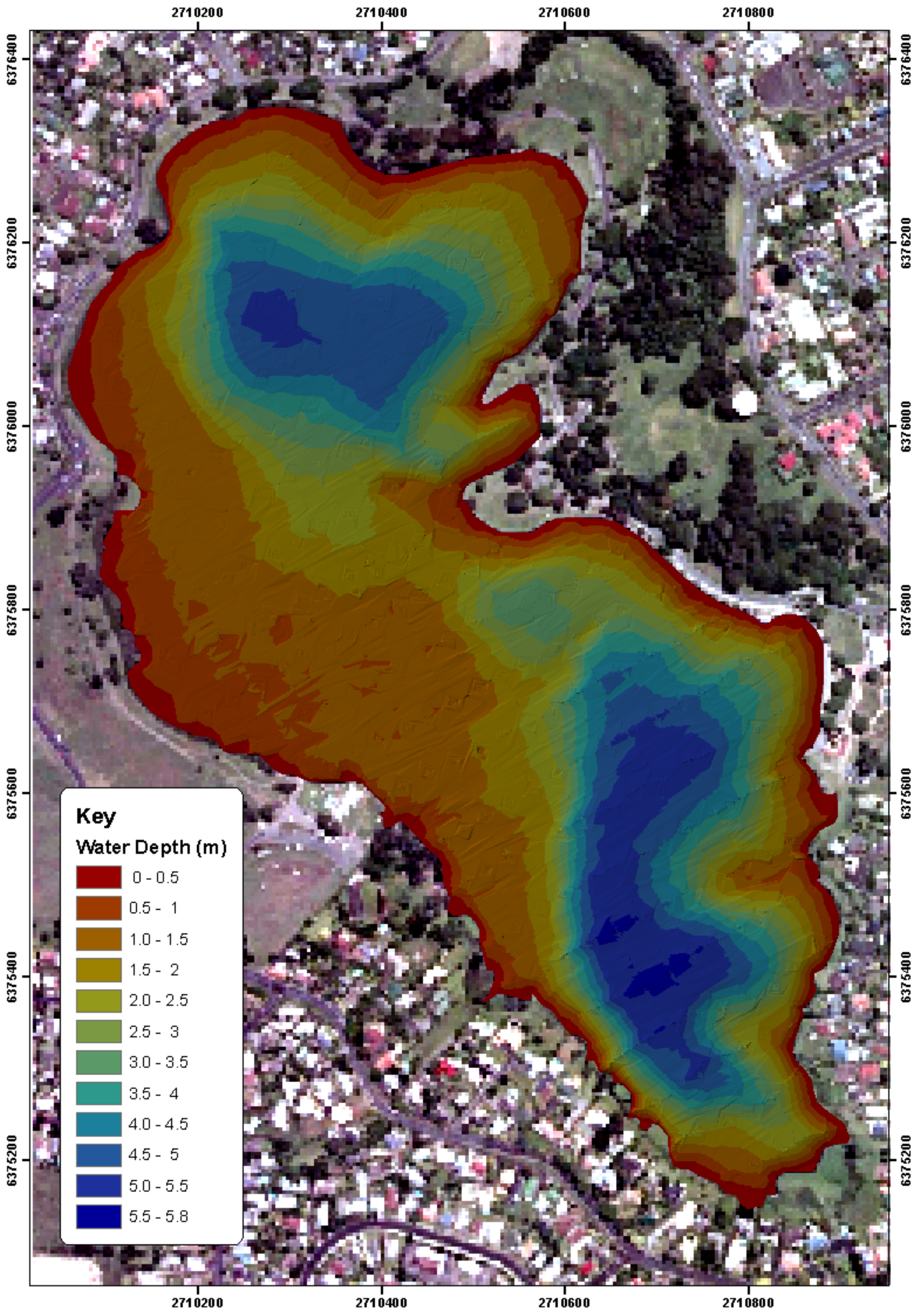

Figure 1.2: Bathymetric contours for Lake Rotoroa, winter 2004, as measured by differential GPS/sonar and plotted by GIS software application Arc Map® (de Winton et al., 2004)

1.4 General setting

The Hamilton basin is located in the Waikato region of the North Island, New Zealand. The surrounding area consists of a broad alluvial plain with prominent widely spaced rounded hills ranging between 30 m and 70 m in height (Edbrooke, 2005). The underlying geology is late quaternary pumice dominated alluvium which blankets the plain derived from remobilisation of sediment by the Waikato River following the Orunanui eruption 26,500 years ago. The river overtopped the Maungatautari Gorge flowing into the Hamilton basin depositing large amounts of sediment partially filling the valleys of the existing landscape. 17,000 years ago the supply of sediment reduced causing the erosion of the current Waikato River valley through the Hamilton basin (Collier et al., 2010b).

1.4.1 Hamilton City

Hamilton City is the fourth largest urban area in New Zealand with a population of 171,600, covering a highly modified area of 9,427 ha (Clarkson and McQueen, 2004). The natural environment of Hamilton was originally dominated by lowland hills, and large peat bogs drained by an extensive gully system. Indigenous vegetation consisted of rimu-tawa forest on the lowland hills, a mixed conifer-broadleaved forest on the well-drained ridges, swamp forest and shrub land on the margins of the peat bogs, with sedge shrub land in the peat bogs (Clarkson et al., 2007). Extensive modification of the natural environment has occurred since the arrival of humans, early Maori first transformed the landscape through fire and cultivation. Since the arrival of European the landscape has been transformed dramatically with agricultural development, urbanisation and their associated infrastructure causing large destruction of natural vegetation, little now remains (Hamilton City Council, 2007).

1.4.2 Climate

Hamilton has a temperate, damp climate characterised by warm humid summers, mild winters and moderate rainfall (Collier et al., 2010b). Mean annual daily maximum temperature of 18.8 °C with a mean of 23.6 °C in summer (December to February) and 14.1 °C in winter (June to August). Annual rainfall is 1230 mm yr^{-1} with rainfall typically higher in winter then summer, mean annual relative humidity of 85%. The prevailing wind direction is from the west (long term means from NIWA climate database 1990-2011).

1.5 Lake history

1.5.1 Origin

Lake Rotoroa has similar geomorphological history to many other small peat lakes in the Hamilton basin. These lakes originated 15,000 -17,000 years ago during the final stages of the deposition of the Hinuera Formation by the ancestral Waikato River (McCraw, 2011). The formations consist of muds, sands and gravels deposited by the river to form a low angled alluvial fan forming at the south near Cambridge. During this period the river consisted of multiple braided channels that migrated across the surface of the fan. As these channels drifted across they formed levees, bars, and cut off embankments in the older hilly landscape impounding the water to form small lakes (Green and Lowe, 1994). Once the river settled in its current channel peat bogs began to develop encroaching on the many lakes within the Hamilton basin causing the water levels to rise and turning their waters acidic and brown (Green and Lowe, 1994).

Lake Rotoroa formed originally as two small lakes, the current northern and southern basins approximately 4 and 5 m deep, respectively. The growth of the Rukuhia bog caused the water level to rise as the peat level grew higher, submerging the low spur to form one lake (McCraw, 2011).

1.5.2 Palaeolimnology of Lake Rotoroa

The Waikato lakes are among the oldest lakes of New Zealand forming between 15,000 and 17,000 years ago. Their sedimentary record spans major climatic changes between the end of the last glaciation and the warmer climates in the last 10,000 years. Speirs (1995) described four sediment core samples from the southern basin, sampling 3.5 m of sediment from the lacustrine sediment down to and the Hinuera formation dating back to the initial formation of the lake. The average sedimentation rate was calculated at 0.2 mm yr^{-1} but had considerable variation between 0.02 mm yr^{-1} and 3.2 mm yr^{-1}. Speirs (1995) attributed the high rates of sedimentation to the early lake development and erosion of the newly deposited Hinuera formation. After this period the sedimentation rates decline to between 0.04 mm yr^{-1} and 0.85 mm yr^{-1}.

1.5.3 Developmental history of Lake Rotoroa

Speirs (1995) divided the development of the lake into five periods based on the analysis completed on the sediment cores. This includes analysis of sedimentology, fossil pigments, cladocera and Chironomidae which were related to water quality and climate.

Sediment deposits from the river spans the period of lake formation and development (17,000 – 14,400 years B.P). Sediments grade from alluvial sands and muds to lacustrine, the lake appeared to be shallow with low productivity. After 16,000 years, the lake appears to have deepened and increased in size with a change in planktonic species. Evidence also suggests that there were frequent inputs of allochthonous material from flooding of the Waikato River.

The further deepening of the lake and decline of allochthonous inputs occurred as the Rukuhia peat bog began to develop, 14,400 - 5,300 years B.P. The change to a wetter climate is associated with an increase of lake area and establishment of littoral macrophytes, leading to increased habitat and greater species diversity. Later during this period increased peat development would have caused the lake to continue to deepen, flooding the shallow littoral macrophytes, declining water clarity, and increasing acidity.

The drying of the climate resulted in the reduced growth of the peat bog 5,300 - 3,300 B.P. The drying of the marginal peat allowed the reestablishment of shallow clear water areas with growth of littoral macrophytes, increases in productivity, and continued low clarity of the lake waters.

Lake productivity continued to rise in Zone four, 3,300-1,650 B.P., dominated by submerged and littoral macrophyte development. The climate may have become wetter and inputs of allochthonous material eroded form the regressing peat by streams. The lake continued to deepen and water clarity was poor.

From 1,650 B.P. to present the lake was characterised by a massive increase in productivity over a 100-year period, and productivity remained high throughout the rest of the period; the cause of the rapid eutrophication is undefined (Speirs, 1995). It is possible this relates to the arrival of humans into the Waikato and the large scale modification of the terrestrial environment leading to increased nutrient inputs.

1.6 Management history of Lake Rotoroa

Lake Rotoroa is a crown-owned recreational reserve controlled by the Department of Conservation but managed by the Hamilton City Council in terms of the Reserves Act 1977 (Featherston, 1994). Lake Rotoroa and the surrounding recreational reserve, collectively known as the Hamilton Lake Domain, is highly valued by the Hamilton community for its recreational and biodiversity values (Hamilton City Council, 1985).

1.6.1 Historical management

The Lake Domain was set aside during the original survey of Hamilton West in 1864 as part of the “west town belt”. The domains’ appearance at this time reflected a history of fires and bush clearance. Initial clearing and planting at the southern end of the lake was the first attempt to create formal gardens, the rest of the lake remained in scrub with some grazing in the well-drained areas. The use of the lake and the surrounding reserve gradually increased with the reserve being used for concerts, boat carnivals, moonlight parties, and rowing races. From 1913 onwards the rest of the lake domain was cleared and developed for recreational use, starting on the eastern side before the development of the western side in the 1950's into what now is referred to as "Innes Common" (Hamilton City Council, 2010).

1.6.2 Hamilton City Council management plans

In 1985 the Hamilton City Council was one of the first territorial authorities in New Zealand to create a formal management plan for the lake (Coffey et al., no date). The objectives of this plan included; the enhancement of the aesthetic appeal; its maintenance as a scenic and recreational attraction; to maintain and improve the water quality; management of species diversity for birdlife, fish, and aquatic vegetation; and the maintenance of Lake Rotoroa in such a manner that established water based recreation activities could continue to use the lake (Hamilton City Council, 1985, Coffey et al., no date). Until the first plan was implemented in 1985 the water quality and aquatic vegetation of the lake was managed on a narrow set of criteria and limited access to scientific expertise. This plan focused on eliminating aquatic weed for increasing the lake suitability for yachting. Hamilton City Council has continually revised the management plan for the lake domain. The most recent revision became operational in 2010 with the overriding aim to "protect the natural environment, while providing for public access, outdoor recreation activities and open space" (Hamilton City Council, 2010). The key ecological objectives are;

1. To manage Hamilton Lake Domain and its catchment as a balanced sustainable and diverse ecological system giving particular recognition to the importance of water quality.
2. To conserve and enhance the natural character and scenic environment of Hamilton Lake Domain.
3. To manage activities at the lake to minimise exposure to contaminants in the lake bed sediments.
4. Retain a wide range of recreation opportunities and enhance existing facilities, consistent with the need to protect and enhance natural habitat.

1.7 History of Lake Rotoroa water quality monitoring

Water quality monitoring in Lake Rotoroa first began in the late 1970s (de Winton, 1994b). Early sampling was done in relation to other scientific studies, not on a basis of identifying the lake status. In 1976 and 1977 Hamilton City health inspectors collected two years of water quality data to supplement the perch (*Perca fluviatilis*) fishery research during the same period (Graynoth, Unpublished). Lake Rotoroa at this time was oligotrophic or marginally mesotrophic. The next sampling occurred 1978 and 1979 by the Waikato Valley Authority (now Waikato Regional Council) in relation to the use of diquat herbicide to control the exotic macrophyte *Egeria densa* (Henriques, 1979). The University of Waikato subsequently collected samples from 1978 to 1980 as a component a plankton studies (Chapman & Green, unpublished data). Since 1981 continued monitoring has been carried out initially by the Water Quality Centre (now NIWA) as part of a survey of small Waikato lake survey (Town, 1981, Boswell et al., 1985). From 1985 the Aquatic Plant Group, MAF, (now NIWA) where contracted by the Hamilton City Council to undertake water quality monitoring as a requirement for using the herbicide diquat. This was run concurrently with a monitoring programme by the Waikato Valley Authority from 1988. Since 1992 Lake Rotoroa was included in the national water quality network run by NIWA and sampling has continued under contract from Hamilton City Council (de Winton, 1994b).

1.8 Ecology of Lake Rotoroa

1.8.1 Phytoplankton species

Etheredge (1987) found Lake Rotoroa contained 146 phytoplankton species with 38 major species. Ten species of euchlorophytes and 7 species of diatoms where the most common classes, *Tetrastrum triangulare*, *Dinobryon cylindricum* and *Cyclotella stekkigera* where the most abundant, but due to its larger size *Peridinium* sp. made up 69% of the biomass (Etheredge, 1987). By 1990 the dominant phytoplankton had changed with Edgar (1993) noting that the dominant phytoplankton was *Botryococcus braunii* forming over 90% of the biomass. Since the 1990s monitoring has shown that changes in phytoplankton population occur seasonally through a range of species with an overall decrease in biomass observed from 1992 to 2003. Between 1992 and 1999 four species dominated the biomass, green algae (*Botryococcus braunii* and *Coelastrum reticulatium*), chrysophyte (*Dinobryon divergens*), and the dinoflagellate, *Peridinium playfairii* (de Winton et al., 1999). Between 1995 and 1999 the dominance of green algae declined with periods of dominance by diatoms (*Asterionella formosa*, *Cyclotella meneghiniane*), desmids (*Coelastrum reticulatium*, *Cosmarium bioculatum*) and dinoflagellates (*Peridinium playfairii*, and *Certium hirundinella)* (de Winton et al., 2000). Phytoplankton species were stable until 2003 when the chlorophyta species *Coelastrum cambricum* and *Botryococcus braunii* became increasingly dominant (de Winton et al., 2005). Between 2005 and 2009 dinoflagellates were again the dominant contributor to biomass with *Peridinium* sp. the largest contributor, *Ceratium hirundinella* and *Gymnodinium* sp. were also significant (de Winton et al., 2011). Since 2009 summer blooms of toxic cyanobacteria have occurred consisting of *Microcystis* sp., *Chroococcus* sp., *Dolichospermum* sp. (previously named *Anabaena*) and *Coelosphaerium* sp. The levels of biovolumes varied over the summer seasons with the bloom continuing into autumn of 2011 (de Winton et al., 2011).

1.8.2 Zooplankton community

Waikato Lakes have a generally diverse zooplankton community. Lake Rotoroa consists of a range of copepod and rotifers. In the year 2000 the copepod community was dominated by the copepod nauplii shared with *Daphnia*, with other Crustacea being calanoid copepodites, *Bosmina* and *Biapertura* (White, 2000). The Rotifer community consists of eight individual species (Duggan et al., 2002). Rotifer community likely changes with trophic condition inorganic turbidity, important in determining rotifer distributions in some Waikato Lakes (Duggan et al., 2002).

1.8.3 Submerged macrophytes

Submerged macrophytes communities in Lake Rotoroa have undergone major changes in species composition and abundance (Table 1.1). Tanner (1990) noted that changes in submerged macrophyte community represent a complex interplay between species presence, competition, succession and the effect of weed control measures. Prior to 1950 Lake Rotoroa was colonised by underwater meadows of characean algae and native vascular hydrophytes in the shallower waters (Coffey et al., no date). During the early 1950s the oxygen weed *Elodea canadensis* was present but coexisted with native vegetation (Table 1.1). In the late 1950's *Lagarosiphon major* was introduced and displaced the native vegetation. Growing to a maximum depth of 5.0 m and reached the surface in depths less 4.0 m interfering with recreational yachting activities and posing a hazard for swimmers. In 1977 *Egeria densa* was first recorded and rapidly replaced *Lagarosiphon major* as the dominant submerged macrophyte (Coffey et al., no date). Native macrophytes (Table 1.1) continued to occupy shallow areas out of *Egeria densa* and *Lagarosiphon major* depth range. In 1989 the macrophyte community collapsed with no submerged macrophytes recorded between 1989 and 1992 before a slow reestablishment of native charophytes beginning in 1994 (Burns et al., 1995) to a peak in 2006 (de Winton et al., 2006) where a gradual decline began with almost a complete loss of charophyte cover in 2011 (de Winton et al., 2011).

Table 1.1: Submerged macrophyte records for Lake Rotoroa.
1973-1989 Adapted from Coffey *et al.* (no date). 1994 (Burns et al., 1995), 1997 (Burns et al., 1997), 2003(de Winton et al., 2004), and 2011 (de Winton et al., 2011). m= depth range (m), NS = present but coverage not surveyed, NR = Not recorded as an individual species, P = Present, (S) sparse coverage (< 50%), (A) abundant coverage (> 50%).

Species	**1973**	**1976**	**1986**	**1989**	**1994**	**1997**	**2003**	**2011**
Charophytes								
Chara corallina	P	P	0.2-4.0 m	-	1-1.5 m (S)	0.7 (S)	0.5-1.8 m (A)	-
Nitella cristata	NR	NR	P	-	1-1.5 m (S)	0.7-1.6 m (S)	0.5-1.8 m (A)	-
Nitella hookeri	P	P	0.3-5.2 m	-	1-1.5 m (S)	0.7-1.6 m (S)	0.5-1.8 m (A)	-
Nitella pseudoflabellata	P	P	0.4-2.0 m	-	1-1.5 m (S)	0.7 (S)	0.5-1.8 m (A)	-
Trachoephytes								
Callitriche hamulata	NS	NS	P	-	-	-	-	-
Egeria densa	-	P	0.2-5.0 m	-	-	-	(S)	(S)
Elodea canadensis	0.4-5.0 m	1.0-3.0 m	1.0-1.2 m	-	-	-	-	-
Glossostigma submersum	P	P	0.5 m	-	-	-	-	-
Lagarosiphon major	0.4-5.0 m	P	0.2-5.0 m	-	-	-	-	-
Limosella lineata	NS	NS	0.2-0.4 m	-	-	-	-	-
Myriophyllum proinquum	NS	NS	0.5-1.0 m	-	-	-	-	-
Potamogeton cheesmanii	1.0-2.5 m	P	0.2-2.0 m	-	-	-	-	-
Potamogeton crispus	0.3-2.0 m	P	0.3-1.5 m	-	1-1.5 m (S)	-	-	-
Potamogeton ochreatus	0.5-4.0 m	P	1.0-4.5 m	-	-	-	(S)	-

1.8.3.1 Submerged macrophyte control

Disruption of recreational activities due to growth of *Lagarosiphon major* and *Egeria densa* in the 1950's lead to the Hamilton City Council to review weed control options. The most effective was the application of sodium arsenite in 1959, which has been described as having "spectacular results" with a complete absence of submerged macrophytes for about five years. However, concerns on the toxicity persistence of sodium arsenite it was discouraged from further use (Tanner et al., 1990). Regrowth of *Egeria densa* beds were treated with diquat, an herbicide that affects vascular species with native charophytes recovering quickly post treatment. Diquat has been used on several occasions with whole lake coverage in 1971, 1974, 1982, 1985, and 1986 with targeted placement between 1987 and the macrophyte collapse in 1989 (Tanner et al., 1990). Since the reintroduction of *Egeria densa* in 2004 hand removal and selective diquat applications have been used to minimise spread (de Winton et al., 2011).

1.8.4 Emergent macrophytes

Emergent macrophytes are important as habitat for nesting birds providing an area for nest establishment. As well as protection of lake banks from wave erosion, covering approximately 50% of the lake shore. The dominant species are listed in Table 1.2. *Iris pseudacorus* is heavily controlled with herbicide, with only a small quantity of viable rhizomes present (Hamilton City Council, 2010).

Table 1.2: Dominant emergent macrophyte species in Lake Rotoroa 1993-2012 (Champion et al., 1993)

Species	Abundance
Baumea articulata	Common
Eleocharis sphacelata	Common
Iris pseudacorus	Sparse (Controlled)
Nymphaea spp.	Common
Typha orientalis	Common

1.8.5 Heavy metal and arsenic build up in the lake sediments and biota.

The application of sodium arsenite to Lake Rotoroa has resulted in health concerns over the toxicity levels of arsenic in sediments, fish and aquatic macrophytes. Results by Tanner and Clayton (1990) showed that arsenic levels where high in macrophytes and surficial sediments with 193-1200 mg kg^{-1} wet weight, and 540-780 mg kg^{-1} wet weight, respectively. Arsenite levels where low in flesh samples from fish and birds, with rudd having the highest concentration of 5.5 mg kg^{-1} wet weight in gut contents. Arsenic levels in flesh where all well below the maximum level of arsenic permissible for human consumption of 2 mg kg^{-1} (Tanner and Clayton, 1990, Department of Health, 1984). Kane (1995) confirmed the low arsenic levels previously reported with the highest levels recorded in catfish (*Ameiurus nebulosus*) with 0.778 mg kg^{-1} wet weight.

Storm water has been confirmed to be a source of heavy metals other than arsenic entering the lake. Analysis of concentration of heavy metals in fish flesh by Kane (1995), found that lead and copper were well below maximum permitted levels for human consumption of 2 mg kg^{-1} and 30 mg kg^{-1}, respectively. Zinc recorded in goldfish (*Carassius auratus*), rudd and shortfin eel (*Anguilla australis*) were, 11.9 mg kg^{-1}, 11.2 mg kg^{-1}, and 11.6 mg kg^{-1}, respectively. Considered high but still below the maximum permitted levels for human consumption of 40 mg kg^{-1} (Department of Health, 1984).

Chapter 2: Fish assemblage of Lake Rotoroa 1976-2012

1.9 Introduction

Lake Rotoroa has a relatively diverse freshwater fish fauna by New Zealand standards, comprising two native and six exotic species (Table 2.1). This chapter will summarise changes in fish assemblage between 1976 and 2012, providing possible explanations into the population structure and condition of fish in Lake Rotoroa.

1.9.1 Native fish

Lake Rotoroa originally comprised of four native fish species (Table 2.1). Shortfin and longfin eels, both diadromous species, requiring passage upstream from the sea as juveniles and passage downstream to spawn as adults (Hicks, 1994). Common bullies can be catadromous or land locked, it is unknown if common bullies in Lake Rotoroa were once seagoing (McDowall, 1990). The current status of common smelt and longfin eel in Lake Rotoroa is unknown. It is likely that due to predation by perch, smelt are now eradicated from the lake (Coffey et al., no date).

Table 2.1 Fish species of Lake Rotoroa

Common Name	Scientific name	Abundance
Native:		
Common bully	*Gobiomorphus cotidianus*	Sparse
Common smelt	*Retropinna retropinna*	Unknown
Longfin eel	*Anguilla dieffenbachii*	Unknown
Shortfin eel	*Anguilla australis*	Common
Exotic:		
Brown bullhead catfish	*Ameiurus nebulosus*	Common
Goldfish	*Carassius auratus*	Sparse
Mosquitofish	*Gambusia affinis*	Common
European perch	*Perca fluviatilis*	Common
Rudd	*Scardinius erythrophthalmus*	Common
Tench	*Tinca tinca*	Common

1.9.2 Exotic fish

Since the settlement of Europeans into Waikato area the fauna of Lake Rotoroa has changed significantly. Six exotic fish species have successfully been legally and illegally introduced. European perch were the first exotic fish to be introduced and were released by the Waikato Angling Club in approximately 1907 in an effort to create a coarse fishery. A previous introduction into New Zealand 1885 was unsuccessful. Perch were one of the first exotic fish introduced and liberated throughout New Zealand between 1868 and 1877. Their diets range between invertebrates, zooplankton and small fish (McDowall, 1990). Mosquitofish and goldfish were introduce some time before 1976 as they were both well-established before the first fish survey by Graynoth in 1976 (Table 2.2). Goldfish are thought to be either introduced intentionally as part of an effort to create a course fishery, or released as unwanted aquarium fish (Coffey et al., no date). Mosquitofish were introduced into New Zealand for experimental control of mosquitos, and can now be found almost anywhere there are suitable habitats north of the Waikato and Bay of Plenty (McDowall, 1990). Brown bullhead catfish where first caught in

1977 (Graynoth, 1978 Unpublished data), it is unknown if they were released intentionally, unintentionally transferred by fyke net (used for commercial capture of eels), or migrated into the lake from the Waikato River. Catfish have been present in the Waikato River since 1900s (McDowall, 1990). Rudd was introduced into New Zealand in 1967 and has since been spread to a number of lakes, ponds, and rivers. Rudd are omnivores, and eat a wide verity of vegetation and invertebrates, with a normal size range of 200-250 mm although can reach up to 410 mm (Hicks, 2001, McDowall, 1990). Tench were introduced into Lake Rotoroa in 1990 by Auckland/Waikato Regional Fish and Game Council, as part of an effort to increase the opportunities for course fishing in the Waikato (Kane, 1995, Hicks, 1994). Tench are large (up to 700 mm), olive-green to dark bronze, mainly found in shallow, still or slow moving water. Diet studies indicate tench are benthivorous and planktivores consuming macroinvertebrates and zooplankton. Tench are regarded slow-growing species that undergo large periods of inactivity with a low intake of prey (Perrow et al., 1996). They been shown to reduce macrophyte development by stimulation greater periphyton growth (Rowe, 2004, Williams et al., 2002).

1.10 Methods

1.10.1 Fishing method

Method of fishing has changed over time (Table 2.2). Each scientific study has used different sampling methods and/or sampling effort focused around specific objectives. Graynoth (unpublished data) (survey 1) focused on assessing the value of a perch sports fishery. Wise (1990) (survey 2) sampling focused on the biology of the exotic species, perch, rudd, and brown bullhead catfish. Sampling of native species was not recorded. Kane (1995) (survey 3) and Roberts (2002) (survey 4 focused on estimating the biomass, abundance and population structure of all fish in the lake. Hicks (unpublished) (surveys 5-9) focused on mean fish density and mean fish biomass. Alternative methods other than those listed below have been used but there results or effort proved poor or were not replicated over time therefor have not been used in this research. In all surveys, fish length was recorded.

Table 2.2 Scientific fish surveys completed on Lake Rotoroa.

Survey Number	Start	Finish	Net/type	Sites	Fish recorded	Source
1a	July 1976	April 1977	Gill/Fyke/Trap	Northern Basin	Perch, Eel, Goldfish	Graynoth (unpublished)
1b	July 1977	July 1978	Gill/Fyke/Trap	Northern Basin	Perch, Eel, Goldfish	Graynoth (unpublished)
2	March 1989	February 1990	Fyke/Gill	3 Shore, 2 Centre	Perch, Rudd, Catfish	Wise (1990)
3	December 1993	February 1994	Fyke/Gill	8 Shore, 3 Centre	Perch, Rudd, Catfish, Goldfish, Trench, Eel	Kane (1995)
4	February 2001	May 2001	Fyke/Gill	6 Shore, 1 Centre	Perch, Rudd, Catfish, Goldfish, Trench, Eel	Roberts (2002)
5	August 3, 2003	August 3, 2003	Boat Electrofishing	Shore fished: 1138 m	Perch, Rudd, Catfish, Goldfish, Trench, Eel	Hicks (unpublished)
6	October 26, 2005	October 26, 2005	Boat Electrofishing	Shore fished: 684 m	Perch, Rudd, Catfish, Goldfish, Trench, Eel	Hicks (unpublished)
7	March 16, 2006	March 16, 2006	Boat Electrofishing	Shore fished: 524 m	Perch, Rudd, Catfish, Goldfish, Trench, Eel	Hicks (unpublished)
8	September 4, 2008	September 4, 2008	Boat Electrofishing	Shore fished: 597 m	Perch, Catfish, Trench, Eel, Common bullies	Hicks (unpublished)
9	January 9, 2011	January 9, 2012	Boat Electrofishing	Shore fished: 2855 m	Perch, Catfish, Trench, Rudd, Eel	Hicks (unpublished)

Gill nets are a highly effective method of fish capture for larger bodied, pelagic species, but are selective on species, fish activity, and size (Erős et al., 2009). Fish are caught by swimming partway through the net and having the net slip behind the opercula (gill covers) or by becoming entangled in the net by spines or fins.

To minimise size selectively of gill nets, all surveys have used a range of mesh sizes (Table 2.3). The use of more than one mesh sizes gives a greater range of fish length, and results are a more accurate estimate of population. Net lengths and mesh sizes have varied over time but in all surveys gill nets were set overnight with an average set time between 16-24 h (Table2.3).

Fyke nets were used to capture fish in the shallow water around the margins of water ways and are selective towards benthic cover seeking, and mobile species (Lapointe et al., 2006). Fyke nets used consisted of three interconnected funnels leading into a closed chamber at the end. The mouth of the first tunnel was connected to a wing supported by float line. The wing was anchored to the lake bank by a stake, with the fyke net perpendicular to the lake shore. The wing acts as a barrier to fish moment, guiding fish into the trap. Mesh size varied between surveys and is unspecified by Wise (1990).

Sampling by boat electrofishing was completed using the University of Waikato 4.5-m long electro-fishing boat. The boat has a rigid aluminium pontoon hull with a 2-m beam, and was fitted with a 6-kilowatt Honda-powered custom-wound generator and a 5-kilowatt gas-powered pulsator (Smith-Root, Inc., model 5.0 GPP); two anode poles created the fishing field at the bow. The two adjustable anode arrays each had l-m long stainless steel rat tails that dangled in the water, and the boat hull itself acted as the cathode (Hicks et al., 2006.) Distance fished was calculated using a Garmin GPSMAP 60CSx global positioning system. Fishing methods have altered over time as research refined the fishing technique (Table 2.3). Fishing effort 2003 to 2006 involved a signal timed shot where a continuous length of shore was fished. 2008 and 2012 involved a series of short timed shots of ten minutes at different shore locations throughout the lake.

Table 2.3: Survey sampling effort; A. gill net fishing effort; B fyke net effort; C. boat electric fishing effort.

A. Gill net fishing effort.

Fish Survey	**Fishing Year**		**Effort**			**Sample Location**	**Net Size**	
	Start	Finish	Set time average (h)	Period	Total Hours		Mesh Size (cm)	Combined Net Length (m)
1a.	July 1976	April 1977	24	Quarterly	-	Northern Basin	3.8/5.2/5.7/6.7/10.8/13.3	429
1b.	July 1977	July 1978	24	Quarterly	-	Northern Basin	3.8/5.2/5.7/6.7/10.8/13.3	429
2	March 1989	February 1990	24	Monthly	288	One set in each basin	2.5/5.7/8.9	90
3	December 1993	February 1994	15.6	Weekly	133.6	8 Shore, 3 Centre	2.5/3.8/5.6/8.4/10.6	120
4	February 2001	May 2001	22	Weekly	154	6 Shore, 1 Centre	2.5/3.8/5.6/8.4/10.6	120

B. Fyke Net Fishing Effort

Fish Survey	**Fishing Year**		**Effort**				**Sample Location**	**Net Size**	
	Start	Finish	Set time average (h)	Total Net Hours	Period	no. of nets per site		Mesh Size (cm)	Wing Length (m)
2	March 1989	February 1990	24	864	monthly	3	3 shore		3
3	December 1993	February 1994	16.2	562	weekly	3	8 Shore, 3 Centre	2.5	4
4	February 2001	May 2001	22	1716	monthly	6	6 Shore, 1 Centre	2	3

C. Boat electric fishing effort

Fish Survey	**Fishing Date**	**Effort**			**Area (m^2)**	**Fishing Location**
		Total time (min)	Period	Length Fished (m)		
5	August 3, 2003	65	1 x 65 shot	1138	4552	Shore
6	October 26, 2005	-	-	684	2736	Shore
7	March 16, 2006	39	-	524	2096	Shore
8	September 4, 2008	30	3 x 10 min shots	597	2388	Shore
9	January 9, 2011	100	10x 10 min shots	2855	11420	Shore

1.10.1 Sampling sites

Sample sites differed between all surveys except Kane and Roberts. Graynoth intensively sampled the northern basin (Figure 2.1) with series of gill nets and trap nets. Wise (1990) sampled five sites, two with gill nets in the northern and southern basins (sites 2a and 2b) and three shore sites with fyke nets (sites 2c, 2d and 2e, Figure 2.1). Kane (1995) and Roberts (2002) sampled the same sites (3#, Figure 2.1) with fyke and gill nets at every location. Roberts (2002) Data from sites 3a and 3h have been excluded as these sites were extensively sampled as part of an enclosure removal programme. Boat electrofishing sampled shore locations only, varying in location and length depending on methods used.

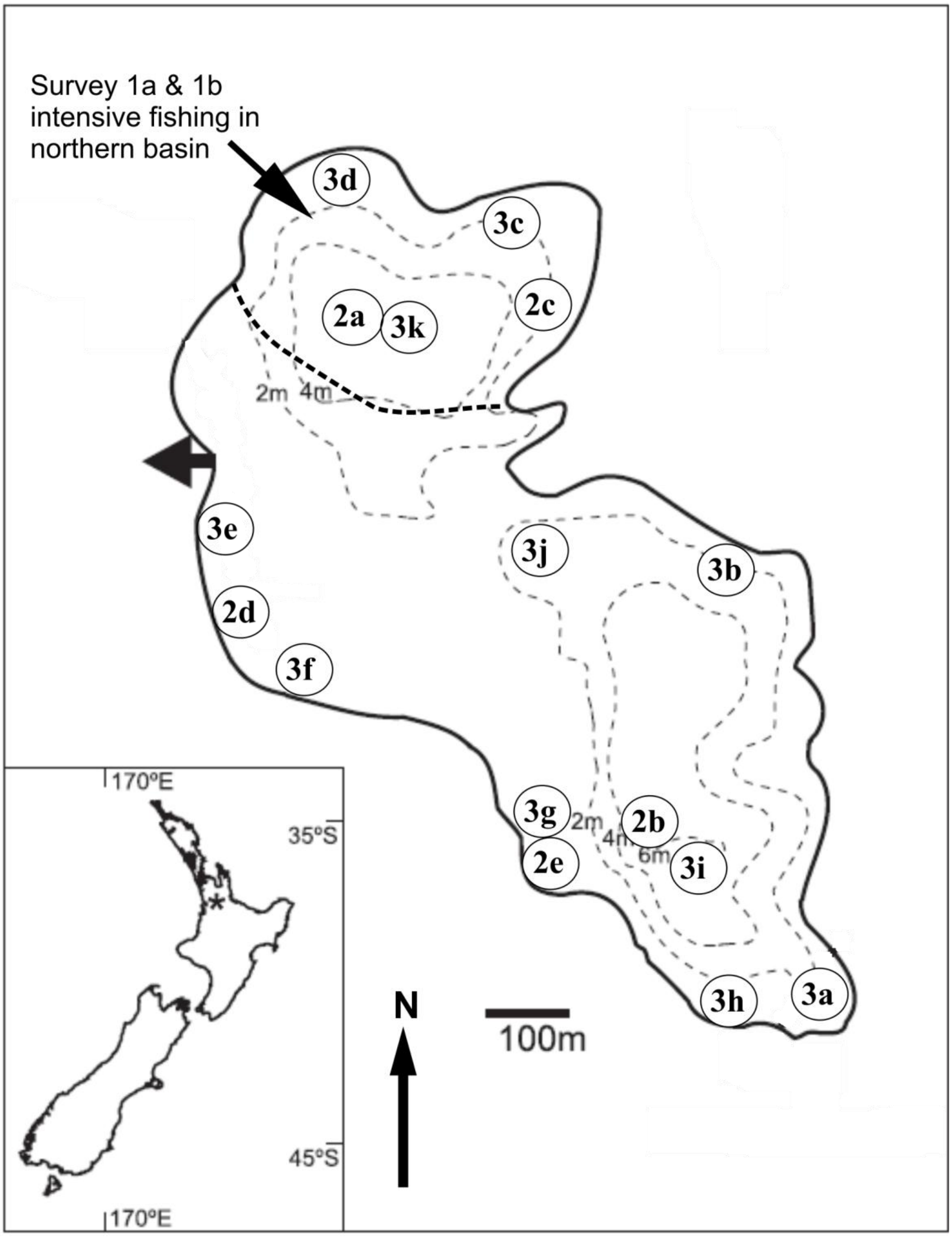

Figure 2.1: Sample locations for Graynoth (northern basin), Wise (sites 2#), Kane and Roberts (sites 3#).

1.10.2 Analysis

1.10.2.1 Catch per unit effort

Catch per unit effort (CPUE) is used to compare surveys, this has been given as catch per set. Fish capture has shown to be greatest during the time of dawn and dusk (Prchalová et al., 2010). It is expected that once a set has been set overnight, an increase in time dose not result in a further relative increase in the catch. To reduce the effect of deferring set times between surveys, CPUE per set has been calculated.

1.10.2.2 Boat electrofishing quantitative bioestimates

Quantitative biomass estimates for boat electrofishing where calculated by correcting signal pass boat electrofishing fish captures by Equation 2.1. Equation 2.1 represents the relationship between the population estimate from Zippin method (Y) of multiple pass removal and the number of fish caught in a first removal (X) (Hicks et al., 2006). Corrected fish captures where then used to correct mean fish density and mean fish biomass.

Equation 2.1: Boat electrofishing signal pass population correction (Hicks et al., 2006)

$$Y = 1.55X^{1.23}$$

1.10.2.3 Fish Condition

Fish condition has been analysed using the linear equation of the natural log-log length and weight relationship (Equation 2.2). Where weight (Y), length (X), slope (b), and Y intercept (a)

Equation 2.2: Length-weight relationship

$$\ln Y = \ln a + bX$$

Length-weight relationships between studies where analysed for significance using analysis of covariance (ANCOVA). Analysis was run with “study year” and "length" as the covariate, to test significant of slope (Equation 2.3). If this was not significant then a separate ANCOVA was run with a single factor "study year" tested for significance separately (Equation 2.3).

Equation 2.3: Analysis of covariance with the covariate length and study year

$$lnW = lnL + study\ year + lnL \times study\ year$$

Equation 2.4: Analysis of covariance with the single factor study year

$$lnW = lnL + study\ year$$

1.11 Results

Total catch has varied between scientific surveys mainly due to differences in methods and effort (Table 2.4). The most notable change in fish assemblage was the introduction of new species with rudd introduced between 1978 and 1989, and the introduction of tench in 1990 shortly after Wise (1990) survey (2) finished. No goldfish have been caught since 2006 and a reduction in fish caught since 2003.

Table 2.4: Total fish caught from gill netting, trap netting, fyke netting and boat electrofishing in fish surveys in Lake Rotoroa between July 1976 and January 2012

Fish survey	Fishing year		Perch	Rudd	Tench	Goldfish	Catfish	Shortfin eel	Common bullies	Total
	Start	Finish								
1a*	July 1976	April 1977	625	0	0	16	0	50	**0**	691
1b*	July 1977	July 1978	999	0	0	7	15	89	0	1110
2**	March 1989	February 1990	114	280	0	23	162	-	0	579
3	December 1993	February 1994	485	305	57	24	78	26	0	975
4	February 2001	May 2001	693	223	80	11	2000	85	0	3092
5	August 3, 2003	August 3, 2003	20	10	10	2	23	16	0	81
6	October 26, 2005	October 26, 2005	21	0	10	15	26	19	0	91
7	March 16, 2006	March 16, 2006	11	1	3	4	6	56	0	81
8	September 4, 2008	September 4, 2008	14	0	1	0	9	15	1	40
9	January 9, 2012	January 9, 2012	57	2	8	0	9	19	0	95

* Graynoth used unspecified trap nets, eel species not specified. ** Wise (1990) - eel captures not recorded.

1.11.1 Gill netting

Gill net total catch relates to effort, with the significant effort by Graynoth's showing in a relatively large catch of perch. The introduction of catfish occurred around 1976-1977 with this species first caught July 1977. CPUE shows a trend of increased fish abundance from survey 1 Graynoth, to survey 4 Roberts, excluding perch in 1990 and goldfish in 2001 (Table 2.5). This increased abundance was attributed to increasing catch of perch and catfish, and the addition of rudd and tench to the fish population.

Table 2.5: Fish caught in Lake Rotoroa from 1976 to 2001. A. gill net total catch. B. gill net catch rate per set per 100 m.

Fish survey	Fishing year		Perch	Rudd	Tench	Goldfish	Catfish	Total
	Start	Finish						
A. Gill net total catch								
1a	July 1976	April 1977	347	0	0	16	0	363
1b	July 1977	July 1978	334	0	0	3	5	342
2*	March 1989	February 1990	89	274	0	-	13	376
3	December 1993	February 1994	436	259	36	23	52	806
4	March 2001	May 2001	563	190	40	6	48	847
B. Gill net catch rate (fish 100 m net $^{-1}$ set^{-1})								
1a*	July 1976	April 1977	0.71	0.00	0.00	0.03	0.00	0.75
1b*	July 1977	July 1978	0.57	0.00	0.00	0.01	0.01	0.58
2	March 1989	February 1990	7.88	11.09	0.00	-	1.04	20.00
3	December 1993	February 1994	36.33	19.62	3.52	2.13	4.81	66.41
4	March 2001	May 2001	86.55	28.33	5.00	0.71	8.33	128.93

* Wise (1990) net type not given for goldfish capture.

1.11.2 Fyke netting

Fyke net total catch shows variability in relation to effort and species, goldfish catch was continuously low. Graynoth caught a large amount of perch using unspecified trap nets, these were set mid-lake rather than the shore-dominated sets in surveys 2 to 4. All species apart from goldfish showed increase in relative abundance. Catfish relative abundance increase was the most significant increasing dramatically between 1994 and 2001.

Table 2.6: Fish caught in Lake Rotoroa from 1976 to 2001. A**.** fyke nets total catch, B. fyke net catch rate.

Fish survey	Fishing year		Perch	Rudd	Tench	Goldfish	Catfish	Shortfin eel	Total
	Start	Finish							
A. Fyke nets total catch									
1a*	July 1976	April 1977	278	0	0	0	0	50	328
1b*	July 1977	July 1978	665	0	0	4	10	89	768
2**	March 1989	February 1990	25	6	0	-	149	-	180
3	December 1993	February 1994	49	46	21	1	26	26	169
4	February 2001	May 2001	130	33	40	5	1952	85	2245
B. Fyke net catch rate (fish net^{-1} set^{-1})									
1a*	July 1976	April 1977	1.75	0.00	0.00	0.00	0.00	0.31	2.06
1b*	July 1977	July 1978	2.79	0.00	0.00	0.02	0.04	0.37	3.22
2	March 1989	February 1990	0.68	0.28	0.00	-	4.13	-	5.10
3	December 1993	February 1994	1.63	1.92	1.40	0.04	0.96	2.53	8.48
4	February 2001	May 2001	1.43	0.36	0.44	0.05	21.45	0.93	24.67

* Graynoth used unspecified trap nets, eel species not specified. ** Wise (1990) net type not given for goldfish capture.

1.11.3 Boat electric fishing

Water conductivity during boat electrofishing ranged from 106 µS cm^{-1} to 113 µS cm^{-1} specific conductivity with water temperature between 11 °C and 20 °C. Perch, catfish and shortfin eel were the most numerous species caught (Table 2.7). Goldfish have not been caught since 2006, and rudd capture rates have also decreased over the same time period. Tench had a high biomass when compared with their density due to the large size of this species (mean weight 968 g).

Table 2.7: Fish caught in Lake Rotoroa from August 2003 - January 2012. A. boat electrofishing total catch. B. boat electrofishing mean fish density. C. boat electrofishing mean fish biomass.

Fish survey	Fishing year	Perch	Rudd	Tench	Goldfish	Catfish	Shortfin eel	Total
A. Electrofishing total catch								
5	August 3, 2003	20	10	10	2	23	16	81
6	October 26, 2005	21	0	10	15	26	19	91
7	March 16, 2006	11	1	3	4	6	56	81
8	September 4, 2008	14	0	1	0	9	15	39
9	January 9, 2012	57	2	8	0	9	19	95
B. Electrofishing mean fish density (fish 100 m^{-2})								
5	August 3, 2003	0.64	0.32	0.32	0.06	0.73	0.51	2.58
6	October 26, 2005	0.77	0.00	0.37	0.55	0.95	0.69	3.33
7	March 16, 2006	0.40	0.04	0.11	0.15	0.22	2.05	2.96
8	September 4, 2008	0.55	0.00	0.04	0.00	0.56	0.67	1.82
9	January 9, 2012	0.50	0.01	0.06	0.00	0.09	0.17	0.83
C. Electrofishing mean fish biomass (g m^{-2})								
5	August 3, 2003	0.97	0.65	1.90	0.14	0.96	1.40	5.2
6	October 26, 2005	1.10	0.00	2.50	1.40	2.00	1.80	8.8
7	March 16, 2006	0.90	0.00	0.90	0.00	0.70	7.00	9.5
9	January 9, 2012	0.67	0.01	0.80	0.00	0.31	0.45	2.24

1.11.3.1 Quantitative estimates of boat electrofishing fish abundance

By adjusting fish captures in Table 2.7 with Equation 2.1, perch, catfish, and shortfin eel have the highest density in Lake Rotoroa (Table 2.8). The logarithmic equation 2.1 results in a larger increase where higher numbers are caught. Corrected the biomass now shows tench biomass lower in relation to the biomass of perch, catfish and shortfin eels, when compared with uncorrected fish biomass as tench were less numerous then the other species (Table 2.7).

Table 2.8: Lake Rotoroa boat electrofishing August 2003 - January 2012. Values adjusted using equation 2.1, A. corrected boat electrofishing fish catch, B. corrected boat electrofishing mean fish density, C. corrected boat electrofishing mean fish biomass.

Fish survey	Fishing year	Perch	Rudd	Tench	Goldfish	Catfish	Shortfin eel	Total
A. Corrected boat electrofishing fish capture								
5	August 3, 2003	62	26	26	4	73	47	238.3
6	October 26, 2005	66	9	26	43	85	58	287.0
7	March 16, 2006	30	2	6	9	14	219	278.8
8	September 4, 2008	40	0	2	0	23	43	107.8
9	January 9, 2012	224	4	20	0	23	58	328.6
B. Corrected mean fish density (fish 100 m^2)								
5	August 3, 2003	1.98	0.84	0.84	0.12	2.35	1.50	7.64
6	October 26, 2005	2.40	0.31	0.96	1.58	3.12	2.12	10.49
7	March 16, 2006	2.91	0.15	0.59	0.84	1.38	21.56	27.44
8	September 4, 2008	1.67	0.00	0.06	0.00	0.97	1.82	4.52
9	January 9, 2012	1.96	0.03	0.18	0.00	0.20	0.51	2.88
C. Corrected mean fish biomass (g m^2)								
5	August 3, 2003	3.02	1.73	5.00	0.26	4.61	4.16	18.78
6	October 26, 2005	3.41	1.45	6.69	3.92	6.43	5.63	27.54
7	March 16, 2006	5.18	0.01	3.87	0.06	3.34	56.54	69.00
9	January 9, 2012	2.61	0.04	2.18	0.00	0.69	1.34	6.85

1.11.1 Relative fish abundance

Relative abundance of fish species in Lake Rotoroa has fluctuated depending on species present, population growth of a single species, and methods of capture (Table 2.9). Perch and catfish are the most abundant species in Lake Rotoroa in most years except when there is an extreme population growth for a short period of time. Before the introduction of rudd and catfish, perch relative abundance dominated fish captures, Graynoth caught 90% perch in two years fishing. A reminder needs to be made that effort was dominated by gill nets and trap nets not targeting shortfin eels, the dominate native species. Rudd once introduced underwent rapid population increase and had high abundance in 1989 to 1990. Since this time the population has gradually deceased with low abundance since 2005. Perch have had relatively stable population with constant 20-35 percentage of total catch apart from 2006. Catfish also have had a relatively consistent abundance since 1989-1990 with a population boom in 2001. Relative abundance in 2006 is dominated by shortfin eels due to sampling of eel specific habitat at the southern end of the lake, this likely results in a under estimate of relative abundance of other species. Goldfish and tench show a stable low relative abundance peaking 2005 before dropping after this survey.

Table 2.9: Fish caught as a percentage of total fish caught from gill netting, trap netting, fyke netting and boat electrofishing in fish surveys in Lake Rotoroa between July 1976 and January 2012

Fish Survey	Fishing Year		Perch (%)	Rudd (%)	Tench (%)	Goldfish (%)	Catfish (%)	Shortfin eel (%)	Common bullies (%)	Total catch
	Start	Finish								
1a*	July 1976	April 1977	90.4	0.0	0.0	2.3	0.0	7.2	0.0	691
1b*	July 1977	July 1978	90.0	0.0	0.0	0.6	1.4	8.0	0.0	1110
2**	March 1989	February 1990	19.7	48.4	0.0	4.0	28.0	-	0.0	579
3	December 1993	February 1994	49.7	31.3	5.8	2.5	8.0	2.7	0.0	975
4	February 2001	May 2001	22.4	7.2	2.6	0.4	64.7	2.7	0.0	3092
5	August 3, 2003	August 3, 2003	24.7	12.3	12.3	2.5	28.4	19.8	0.0	81
6	October 26, 2005	October 26, 2005	23.1	0.0	11.0	16.5	28.6	20.9	0.0	91
7	March 16, 2006	March 16, 2006	13.6	1.2	3.7	4.9	7.4	69.1	0.0	81
8	September 4, 2008	September 4, 2008	35.0	0.0	2.5	0.0	22.5	37.5	2.5	40
9	January 9, 2012	January 9, 2012	60.0	2.1	8.4	0.0	9.5	20.0	0.0	95

1.11.2 Length frequency distribution

Juvenile fish with a fork length <70 mm, were not caught, which is expected due the size selectivity of gill nets and the small distance travelled by juvenile fish. Perch have showed a reduction in fork length >300 mm since 2001 (Figure 2.2) although boat electric fishing may target a select size range as all fish captured were within a fork length between 190 mm and 280 mm. Catfish show a trend of increasing size (Figure 2.3), which might be representative of increasing age, though no aging of fish was done confirm this. Tench showed a range of size classes (Figure 2.3) with an absence of fish with a fork length <300 mm in 1995 and 2012. Length frequency distribution of shortfin eels (Figure 2.6) showed a large decrease in length since the original survey by Graynoth, this could be due to commercial fishing or that Graynoth (unpublished) did not specify eel species and included a mix of shortfin and longfin eels. Roberts (2002) length frequency analysis included data from an intensively fish enclosure so may not be comparable to other studies.

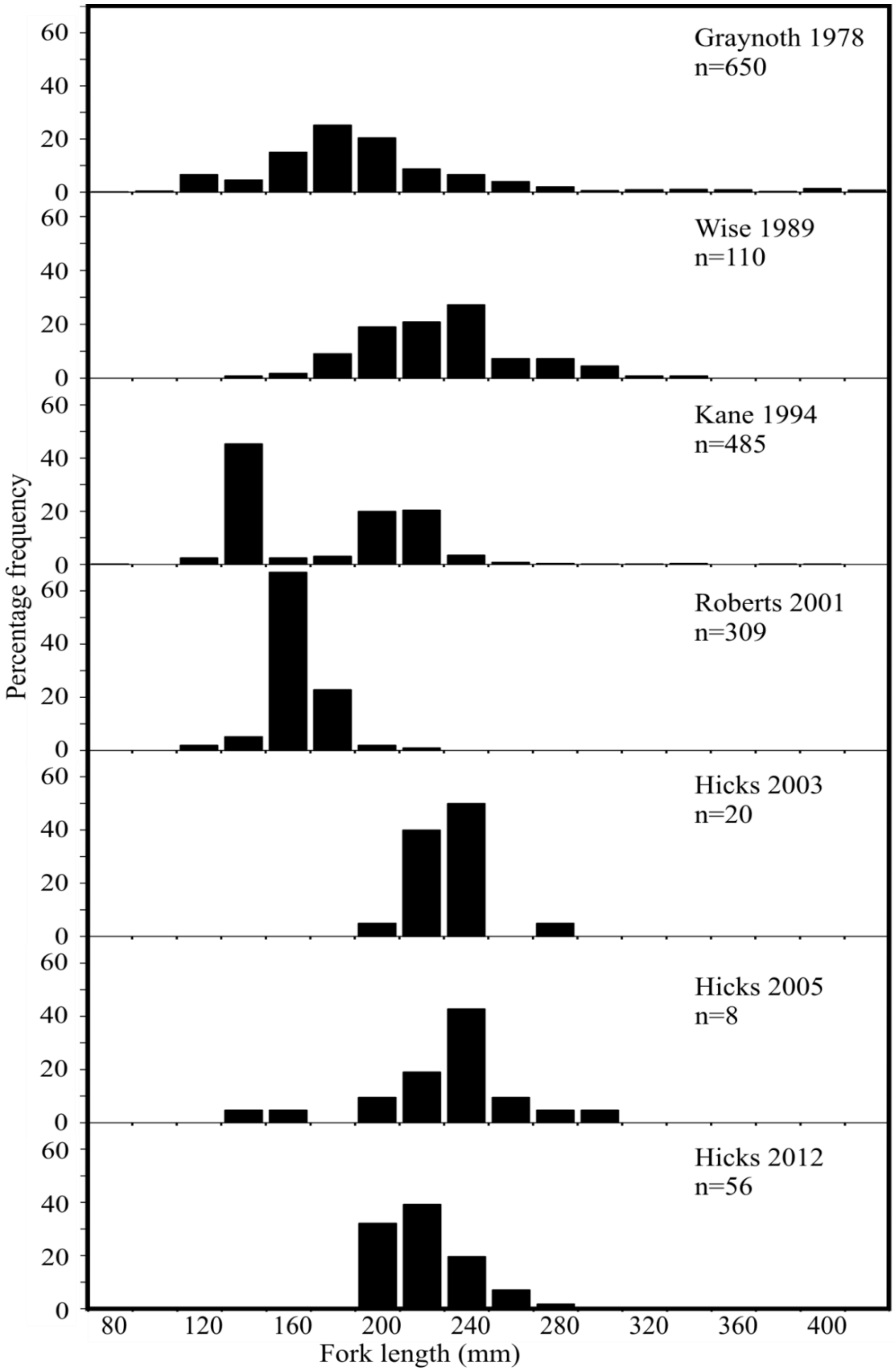

Figure 2.2: Length frequency distribution of perch in Lake Rotoroa 1976-2012. 1976-2001 gill and fyke nets, 2003-2012 boat electrofishing

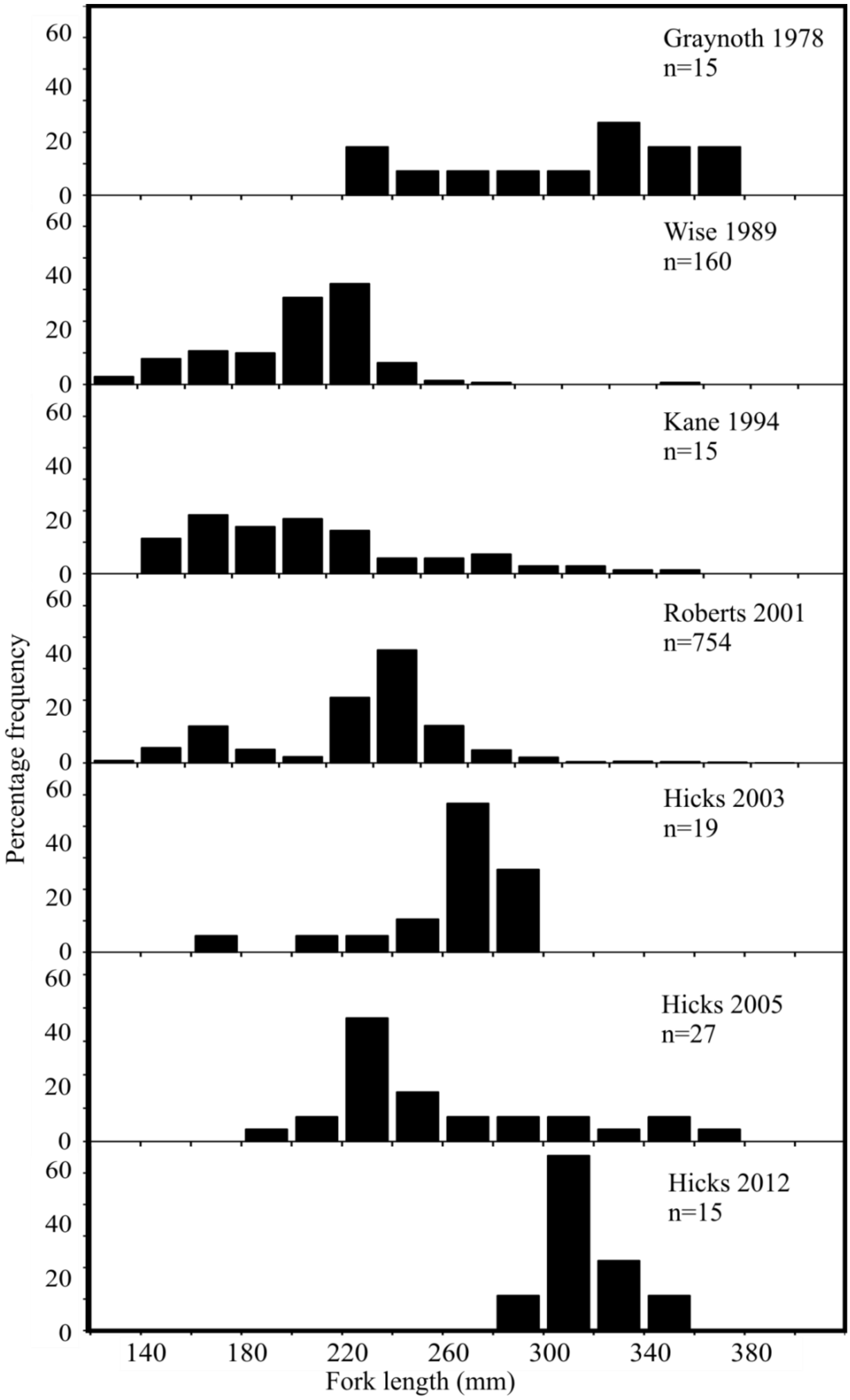

Figure 2.3: Length frequency distribution of catfish in Lake Rotoroa 1978-2012. 1977-2001 gill and fyke nets, 2003-2012 boat electrofishing.

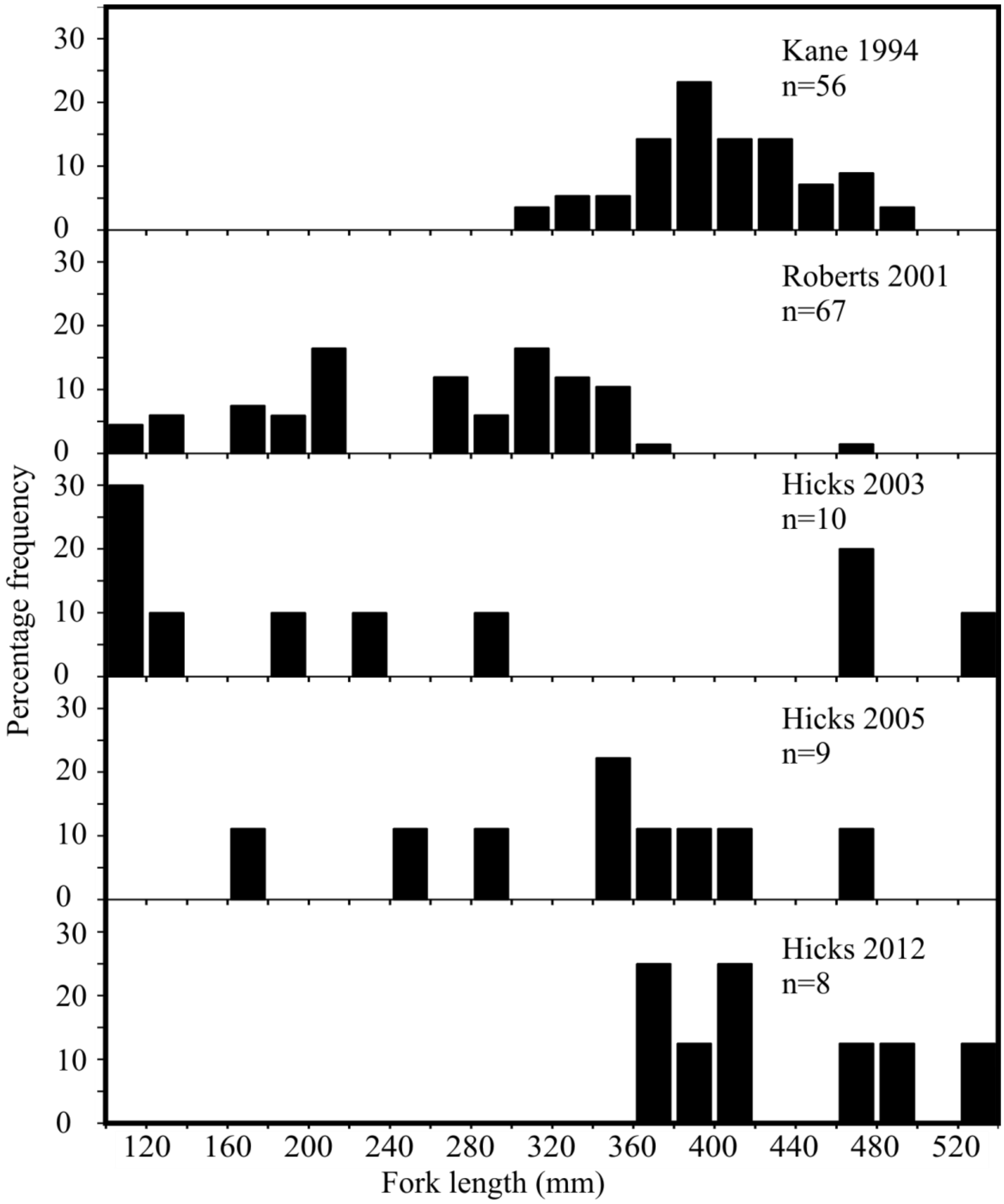

Figure 2.4: Length frequency distribution of tench in Lake Rotoroa 1994-2012. 1994-2001 gill and fyke nets, 2003-2012 boat electrofishing

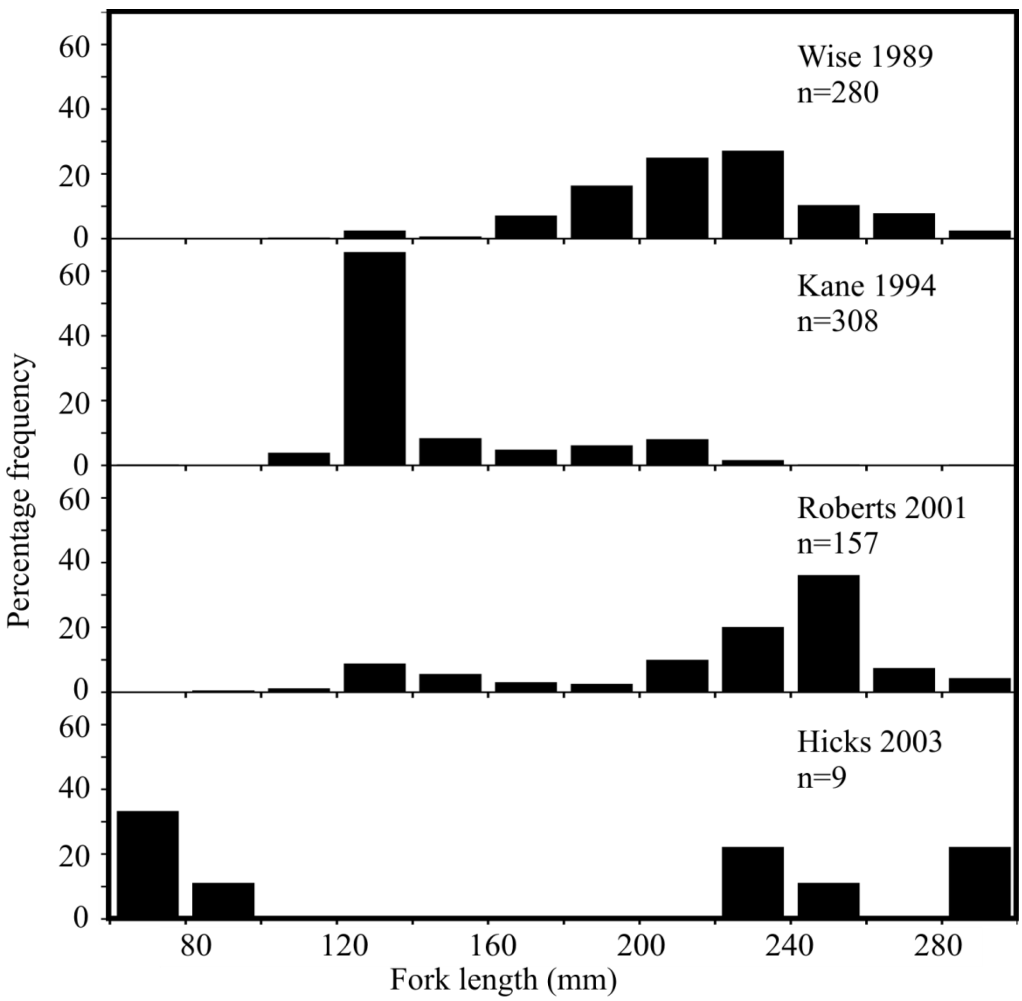

Figure 2.5: Length frequency distribution of rudd in Lake Rotoroa 1989-2003. 1989-2001 gill and fyke nets, 2003 boat electrofishing.

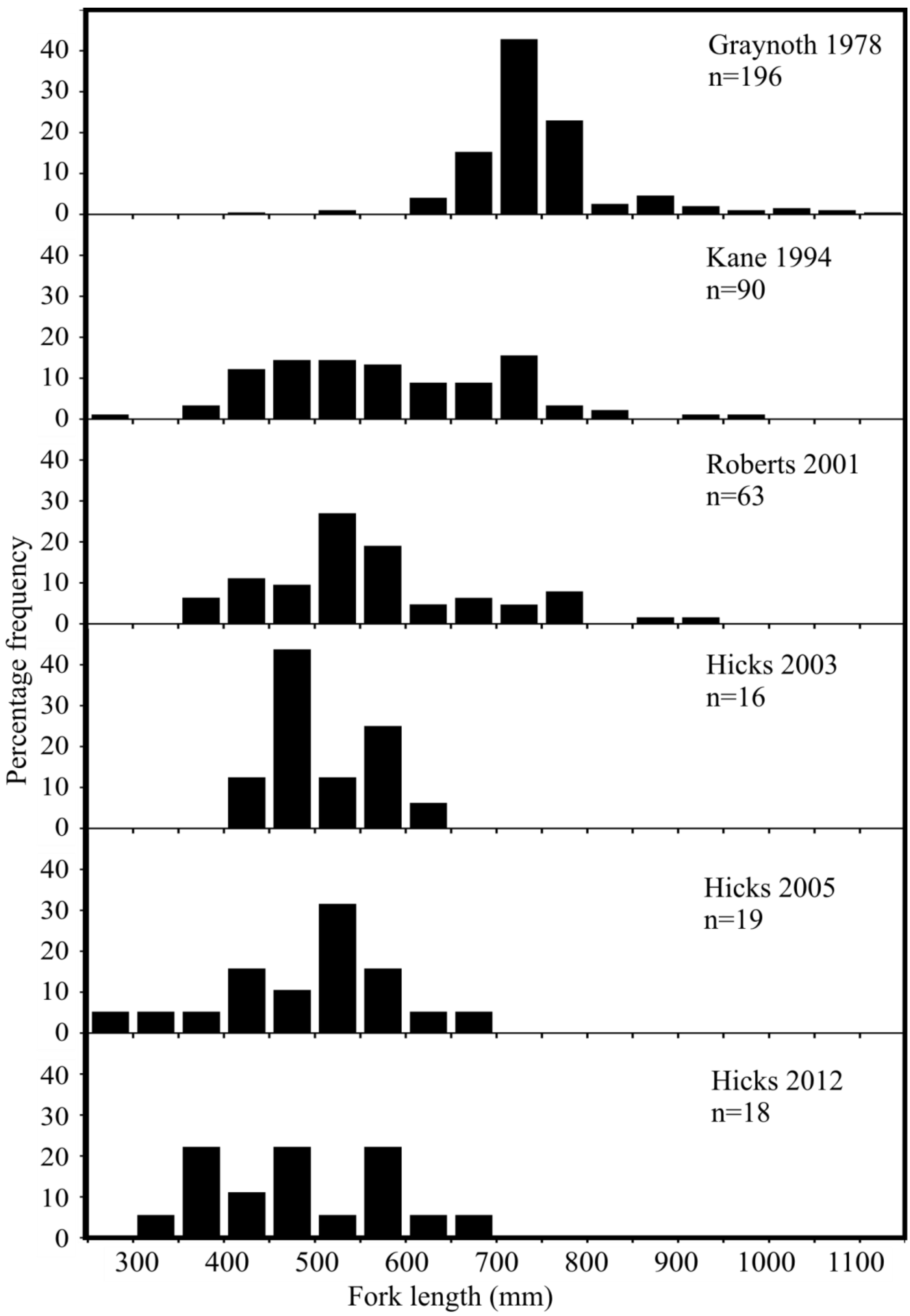

Figure 2.6: Length frequency distribution of shortfin eel in Lake Rotoroa 1978-2012. 1978 - 2001 gill and fyke nets, 2003 - 2012 boat electrofishing. Graynoth (unpublished) did not specify eel species.

1.11.3 Length-weight relationships

Length weight relationship of fish species is shown in Table 2.9. Weight data was only available for surveys 2-4, 6, and 9. Analysis of rudd was limited, as surveys 6 and 9 are excluded in from the analysis because of low catches rates. Caution is needed when comparing tench length-weight regressions as survey 9 had a different size range compared to the previous studies. No length frequency and length weight analysis has been completed on goldfish due to relatively low catch numbers.

ANCOVA analysis confirms that catfish are significantly different ($P < 0.05$) in study and length-weight relationship. Indicating a decrease in condition with time as slope reduces representing longer but leaner fish. Rudd was non-significant ($P=0.71$) between ANCOVA with the covariates study and length, but were significant ($P<0.05$) between weight and the covariate, study year. This represents a decrease in the condition of rudd. Perch, shortfin eel, and tench were not significant in both the ANCOVA of weight between study year and length, and the ANCOVA between study and weight. P values, $P = 0.685$ and $P = 0.872$ respectively for perch, $P = 0.154$ and $P = 0.844$ respectively for shortfin eel, $P = 0.213$ and $P = 0.355$ respectively for tench. All species met the assumption of homogeneity of slopes.

Table 2.10: Natural log fish length (mm) (X) vs. natural log fish weight (g) (Y) relationship of fish caught in Lake Rotoroa between 1990 and 2012 ($\ln Y = \ln a + bX$).

Fish Survey	n	Slope (*b*)	ln*a*	R^2	Length		
					Min	Mean	Max
Catfish							
2	161	3.29	-12.76	0.96	103	189	346
3	84	3.26	-12.68	0.93	124	197	344
4	736	3.00	-11.35	0.96	35	212	389
6	27	2.93	-10.79	0.90	173	237	334
9	9	2.82	-10.23	0.89	270	293	330
Perch							
2	113	2.77	-9.96	0.86	115	220	330
3	530	3.00	-11.33	0.89	75	167	388
4	305	2.77	-10.01	0.97	100	213	264
6	21	3	-10.99	0.93	134	221	285
9	57	3.01	-11.32	0.70	186	214	282
Rudd							
2	280	3.19	-11.83	0.97	107	218	296
3	308	3.28	-12.43	0.89	78	146	281
4	158	3.17	-11.77	0.99	115	152	252
Shortfin eel							
3	80	3.30	-15.05	0.92	257	560	978
4	63	2.94	-12.16	0.96	381	534	910
6	19	3.24	-14.65	0.87	276	491	665
9	18	3.29	-14.94	0.92	315	487	690
Tench							
3	59	2.93	-10.76	0.74	310	401	485
6	10	2.94	-10.75	0.99	250	360	470
9	7	2.12	-5.76	0.96	365	436	540

1.11.4 Net selectivity

A comparison between net selectivity of gill nets and trap nets 1976-1978 is shown in Figure 2.2. Trap nets caught a higher percentage of young perch (<150 mm), whereas gill net favoured fish larger than 150 mm while capturing a greater number of perch over 300 mm. Trap nets show perch population with a bimodal distribution, clearly showing multiple age classes, whereas gill nets showed a normal distribution under-representing fish smaller than 140 mm.

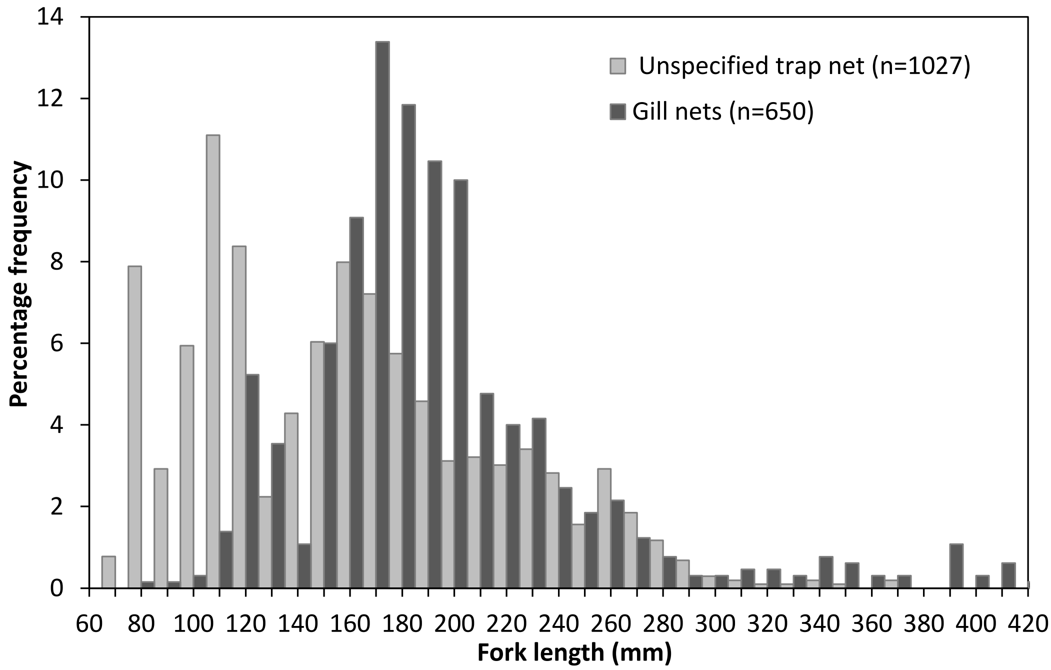

Figure 2.7: Net selectivity compassion between unspecified trap nets and gill net (mesh size 2.5, 3.8, 5.2, 5.7, 6.7, 10.8, 13.3 cm) in Lake Rotoroa July 1976 – April 1978 (data from Graynoth unpublished).

1.12 Discussion

1.12.1 Comparative changes in relative abundance

Gill net CPUE (Table 2.5) shows that perch have the highest relative abundance in all years except for 1990, with increasing abundance through time. An increasing abundance of rudd, catfish and tench is also evident up to 2006 (Table 2.7). Capture rates for catfish show rapid population growth between 1994 and 2001 with the CPUE increasing four fold. The low CPUE for Kane's (1995) fyke netting is expected to be due to the low effort put into fyke netting (3 nets per set), resulting in an skewed representation of relative abundance. In 2006, shortfin eel density was extremely high, specific targeting of this species or sampling of a single habitat type at the southern end of the lake is the likely cause. Hicks (pers.

com., 2012) advised that that shortfin eel catch rates could vary significantly depending on habitat sampled and cover present. Rudd has decreased significantly when comparing gill netting and electrofishing. In 2001 rudd consisted of 21% of the total catch, this had dropped to 10% in 2003 and 2% in 2012 with no fish caught on two occasions 2005 and 2008. The reasons for this decline are unknown. Tench relative abundance between 1994 and 2001 appeared to be increasing associated with higher CPUE. Comparatively boat electrofishing shows a stable population over the 2003 to 2005 period then a decrease in density to 2012.

1.12.2 Capture method selectivity

Different sampling methods capture different fish species or proportion of the assemblage. Passive methods capture mobile species while active methods capture more inactive species (Weaver et al., 1993). The use of multiple sampling gear provides a better representation of fish assemblage composition and size structure (Ruetz et al., 2007). Graynoth (unpublished), Wise (1990), Kane (1995), and Roberts (2002) all used at least two methods gill nets and trap or fyke nets. Beach-seine, purse-seine and Gee-minnow traps were also used but proved to be largely unsuccessful with few fish caught (Kane, 1995, Roberts, 2002, Wise, 1990). Daniel & Morgan (2011) found that in a shallow Waikato lakes species capture was highly dependent on net type with baited traps proving successful in capturing goldfish, whereas fyke nets were successful at capturing catfish and shortfin eels. The use of multiple methods does not guarantee sampling of entire fish populations as methods often do not balance out the selectivity of other gear (Beamesderfer and Rieman, 1988). Fish condition is also important, with gill-net selectivity favouring fatter fish among short fish, and thinner fish among longer fish (Cren, 1951). Electrofishing is selective to larger bodied more passive species with fishing conditions effecting capture rates, for example water clarity, water temperature conductivity, and the presence of littoral and submerged vegetation reducing fishing successes (Weaver et al., 1993, Reynolds, 1996, Hicks et al., 2006). Electrofishing is predicted to underestimate biomass of certain cover seeking species with capture rates less than 50% especially for catfish and eels due to their benthic cover seeking existents and reaction to the electric field (Hicks, personal communication, 2012). A comparison between fyke netting and boat electrofishing by Ruetz et al. (2007) found that captured species differ between the fyke nets and boat electrofishing with fish captured by electrofishing significantly larger then fish captured by fyke nets but the use of both limits the bias of using one method. The high capture rates of perch in Graynoth's unspecified trap nets are possible due to the mid lake location of the traps set in the northern basin of Lake Rotoroa (Figure 2.1). Fyke nets favour the cover seeking eels and catfish, with capture rates high for these species, when compared with gill nets (Table 2.5).

1.12.3 Native species

Shortfin eel is the most common native fish at present, with only one common bullies caught in 2008, and the last longfin eel caught by Kane (1995). The introduction of exotic species appears to have resulted disruption of the original fish population with the loss of common smelt, and possibly longfin eel from the ecosystem. The loss of common smelt appears to be due to predation by mosquitofish and reduction in water quality. Wakelin (1986) recorded predation of smelt by mosquito fish in Lake Waahi. A reduction in smelt survival and habitat due to a decrease in water clarity could also be a significant factor (Rowe and Taumoepeau, 2004) with the demise of smelt in Lake Waahi was attributed to increase in turbidity (Northcote and Chapman, 1999). The decrease in abundance of longfin eel is uncertain but there are several possible factors that may have contributed. Longfin eels show avoidance of degraded environments (Aldridge and Hicks, 2006, Ryan, 1991). The reduction of water quality may have resulted in possible avoidance of Lake Rotoroa during migration. The piping of the Lake Rotoroa outlet could have resulted in a migration barrier, but this seems unlikely as shortfin eels show recruitment into the lake through size class <350 mm. Common bullies are thought to be still present in the lake, with a bully found in the stomach contents of perch in an unpublished investigation of fish diet. Graynoth (unpublished report) suggested that catch rates of common bullies are low due to their secretive nature and that they are not prone to capture in the methods used.

Length frequency analysis of shortfin eels shows a change in size distribution with the removal or loss of large eels over 1000 mm (Figure 2.6). This can be explained by removal of larger species by recreational or commercial eel fishing since 1978 or migration to breeding grounds. Method selectivity may also be a factor thorough Graynoth's trap nets favouring larger individuals, with fyke nets and boat electrofishing favouring smaller size classes. The absence of small eels can be explained by sampling methods with fyke nets not adequately sampling the smallest size groups (Jellyman and Chisnall, 1999). Analysis of length weight relationship confirmed that there has been no change in eel population between

1994 and 2012. The low sample size for boat electrofishing would have limited the statistical comparison.

1.12.4 Exotic fish

1.12.4.1 Perch

CPUE analysis (Table 2.5 and Table 2.6) and electrofishing mean density's (Table 2.7) show perch to be consistently the most abundant fish in Lake Rotoroa. Expect during a period where there is an explosion of a single fish species, for example catfish in 2001 (Table 2.6). Perch length frequency (Figure 2.2) is typical of a population in New Zealand waters, with a dense population of small fish (<250 mm) due to the lack of predators (McDowall, 1990, Jellyman, 1980). The stability of the population is expressed in length-weight analysis, with no significant difference between studies. Roberts (2002) found that the low R^2 value expressed in the length-weight relationship (Table 2.9), is due to isolated populations in the lake with limited migration between them. Because of the small size of perch, there is little interest this species as game fish and the virtual absence of predator fish species means there is likely to be little change in population structure (Rowe, 1986).

1.12.4.2 Catfish

Since their introduction between 1976 and 1977 catfish population has increased dramatically with the highest CPUE of 21.45 fish net^{-1} set^{-1} in 2001(Table 2.6). Comparatively this dominance continued with high mean fish density through till 2008 (Table 2.7). Length frequency distribution of catfish show that there was a lack of juveniles (< 130 mm) (Sinnott and Ringler, 1987) captured. This could be a result of method selectively or due to a stable population of large individuals. A stable population can be explained by Johnson (1994) theory, that in unexploited, large populations, biomass is maintained indefinitely through time by the gradual and ordered replacement of individuals. Significant difference in length-weight relationship between species (Table 2.9) can be concluded that the population either still in progressing towards a stable population of large individuals with little recruitment occurring or that competition for food may resulting in reduced condition.

1.12.4.3 Rudd

Rudd showed high relative abundances through to 2003, since then mean fish density has been below 0.05 fish 100 m^{-2} with no fish caught in 2005 and 2008 (Table 2.7). Rudd were thought to be a factor in the collapse of submerged macrophyte beds in between 1988 and 1990, which Wise (1990) had identified as a major food source. After the collapse of macrophytes the diet of rudd had altered with the majority of small rudd (110 mm – 127 mm) feeding exclusively on chironomids with large rudd (166 mm – 247 mm) feeding on emergent plant material (*Nymphaea* cultivars, *Iris Pseudacorus* and *Baumea articulata*). The cause of declining abundance of rudd is unknown, as macrophytes where still present over large areas of the lake until 2006, with decline in rudd occurring before October 2005 (de Winton et al., 2011). Competition between species is one possibility of rudd’s decline in abundance. Diet analysis by Wise (1990) and Kane (1995) demonstrating that chironomids make up a large proportion of the diet of exotic fish species in Lake Rotoroa. Increases in abundance of other species over this time period may represent a competitive advantage over rudd.

1.12.4.4 Tench

Tench introduced in 1990 now dominate the lake biomass when compared with mean fish density (Table 2.8), with 31% of the total corrected biomass in 2012, an increase of 30% since 2003. This equates to a corrected total lake biomass of 11,772 kg in 2012. This is attributed to the large size individual fish can reach. ANCOVA results suggest that there has been little change in the length-weight relationship of tench. This could be due to the low sample size from boat electrofishing. The length-weight relationship in fish may change with age, season, nutrition, sexual maturity and species (Ricker, 1975). Tench length frequency distribution (Figure 2.4) shows evidence of reproduction since 1994 with small fish (<150 mm) caught in 2001 and 2003. Kane (1995) suggested that due to cool water temperatures or wind exposure at the spawning sites prevented breeding in Lake Rotoroa. Tench require warm water temperatures above 18 °C for an extended period to allow for eggs and milt ripen generally spawning late spring (Kennedy and Fitzmaurice, 1970, Rowe, 2004). It is likely that the temperature in Lake Rotoroa is above 18 °C long enough to allow spawning. The collapse of macrophyte beds could prevent reproduction of tench, as they require submerged vegetation to spawn (O'Maoileidigh and Bracken, 1989). The lack to juveniles in 1994 and 2012 (Figure2.4) coincides with periods were submerged vegetation had collapsed (de Winton, 1994b, de Winton et al., 2011). Rowe (2004) indicated that the loss of macrophytes could be a factor that limits tench populations in lakes. Sampling by boat electrofishing may result in under estimation of population if survey sites are limited. Perrow et al. (1996) found that tench form aggregations in favoured locations, if these sites where missed it may result in limited sample of the population.

1.12.4.5 Goldfish

No goldfish have been caught since 2006 (Table 2.7) with generally low abundance except for 1993-1994. Competition from other species is anticipated to be the possible cause of this decline and low abundance in goldfish. When comparing goldfish abundance to other lakes in the Hamilton basin and urban areas in New Zealand. Fish density and biomass is lower in Lake Rotoroa then other shallow lakes with comparable species assemblage. Goldfish in Lake Rotokaeo, Lake Ngaroto and Hokowhitu Lagoon mean density varied from 2.32, 0.55, 0.77 fish 100 m^{-2}, respectively, while mean biomass was 0.86, 0.22, and 2.93 g m^{-2}, respectively (Hicks et al., 2009, Hicks and Brijs, 2009, Brijs et al., 2009). A survey of Lake Rotoroa in 2008, 3 months before the Lake Rotokaeo survey had failed to capture any goldfish, while the highest biomass was in 2005 at 1.40 g m^{-2} (Table 2.7). Comparing fish assemblage from the above lakes to Lake Rotoroa, apart from the presence of koi carp (*Cyprinus carpio*) in Lake Ngaroto it has a similar assemblage to Lake Rotoroa with lowest goldfish density and biomass out of the three lakes. When compared Lake Rotokaeo which has a limited species diversity of, goldfish, shortfin eel, and common bully but the highest mean goldfish density and biomass. Limited inter species competition within the lake suggest that low competition and predation allows for high abundance.

1.13 Summary

Perch and catfish were the dominant fish species in Lake Rotoroa, as shown by consistently high CPUE in gill and fyke nets surveys, and more recently by high fish density in boat electrofishing surveys. Both species show relatively stable population structures, with perch dominated by small fish due to the lack of a large predator. Catfish have a stable population structure dominated by large individuals with steady replacement over time. Shortfin eels were the only abundant native species caught between 1994 and 2012, with unpublished diet evidence suggesting common bullies are also present. No Longfin eels were captured in Lake Rotoroa between 1994 and 2012 suggesting the degradation in water quality and the piping of the lake outlet may have resulted in the loss of this species from the ecosystem. Further research on the difference in capture selectivity between different netting techniques and boat electrofishing is needed to allow quantitative comparison between methods, and to enable accurate assessment of changes in fish assemblage.

Chapter 3: Fluctuations in macrophyte abundance and water quality in Lake Rotoroa

1.14 Introduction

Lake Rotoroa water quality has changed significantly over the last 40 years transforming from a oligotrophic to slight mesotrophic lake condition in 1978 (Graynoth unpublished) to a supertrophic lake in 1992 (Burns et al., 1995). The local media labelled the lake as "on a downward spiral to biological death" (Clayton and de Winton, 1994a). The cause of water quality decline have been contributed to the collapse of macrophytes populations between 1988 and 1990, resulting in a release of nutrients and resuspension of sediment as the lake switched from a mesotrophic-eutrophic, clear water state, to a turbid, supertrophic state (de Winton, 1994b). This chapter will also explore relationships between macrophytes, key nutrients, and critical indicators of water quality in Lake Rotoroa between 1992 and 2012.

1.14.1 Trophic level index

Trophic level index (TLI) is a classification system based on key measurable variables used to define the biological condition of a waterbody (Carlson, 1977, Burns et al., 1999). Early trophic level indices developed by Carlson (1977), and Chapra and Dopson (1981), proved to be inadequate for New Zealand's lakes. Carlson's (1977) TLI, based on Secchi depth measurement, was too coarse at the higher trophic levels. While Capra and Dopson's (1981), TLI was too fine, with five levels within the mesotrophic range. Total nitrogen important for nitrogen limited lakes in New Zealand, was also not incorporated into either of these indices (Burns et al., 1999). Burns et al. (1999) developed a TLI based on chlorophyll *a* and incorporating total nitrogen, total phosphorus, and Secchi depth. Chlorophyll *a* concentrations were assigned to trophic level values, originally proposed by Vant, in Davies-Colley et al. (1993). The chlorophyll *a* concentrations 2, 5, and 30 mg m^{-3} where assigned the trophic values 3, 4 and 6, respectively, these were plotted and a regression given, this is then used to normalised chlorophyll *a* concentrations to within the index range 0-7 (Table 3.1). Total nitrogen, total phosphorus, and Secchi depth regressions were then calculated from lake data at a known chlorophyll *a* trophic level, so that the data is normalised. The four variables are then combined equally to give a trophic level between 0 and 7 relating to biological condition of a water body (Table 3.1) (Burns et al., 2000).

Table 3.1: Values of TLI variables that define the boundaries of different trophic levels (Source: Burns et al., 2000)

Lake type	Trophic level index	Chl *a* (mg m^{-3})	Secchi depth (m)	TP (mg m^{-3})	TN (mg m^{-3})
Ultramicrotrophic	0.0 - 1.0	0.13 - 0.33	33 - 25	0.84 - 1.8	16 - 34
Microtrophic	1.0 - 2.0	0.33 - 0.82	25 - 15	1.8 - 4.1	34 - 73
Oligotrophic	2.0 - 3.0	0.82 - 2.0	15 - 7.0	4.1 - 9.0	73 - 157
Mesotrophic	3.0 - 4.0	2.0 - 5.0	7.0 - 2.8	9.0 - 20	157 - 337
Eutrophic	4.0 - 5.0	5.0 - 12	2.8 - 1.1	20 - 43	337 - 725
Supertrophic	5.0 - 6.0	12 - 31	1.1 - 0.4	43 - 96	725 - 1558
Hypertrophic	6.0 - 7.0	> 31	< 0.4	> 96	> 1558

1.14.2 Deseasonalisation of data

Seasonal effects on the environment, results in a predictable weather associated changes in environmental variables, for example, temperature or rainfall. These changes result in a general periodic fluctuation of environmental variables that affect the ability to statistically detect trends in environmental data.

Deseasonalisation allows for the removal of seasonal trends leaving a statistically stationary residual suitable for modelling and statistical analysis. Deseasonalisation does not remove multi-year variation but adjust the time series data for the seasonality experienced within the year.

1.14.3 Chapter objectives

The objectives of the chapter are to:

1. Illustrate fluctuations in macrophyte coverage in Lake Rotoroa since the recovery of macrophytes in 1998
2. Describe changes in Lake Rotoroa trophic level and key nutrients variables between 1992 and 2012.
3. Relate changes in water quality and macrophyte cover of Lake Rotoroa to other similar lakes in the Waikato region.

1.15 Methods

1.15.1 Lake monitoring programme

Data presented in this chapter was collected by NIWA as part of the national water quality network and continued on under contract from Hamilton City Council. Water samples have been taken from the southern basins, which has been sampled a minimum of four times a year since 1992. Samples have been collected and analysed in accordance of methods by Burns et al. (2000).

1.15.2 Analysis

1.15.2.1 Deseasonalising data

Seasonal variation in monitoring data was removed through a deseasonalised processes and trends tested for equivalence using the programme Time Trends (Jowett, 2011). The seasonal pattern was determined by fitting a generalised additive model with seven degrees of freedom, as a smother, to the annual pattern. Residuals are then plotted with the seasonality removed and tested for a trend using a equivalence test (Jowett, 2011). The generalised additive model developed by Hastie and Tibshirani (1990), assumes that the mean of the dependent variable depends on an additive predictor through a non-linear link function.

1.15.2.2 Equivalence testing

Interannual trend in monitoring data was tested using an equivalence test. The advantage of an equivalence test is that it can provide strong evidence for or against an environmentally significant difference or trend. A traditional statistical test of trend or difference is not a test of whether the trend or difference is environmentally important (Jowett, 2011). The null hypothesis is that there is no trend.

1.15.2.3 Trophic level index

Trophic level index (TLI) widely used to calculate changes in the nutrient status of lake. TLI was calculated using methods described by Burns et al. (2000) using total nitrogen, total phosphorus, chlorophyll *a* and Secchi depth. The data was averaged for the year and normalised using a regression equations for each variable so that they are same range as the TLI (Table 3.1)(Burns et al., 2000).

1.16 Results

1.16.1 Submerged macrophyte cover

Submerged macrophyte cover has undergone significant change between the collapse the in 1990 and its present condition (de Winton, 1994a). In 1997 the first significant charophytes growth was recorded in a seasonal vegetation survey (Burns et al., 1998), charophytes continued to expand up to 2005 with 30% total lake bed covered within the 0.5 m to 1.6 m depth range (Figure 3.1) (de Winton et al., 2005). In 2001, *Egeria densa* was first recorded in the lake, since the first macrophyte collapse in 1990. Since this time *Egeria densa* has continued to spread, hand weeding and selective Diquat applications in December 2004 and December 2009, have been applied to manage the population (de Winton et al., 2005, Burns et al., 1998, de Winton et al., 2010). Since 2005 macrophyte coverage has under gone a steady decline (Figure 3.1). In 2011, there were only a few clumps of charophytes and *Egeria densa* present. Strands of filamentous algae *Spirogyra* sp. were also sparse (de Winton et al., 2011).

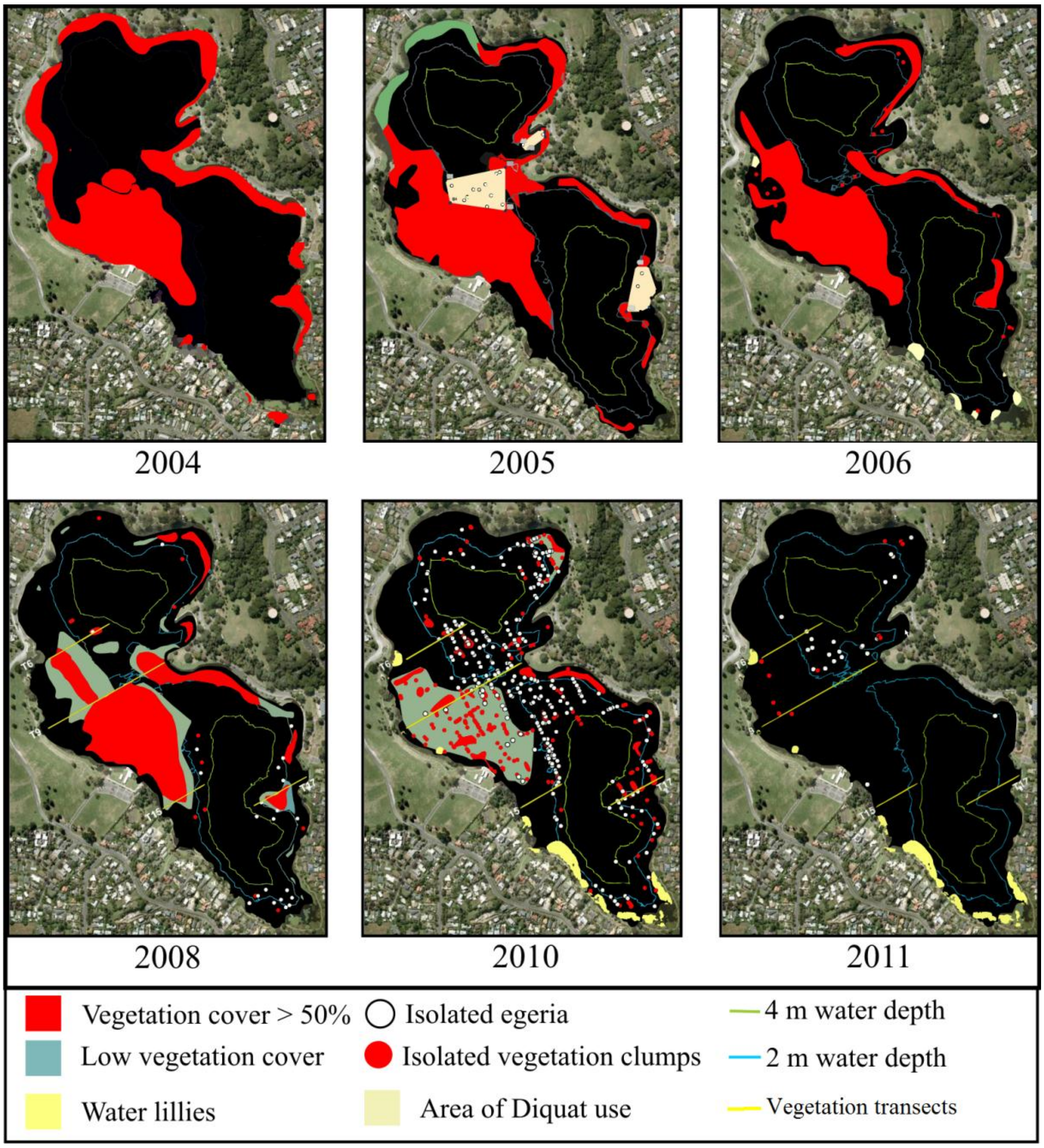

Figure 3.1: Mapped distribution of macrophyte cover in Lake Rotoroa from 2004 to 2011 as detected by differential GPS/sonar. Figures summarised from de Winton et al. (2004, 2005, 2006, 2008, 2010, 2011).

1.16.2 Secchi Depth

Secchi depth is used as an indication of water clarity in lakes and marine waters (Tyler, 1968). Historically, Secchi depth has ranged from >5 m in 1977 (Graynoth unpublished) decreasing to 2.2 m in 1981 (Town, 1981), and 1.9 m in 1983 (Etheredge, 1987), with consistently low recordings of 0.5 m between 1988 and 1992 (de Winton, 1994b). The Secchi depth of Lake Rotoroa (Figure 3.2) shows an improvement in water clarity, from less than 1 m in 1992 to over 2 m in 2009. Since 2009 Secchi depth has deteriorated with a decrease to between 1.5 m and 1 m. The increased Secchi depth is a partial response to decrease chlorophyll *a* (Figure 3.2) and inorganic suspended solids between 1992 and 2012. It is unknown if the poor water clarity in 2011 is a trend or due to specific conditions during dates of sampling.

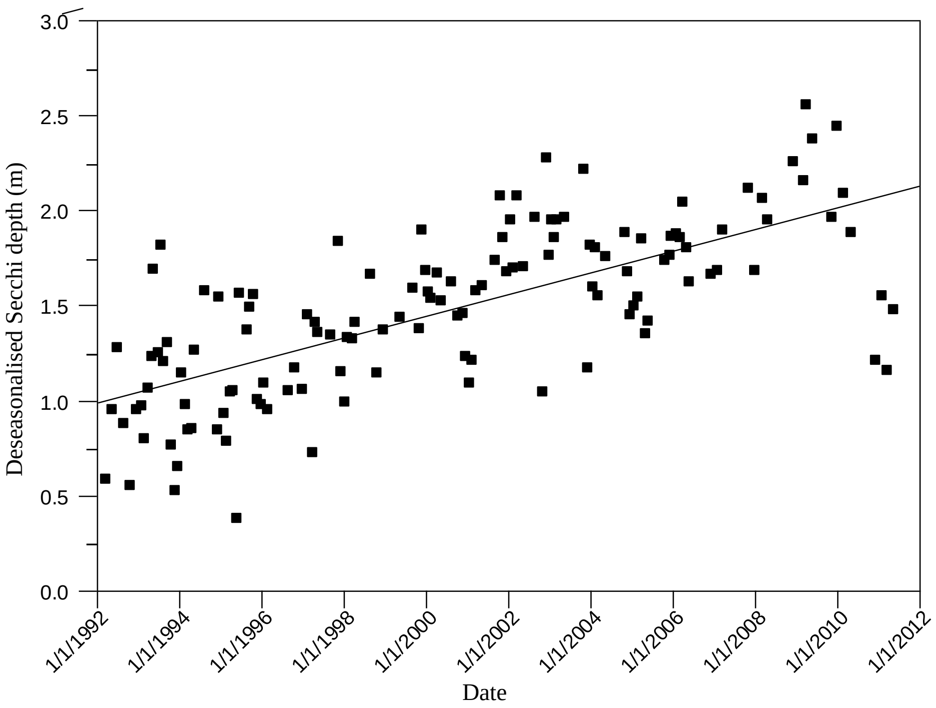

Figure 3.2: Deseasonalised Secchi depth in Lake Rotoroa between 1992 and 2012 (n=131), equivalence test of deseasonalised slope was significant (*P* < 0.01).

1.16.3 Chlorophyll *a*

Historically, Town (1981) recorded chlorophyll *a* between 5.5 mg m^{-3} and 15 mg m^{-3} these levels had increased to between 5 mg m^{-3} and 77 mg m^{-3} in 1987 (de Winton, 1994b). There has been an overall significant decrease in chlorophyll *a* between 1992 and 2012 (Figure 3.3). Chlorophyll *a* has fallen from a yearly mean of 28.1 mg m^{-3} in 2003 to 6.1 mg m^{-3} in 2011. Over this time period there has been continual adaption of the dominant phytoplankton species (outlined in section 1.8.1) as a result of changing nutrient conditions.

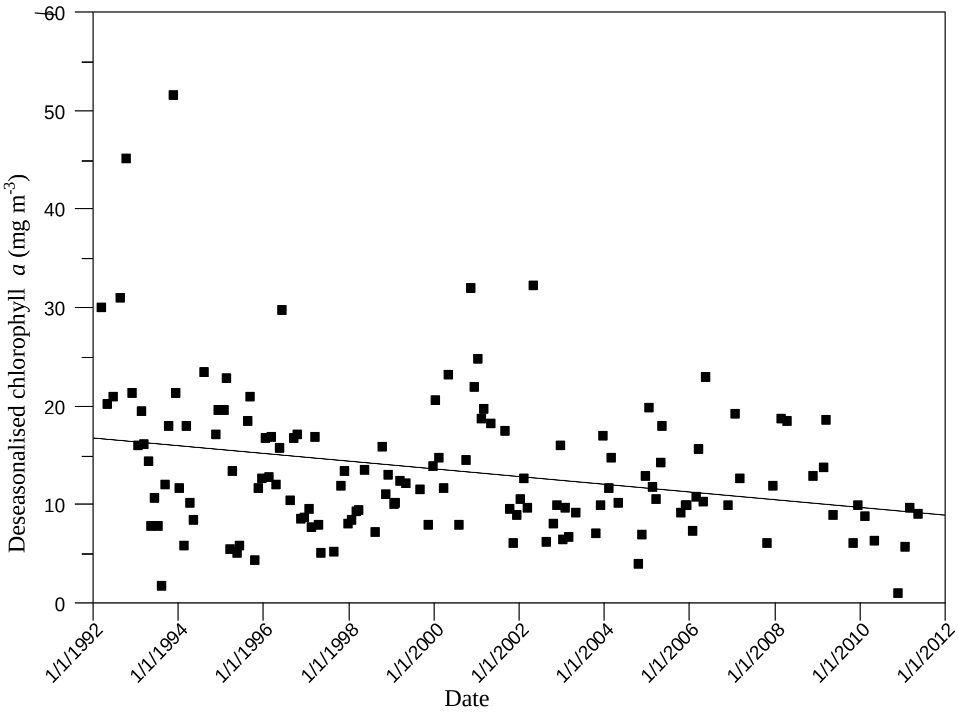

Figure 3.3: Deseasonalised chlorophyll *a* in Lake Rotoroa between 1992 and 2012 (n=141), equivalence test of deseasonalised slope is significant ($P < 0.01$)

1.16.4 Nitrogen

Nitrogen has fluctuated historically, with Henriques (1979) documenting total nitrogen levels ranging from 415 mg N m^{-3} to 2325 mg N m^{-3}. Nitrogen concentrations in have not decreased significantly in the 20 years of monitoring (Figure 3.4). Figure 3.3 shows cyclical periods of extreme high nitrogen levels, peaking at 1355 mg N m^{-3} in June 2002, and 1500 mg N m^{-3} January 2007. There has been no significant decline in nitrogen between 1992 and 2012.

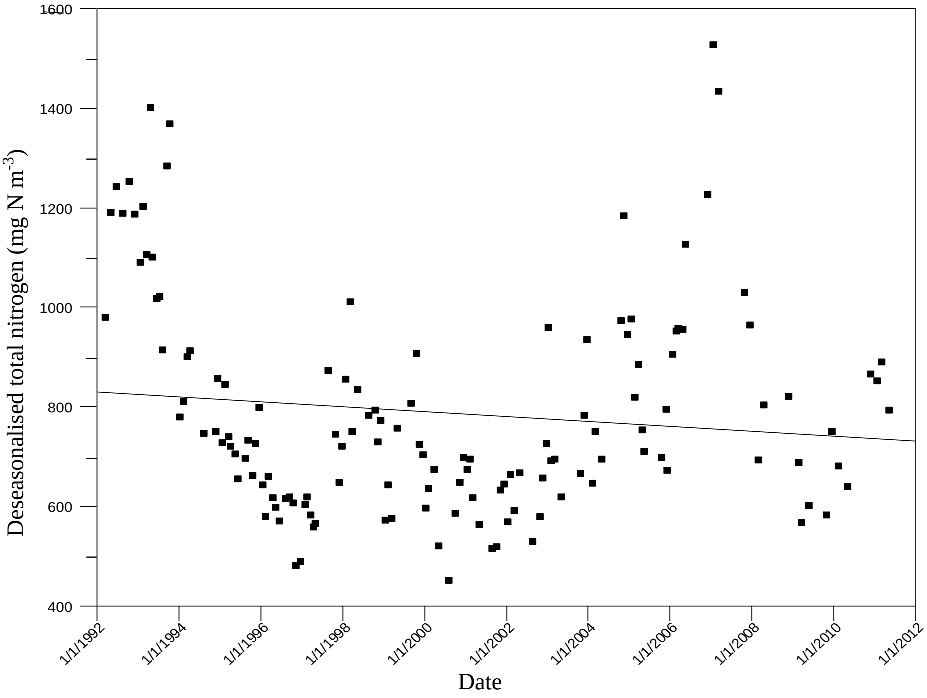

Figure 3.4: Deseasonalised total nitrogen concentration in Lake Rotoroa between 1992 and 2012 (n=139) Equivalence test of deseasonalised slope was not significant (P = 0.17)

1.16.5 Phosphorus

Historically total phosphorus has ranged from 17-112 mg P m^{-3} and 12-39 mg P m^{-3} in 1979 (Henriques, 1979) and in 1981 (Town, 1981), respectively. Figure 3.5 shows there has been a significant decrease in total phosphorus. Annual mean total phosphorus concentrations have decreased from 33.5 mg P m^{-3} in 1992 to 19.75 mg P m^{-3} in 2011 mg P m^{-3}. Periods of anoxia in the deep northern and southern basins during summer cause releases of phosphorus from the sediments (de Winton et al., 2011). Dissolved reactive phosphorus levels are low with average summer concentrations below 2 mg m^{-3}.

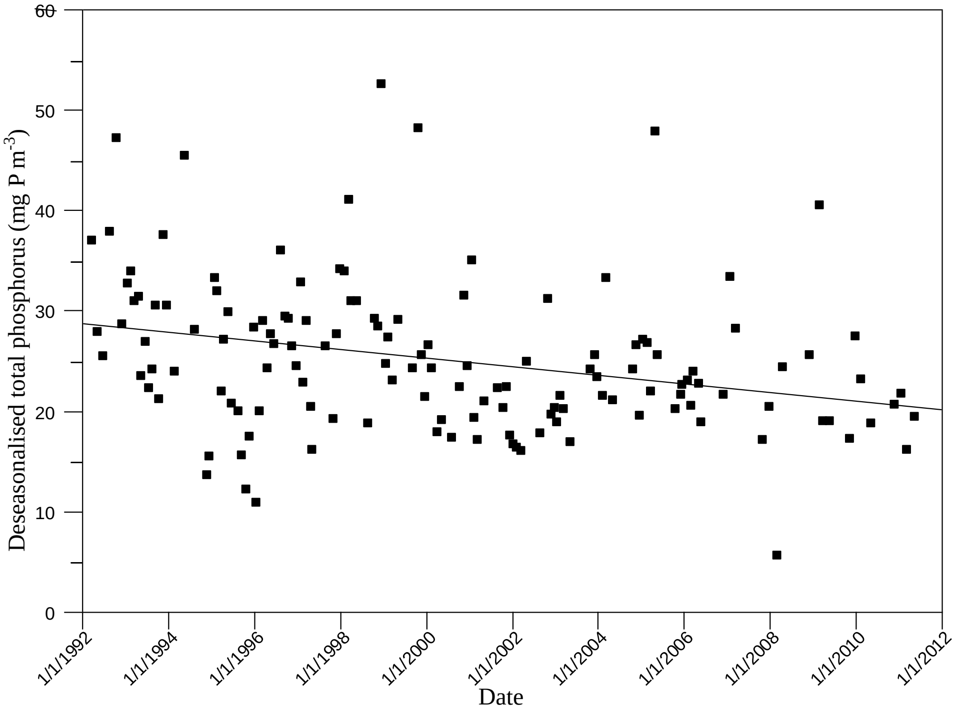

Figure 3.5: Deseasonalised total phosphorus concentration in Lake Rotoroa between 1992 and 2011 (n=139), Equivalence test of deseasonalised slope was significant ($P < 0.01$).

1.16.5.1 Nitrogen: phosphorus ratio

Nitrogen: phosphorus (N:P) ratio shows an insignificant ($P = 0.11$) increase associated with decreasing levels of phosphorus as the trophic state of the lake has improved (Figure 3.6). In 1992 the annual ratio of N:P was 37.5 this is similar to 2011 ratio of 39.6, the highest annual average ratio was recorded in 2007 at 51.4.

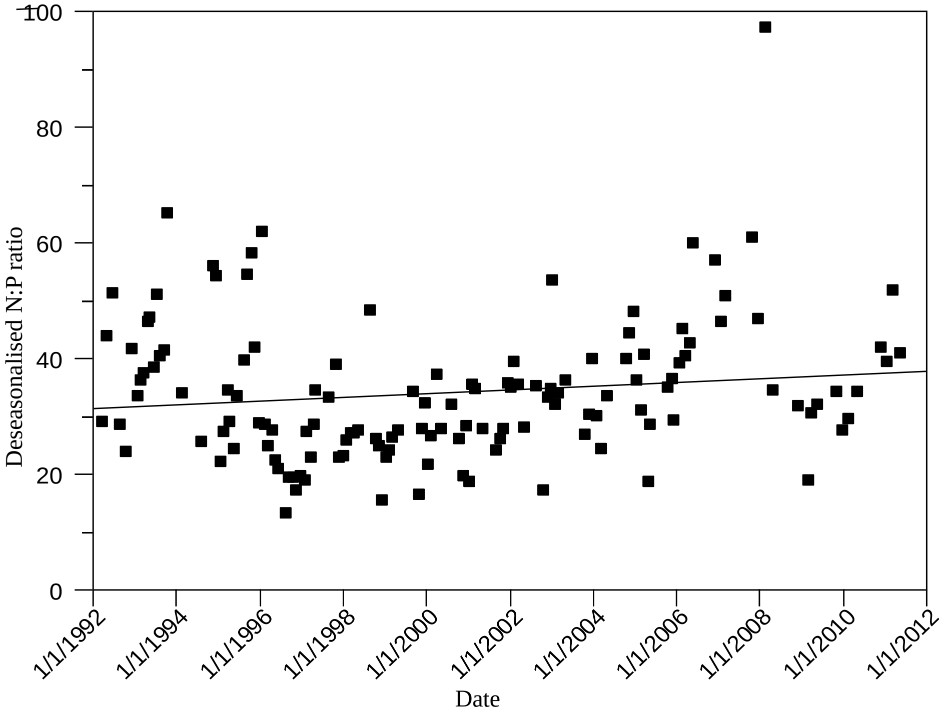

Figure 3.6: Deseasonalised ratio of total nitrogen concentration: total phosphorus concentration in Lake Rotoroa between 1992 and 2011 (n=136). Equivalence test of deseasonalised slope was not significant (P = 0.11).

1.16.6 Trophic level index

Annual trophic levels have decreased from supertrophic scores of 5.4 and 5.1 in 1992 and 1993 respectively, to annual trophic levels scores of 4.4 and 4.5 in 2010 and 2011, respectively (Figure 3.7). This represents a shift from a supertrophic lake to a eutrophic lake, mainly due to the decrease in total phosphorus, chlorophyll *a*, and increase of Secchi depth.

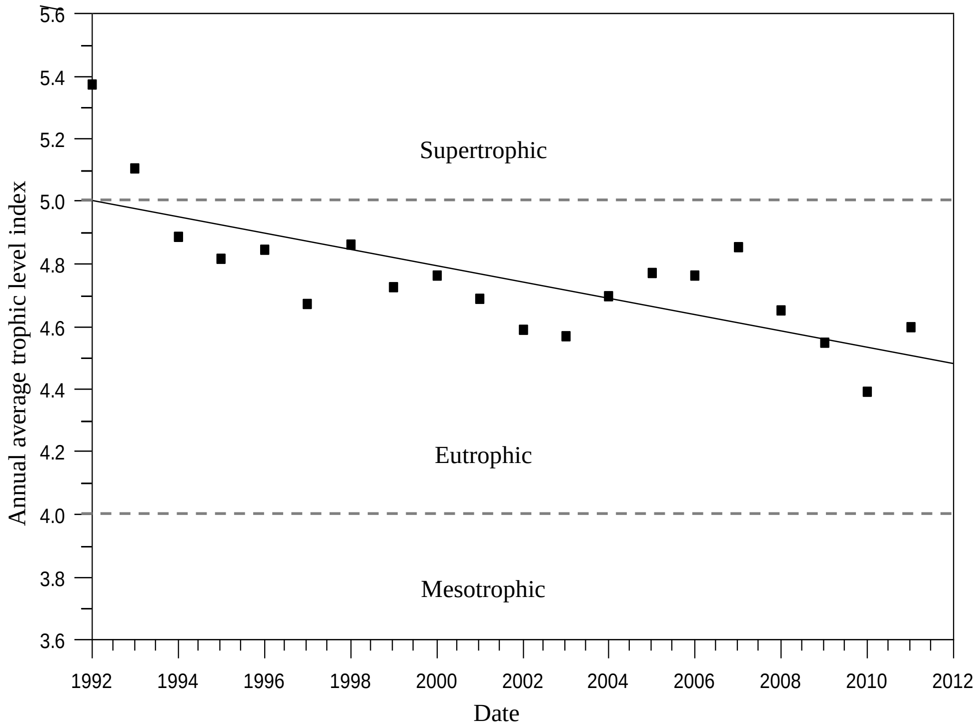

Figure 3.7: Annual trophic level index in Lake Rotoroa between 1992 and 2011 (n=123), Equivalence test of slope was significant ($P < 0.01$). Dashed lines indicate boundaries between trophic levels (Burns et al., 2000).

1.16.7 Water clarity relationships

Inorganic suspended solids (Figure 3.8) and organic suspended solids (Figure 3.9) have decreased significantly over the 20 year monitoring period, while the ratio of inorganic suspended solids to total suspended solids has remained unchanged (Figure 3.10). The proportion of inorganic suspended solid of total suspended solids is closely correlated (Figure 3.11); approximately 75% of total suspended solids are organic. This decrease in total suspended solids is derived from a combination of a reduction in chlorophyll *a* and inorganic suspended solids.

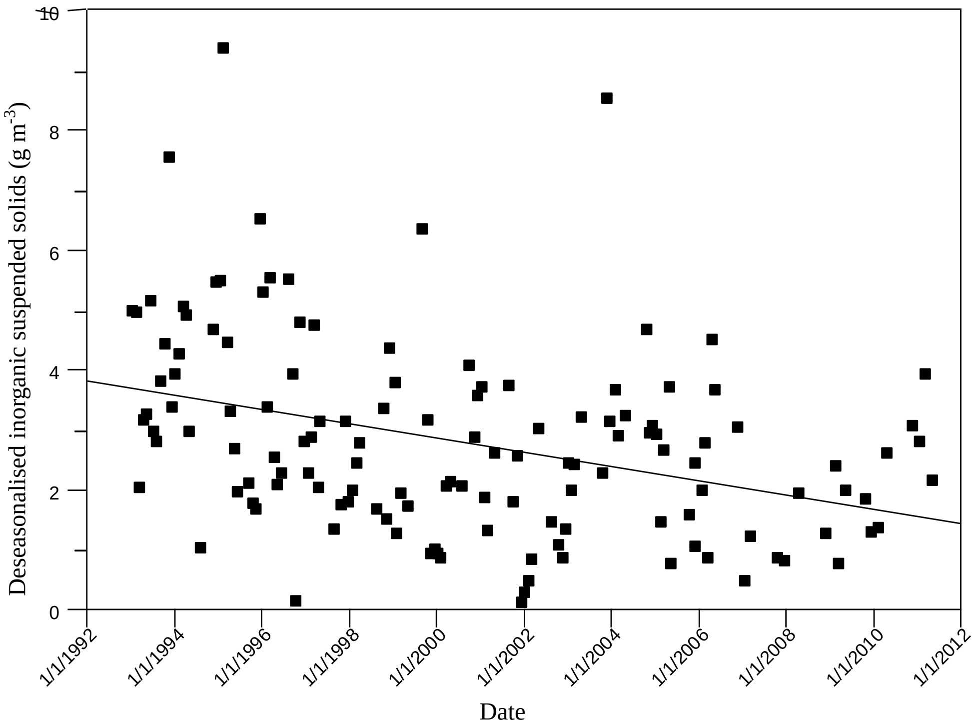

Figure 3.8: Deseasonalised inorganic suspended solids in Lake Rotoroa between 1992 and 2012 (n=131), equivalence test of deseasonalised slope was significant (*P* < 0.01).

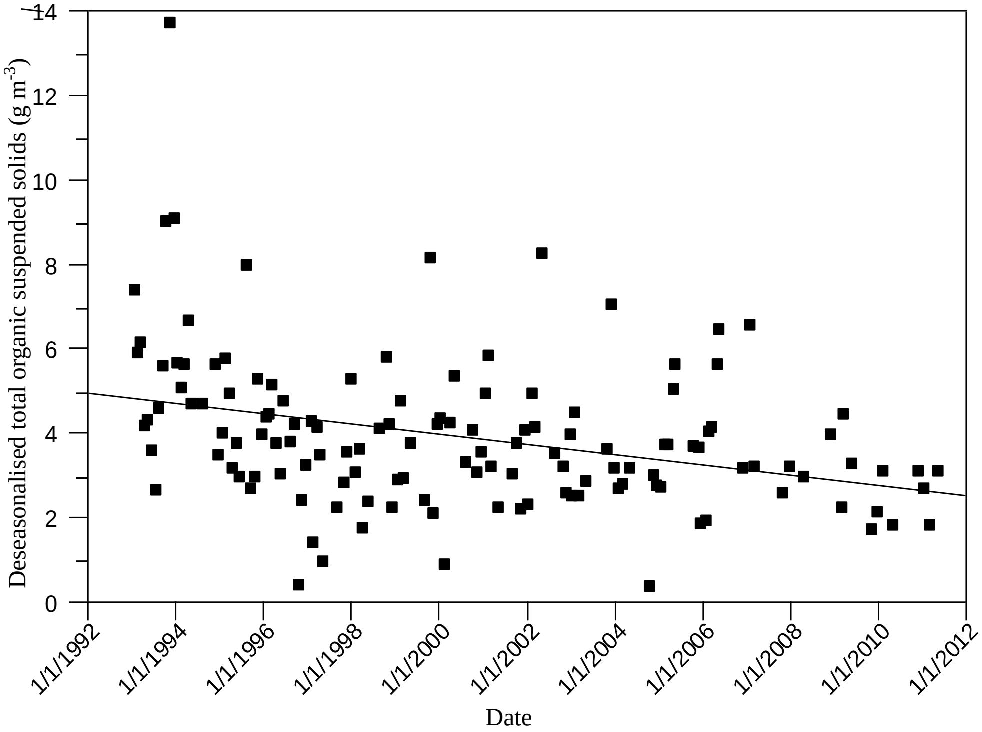

Figure 3. 9: Deseasonalised organic suspended solids in Lake Rotoroa between 1992 and 2012 (n=131), equivalence test of deseasonalised slope was significant (*P* < 0.01).

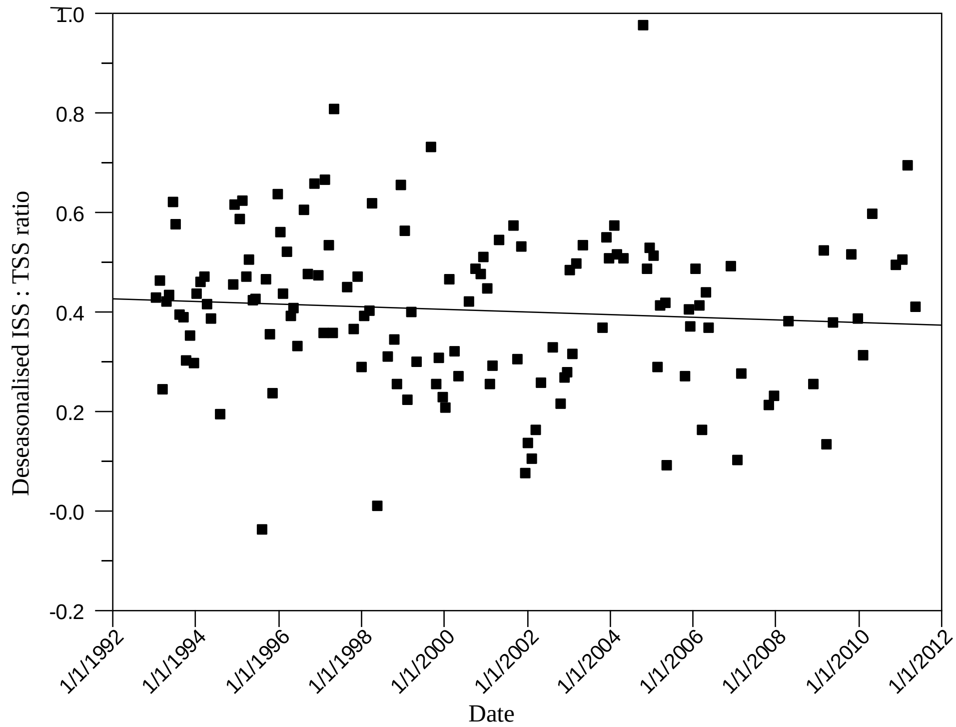

Figure 3.10: Deseasonalised ratio of inorganic suspended solids (ISS): total suspended solids (TSS) in Lake Rotoroa between 1993 and 2012 (n=132), Equivalence test of deseasonalised slope was not significant (P = 0.24).

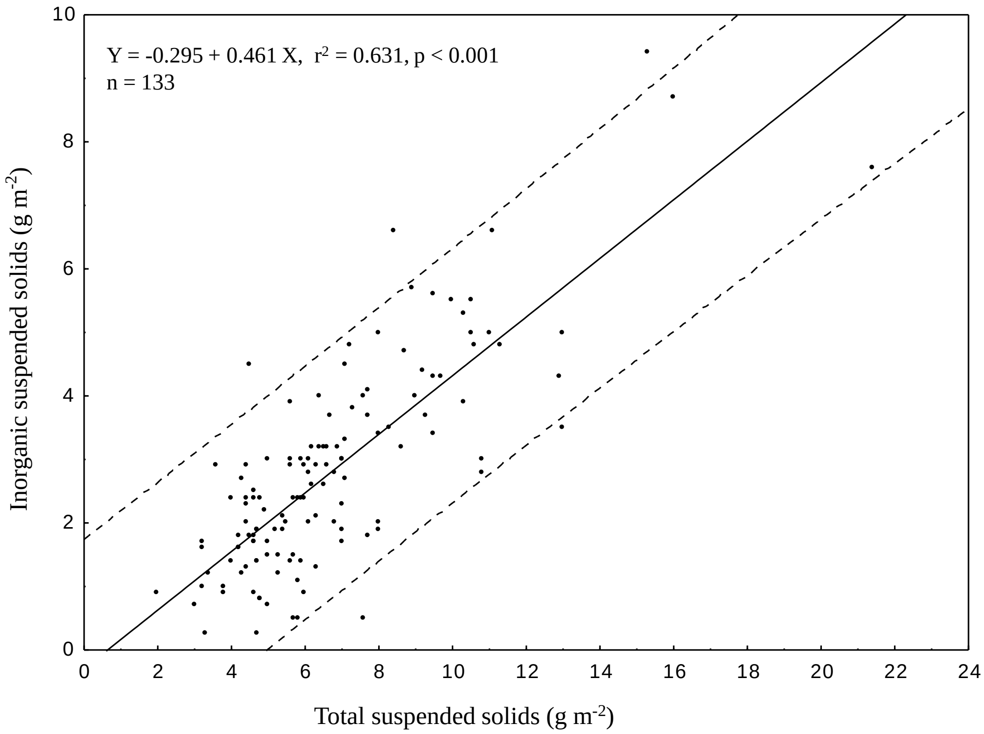

Figure 3.11: Regression of inorganic suspended solids vs. total suspended solids, with linear trend line and 0.95 prediction bands

1.16.8 Ammonia: nitrate ratio

Nitrogen in Lake Rotoroa is dominated by ammonium, with a high ratio of ammonium to nitrate, with a yearly average of 17.7 in 2003, decreasing to 3.36 in 2011. Figure 3.12 shows the statistically significant decrease in the ratio of ammonium to nitrate, illustrating an increase nitrate levels as there has been no significant change in levels of ammonium. Levels of nitrate and ammonium are poorly correlated (Figure 3.13) with increasing nitrate levels associated with high levels of ammonium. Ammonia and nitrate levels follow an annual cycle of depletion between February and April caused by uptake by phytoplankton followed by accumulation in the water column between June and October.

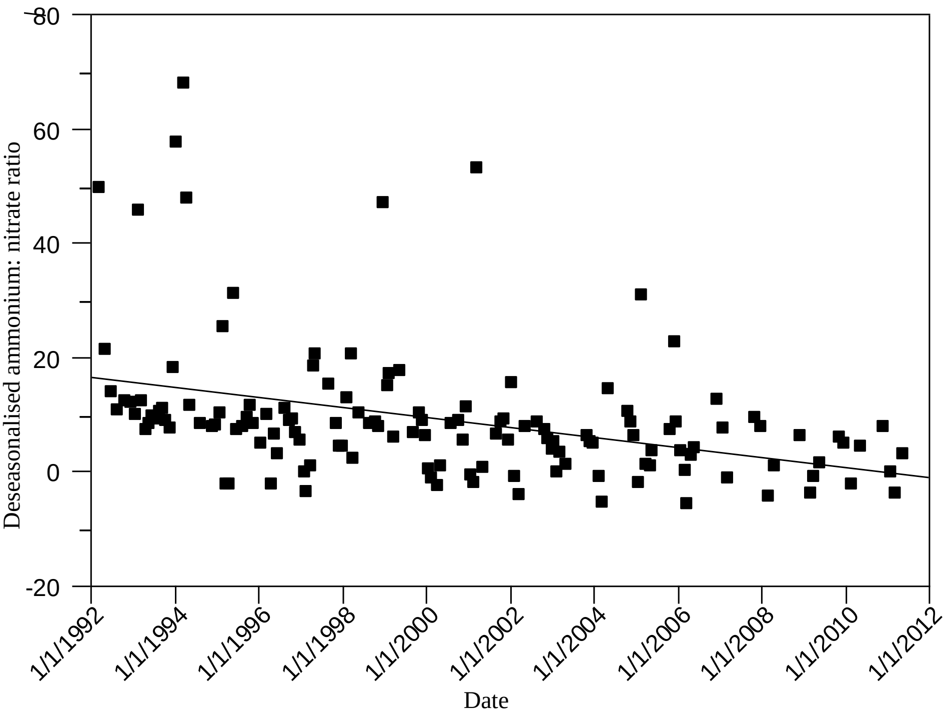

Figure 3.12: Deseasonalised ratio of ammonium concentration: nitrate concentration in Lake Rotoroa between 1992 and 2011 (n=139). Equivalence test of deseasonalised slope was significant (P < 0.01)

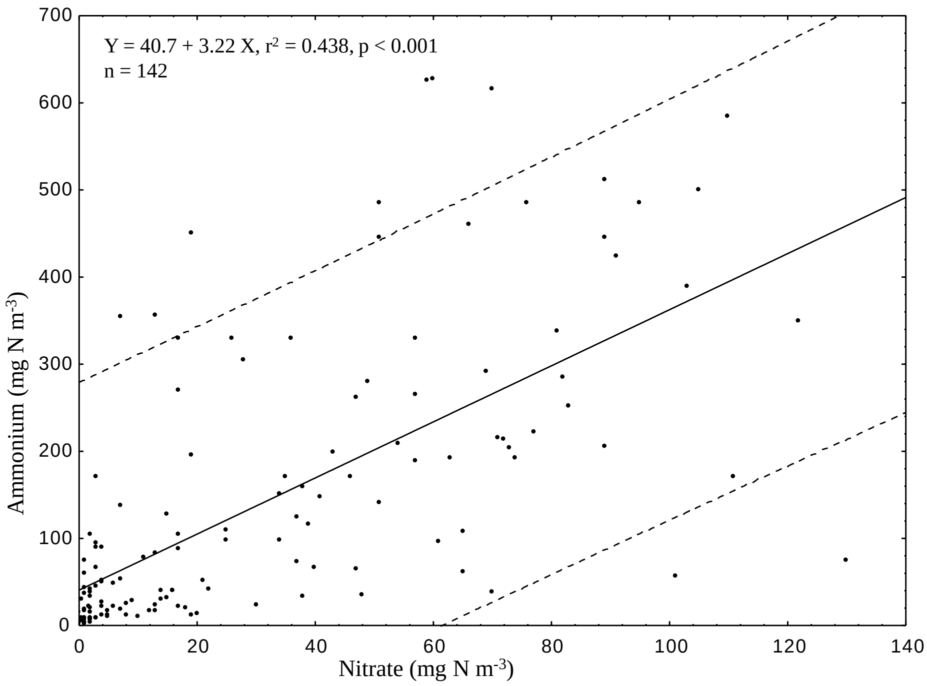

Figure 3.13: Regression ammonium vs nitrate in Lake Rotoroa between 1992 and 2011, linear trend line and 0.95 prediction bands.

1.17 Discussion

1.17.1 Macrophyte collapse

Several hypotheses have been given to explain the decline of macrophytes in Lake Rotoroa. Biotic factors include grazing and foraging by lake fauna, plant population cycles, and microcystins from cyanobacteria, while abiotic factors include resource depletion, competitive interactions, reduction in water clarity, and metrological events (de Winton, 1994a).

1.17.1.1 Fish

Generally fish can have a range of effects on macrophyte population through direct and indirect processes. Directly certain species will feed on submerged macrophytes leading to reduction in macrophyte cover (Williams et al., 2002).

Indirectly, forging by benthic feeding fish results in sediment and nutrient resuspension (Meijer et al., 1990).

In Lake Rotoroa, rudd has been proposed as a species that may have led to the original macrophyte decline in 1990, although no direct evidence was found (Wise, 1990, Hicks, 1994, Roberts, 2002). *Egeria densa* and the native cryophyte species that have been shown to be highly palatable to rudd (Lake et al., 2002), although they are not selective species, eating the most suitable sized submerged macrophytes that are available (Nurminen et al., 2003). Lake et al. (2002) found that the native *Nitella* spp. was the most preferred by rudd, while the consumption rate of *Egeria densa* was 16.5 mg dry weight plant per g of flesh weight per day. The lack of a large aquatic predator in New Zealand allows rudd to reach high populations densities. Thus rudd have a significant ability to modify and limit macrophyte communities in New Zealand (Lake et al., 2002). Rudd may have played a significant role in the original decline in macrophytes, but the low densities recorded between 2005 and 2012 (Chapter 2) support a reduced effect since 2003 that does not explain the current decline.

Indirect effects of fish on macrophytes are associated with benthic feeding fish resuspension sediment, increasing nutrients and turbidity (Meijer et al., 1990, Brönmark and Weisner, 1992). Meijer et al. (1990) found that reducing benthic fish population resulted in a significant reduction in inorganic suspended sediment when compared with control lakes. Rowe (2007) concluded that the resuspension of sediment and increased nutrients in the water column, promoted phytoplankton growth reducing water clarity and light availability to macrophytes. This switches a lake from a clear-water state to a turbid, phytoplankton dominated state. Evidence of disturbance by benthic fish has been observed by divers in Lake Rotoroa (de Winton et al., 2002). de Winton et al. (2007) noted that fish disturbance was concentrated in shallow areas and that clumps of plants in these areas are associated with debris such as sticks, blocks, and tyres, suggesting that underwater obstacles limit access and disturbance from benthivorous fish. Dugdale et al. (2006) made comparisons between charophyte establishment in small fish excluding enclosures and in the open lake in Lake Rotoroa. They found that charophytes in the open water had a significantly reduced the ability to recolonise the lake bed. The biomass in the enclosures was approximately 9 g of dry weight per pot heavier than the charophytes in open water (Dugdale et al., 2006). Grazing by water fowl is known to cause significant disturbance to submerged macrophytes (Lauridsen et al., 1994). De Winton et al. (2002) concluded that the effect of water fowl of macrophytes was low as densities of coot (*Fulica atra*) and black swan (*Cygnus atratus*) were one to two orders of magnitude less than that required to influence macrophytes.

Although there is no record of high periphyton in Lake Rotoroa, the shading effect of periphyton can lead to reduction in macrophytes. Tench have been speculated to stimulate periphyton growth through trophic cascade effects by predation of gastropods (Brönmark, 1994, Jones and Sayer, 2003). Williams et al (2002) found that tench biomass as high as 20 g m^{-2} may be needed to cause a macrophyte reduction, but the highest recorded tench biomass in Lake Rotoroa was 2.5 g m^{-2}, well below this biomass.

1.17.1.2 Macrophyte cover and exotic fish abundance

The relative abundance of fish in Lake Rotoroa has been changed through the introduction of exotic species and modifications to the fish assemblage. (Chapter 2). These changes are expected to have caused fluctuations in macrophyte coverage (Table 2.9) (Figure 3.14). The peak of rudd relative abundance in 1990 has been predicted to be linked to the collapse of macrophytes in 1990 (Figure 3.14) (Hicks, 1994), since this period relative abundance of rudd has collapsed with low capture rates since 2005. Catfish are capable of macrophyte disturbance when foraging for food; relative abundance catfish peaked in 2001 and appear to be unrelated to macrophyte abundance. Macrophyte surveys linked pitting in lake bed from foraging benthic feeding fish this might have contributed to macrophyte collapse (de Winton et al., 2007). High catfish densities could be a contributing factor in the collapse of macrophytes in 2011. Perch population seems to increase after macrophyte collapse, with increasing relative abundance during the periods of low macrophyte coverage (Figure 3.14). This may be due to favourable conditions when searching for food, for example, reduced cover for zooplankton increasing exposure to predation (Jeppesen et al., 1997a), or due to reduced competition from those species that rely on macrophytes directly or indirectly. Tench had low population biomass that did not seem to respond to macrophyte abundance. Their dependence on submerged macrophytes to spawn may limit breeding in Lake Rotoroa while macrophyte coverage is low. The irregularity of fish population surveys on Lake Rotoroa and the change in methods over the last 20 years did not permit definite conclusions to be made on the exact species that are causing or influencing macrophyte coverage, but perch abundance seems to be inversely related to macrophyte abundance.

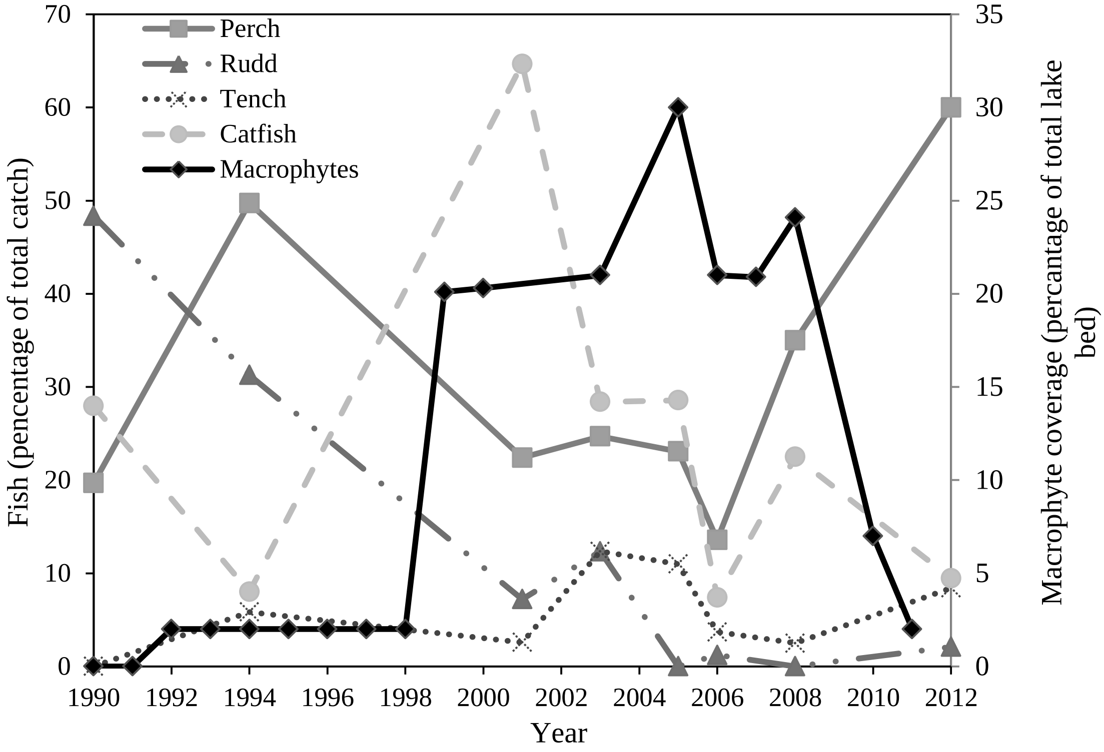

Figure 3.14: Macrophyte coverage and exotic fish species relative abundance in Lake Rotoroa 1990-2012 (chapter 2). Macrophyte lake Coverage, 1990 – 1999 estimated from Lake Rotoroa monitoring reports ((Burns et al., 1995, Burns et al., 1997, Burns et al., 1998, de Winton et al., 1999), 2000-2012 lake sonar surveys (de Winton et al., 2008, de Winton et al., 2010, de Winton et al., 2011)

1.17.1.3 Nutrient limitation

Nutrient limitation could be causing macrophyte decline. The effect of nutrient limitation is not predictable as the there is a complex set of processes that affect nutrient availability to submerged macrophytes (Barko et al., 1991). There has been no analysis of sediment nutrient availability to make conclusions on its role in macrophyte decline. de Winton (1994a) proposed that high concentrations of arsenic in the sediments might cause competition between arsenic and phosphorus by macrophyte roots.

1.17.1.4 Effects of microcystins

Microcystins are produced by several genera of phytoplankton including two that have recently blooming in Lake Rotoroa (*Microcystis* and *Dolichospermum* (*Anabaena*)). Adverse effects of microcystins on submerged macrophytes include inhibition of germination or growth, inhibition of photosynthesis, and induction of oxidative stress (Babica et al., 2006). Although Babica et al. (2006) concluded that little evidence of the effects of microcystin exists in relevant environment conditions, exposure could make macrophytes prone to other stresses.

1.17.1.5 Water clarity

Reduction in water clarity has been suggested to be the cause of the original macrophyte collapse in 1990 (de Winton, 1994a), with a large decrease in Secchi depth occurring before the decline. Decreases in water clarity are assumed to be one of the triggers that causes lakes to switch to a turbid state (Scheffer et al., 1993). Improvements in water clarity over the last 20 years associated with and decrease in phytoplankton abundance (Figure 3.3) and suspended sediment (Figures 3.8 and 3.9) conflicts with the current depletion of macrophytes. Thus water clarity cannot be causing the current decline in macrophytes.

1.17.1.6 Meteorological events

Extreme meteorological events have known to cause catastrophic reductions in macrophyte cover through wind turbulence or high rainfall influencing catchment runoff. For example, in 1968, after a massive storm Lake Ellesmere, South Island, New Zealand switched from a clear to a turbid state, when it destroyed large areas of submerged macrophytes (McKinnon and Mitchell, 1994, Williams, 1979). The gradual decline of macrophytes recently experienced 2006 to 2012, would not support of extreme wind turbulence as a cause of the macrophyte collapse. Although smaller isolated metrological events may place increased stress on macrophytes and have a contributing factor to their decline.

Combinations of two or more of the listed disturbances are likely to be the cause of macrophyte decline in Lake Rotoroa, but we cannot conclude which processes are directly or indirectly contributing to the decline. It is likely that benthic-feeding catfish and the herbivorous rudd caused sufficient disturbance to macrophytes to cause the decline demonstrated by Dugdale et al. (2006) and de Winton et al. (2002) . The effect of tench on promoting periphyton growth cannot be excluded, and recent blooms of cyanobacteria, *Microcystis* and *Dolichospermum* (*Anabaena*), since 2009 may also be contributing though the production of microcystins.

1.17.2 Decrease in trophic level index

The decrease in trophic level index (TLI) (Figure 3.7) is mostly attributed to the decrease internal phosphorus concentrations in Lake Rotoroa. This contributed to reduced chlorophyll *a* concentrations and improvement in water clarity up to 2010, which was consistent with the reductions in organic and inorganic suspended sediment and reducing internal recycling of phosphorus from the sediments (Scheffer, 2004, Pettersson, 1998). The lake receives little external nutrients due to its urban catchment (Jenkins and Vant, 2007), with the major external phosphorus inputs from storm drains and faecal deposition from birds (Burns and Singleton, 1994). Burns and Singleton (1994) calculated that water fowl contributed 34% - 42% of the total external phosphorus load approximately 35.0 kg P per year (Dickie, 1994), while loading from stormwater drains contributes the rest (Burns and Singleton, 1994).

In comparison to other shallow peat lakes in the Waikato area with catchments in diary pasture (Figure 3.15), Lake Rotoroa is in comparatively good condition. Due to the fertile nature of the soil in the Hamilton basin, much of the natural vegetation has been cleared for pasture. Intensification in agricultural practices have seen an increase of fertiliser use, stocking rates and irrigation (Hamilton et al., 2010). Contributing to increased levels of nutrient runoff and leaching into the lakes with an associated increase in trophic level as water clarity has reduced. The trophic levels of the Waikato peat lakes (2003-2007) range from eutrophic Rotomanuka (TLI 4.7), to the hypertrophic, Kaituna (TLI 7.4) and Kimihia (7.5) (Figure 3.14), with the current trend of increasing trophic levels (Barnes, 2002, Hamilton et al., 2010). Verburg et al. (2010) noted that New Zealand wide pastoral farming was associated with eutrophication and ecological deterioration. Lake Rotoroa has been sheltered from large increases in lake nutrients due to its urban catchment limiting external nutrient inputs as a result of agricultural intenfacation (Jenkins and Vant, 2007).

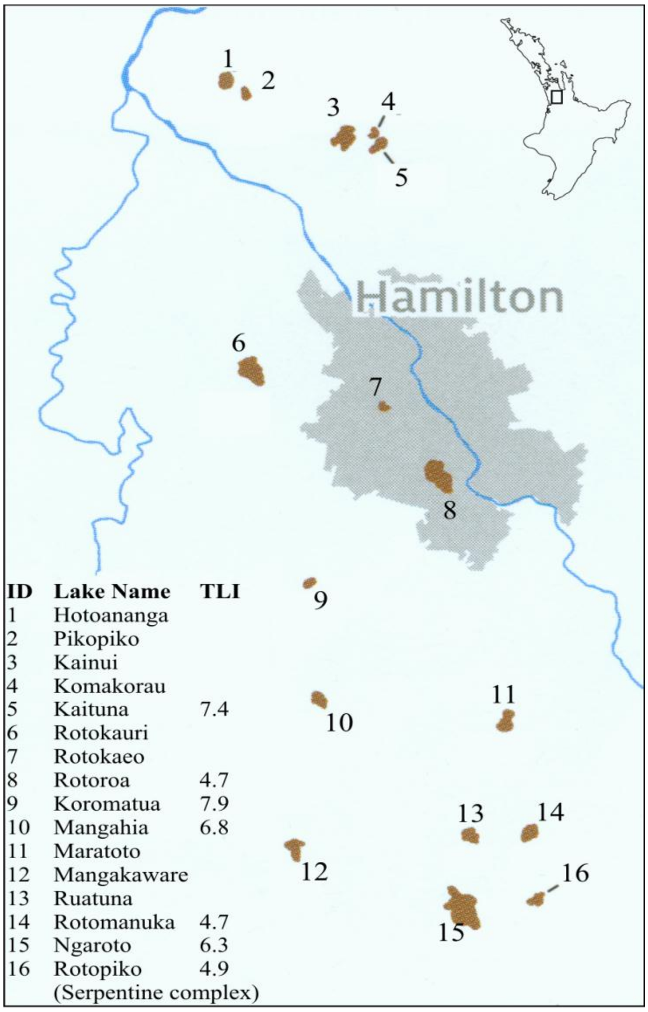

Figure 3.15: Peat lakes of the Hamilton basin. Modified from Collier et al. (2010a). Trophic level index data from 2006 – 2008, Lake Ngaroto 2002 (Hamilton et al., 2010).

1.17.3 Phosphorus limitation

Phosphorus has said to be the limiting nutrient for phytoplankton growth (de Winton, 1994b). Low concentrations of dissolved reactive phosphorus (summer average below 2 mg m^{-3}) suggest phosphorus is limiting. When comparing chlorophyll *a* used as a surrogate parameter for biomass, with total phosphorus, and predicted chlorophyll *a*, using Vollendeider and Kerkes (1982) OECD empirical model (Equation 3.1) (Figure 3.16). Where chl *a* is the yearly average chlorophyll *a* in the euphotic zone and P_{tot} is average yearly total phosphorus in the lake (Lampert and Sommer, 1997). Annual averaged chlorophyll *a* values are higher than the predicted chlorophyll *a*, suggesting that productivity is higher in Lake Rotoroa then other eutrophic lakes (Figure 3.16) used in the model. This higher productivity could be due to, limited top-down control of phytoplankton because of low zooplankton abundance or a delay in the phytoplankton response to reduced loading of phosphorus. This lag can result in phytoplankton becoming nitrogen limited and my explain blooms of nitrogen fixing cyanobacteria (Lampert and Sommer, 1997).

Equation 3.1: Vollendeider and Kerkes (1982) empirical model of primary production based on total phosphorus concentrations.

$$\text{Chl } a = 0.28\mathrm{P}_{tot}{}^{0.96}$$

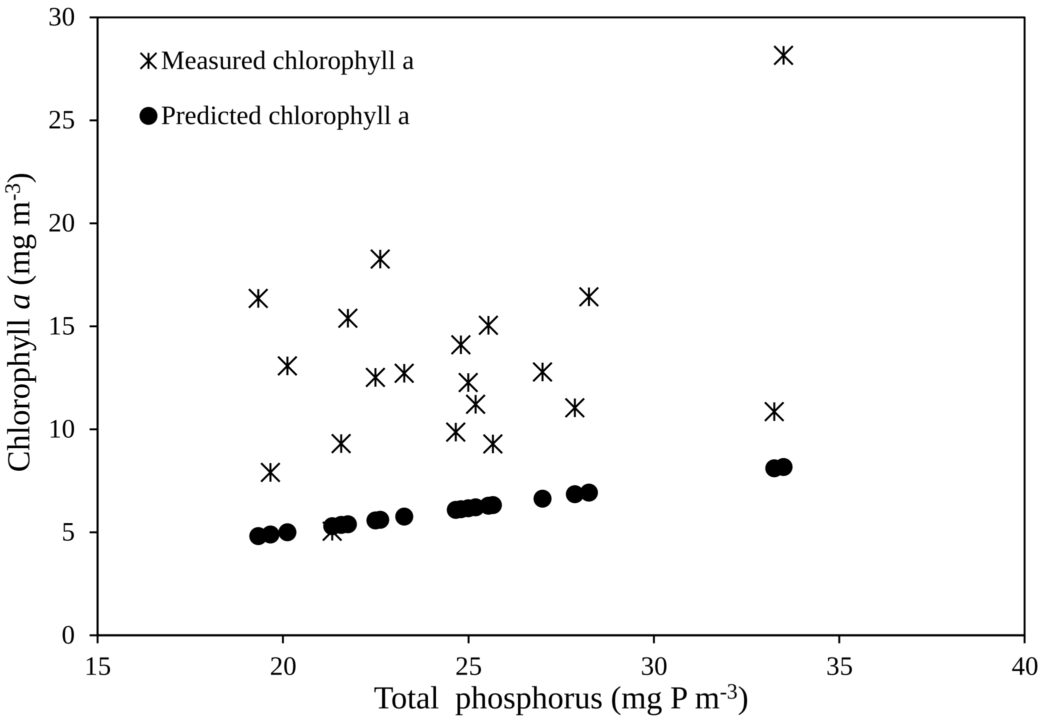

Figure 3.16: Yearly average measured and predicted chlorophyll *a* at a given total phosphorus level in Lake Rotoroa 1992 - 2012. Chlorophyll *a* predicted using Vollendeider and Kerkes (1982) model in Lampert and Sommer (1997) (Equation 3.1).

1.17.4 Nitrogen: phosphorus ratio

The N:P ratio has not changed significantly despite the decrease in phosphorus loading in the lake. It has been suggested that a N:P ratio below 29:1 by mass increases the likelihood of cyanobacterial blooms (Smith, 1983, Smith and Bennett, 1999). This is within the N:P range of Lake Rotoroa and it is plausible that recent summer blooms of cyanobacteria are due to nutrient concentrations, although several studies have shown that N:P ratios are is not a key factor for the dominance of cyanobacteria in northern temperate shallow eutrophic lakes (González Sagrario et al., 2005, Jensen et al., 1994). High N:P ratio for a phosphorus limited lake, and greater phytoplankton productivity at the a given total phosphorus level (Figure 3.16), could result in phytoplankton taking advantage of greater levels of nitrogen than normal, in phosphorus limiting conditions. This explains high productivity of the lake at the given phosphorus concentrations and explains the blooms of nitrogen fixing cyanobacteria in late summer. Nitrogen levels between 1200 mg m^{-3} and 2000 mg m^{-3} with phosphorus levels higher than 100 mg m^{-3} have been shown to contribute to loss of submerge macrophytes (González Sagrario et al., 2005). Nitrogen levels have been higher than 1200 mg m^{-3} in 1993 and in 2007. The latest nitrogen peak occurred at the same time as a decrease in macrophytes (de Winton et al., 2008), although it is unknown if high nitrogen levels experienced are a cause or a response to macrophyte decrease as phosphorus levels are below the 100 mg m^{-3}.

1.17.5 Macrophyte and water quality decline

Well documented declines in macrophytes are accompanied shift towards a high nutrients and suspended sediment in lakes (Scheffer, 2004, Sondergaard and Moss, 1997, Scheffer and Van Nes, 2007, Jensen et al., 1994). Unlike the vegetation declines in 1988-1990, the current decline appears not to have affected water quality as there has been a continued in decline in phosphorus and chlorophyll *a*, although a recent decreases in Secchi depth in 2011 may be an indication of a switch of lake conditions to a turbid state.

1.17.6 Ammonia and nitrate levels

Although Lake Rotoroa is well mixed for most of the year ammonia levels are high. Wetzel (2001) showed in a generalised model when that oxygen is available nitrification should be high, and should result in low levels of ammonia. Rates of nitrification are known to be influenced by many factors including substrate, oxygen, light, suspended sediment, pH (Berounsky and Nixon, 1990), and certain dissolved organic compounds such as tannins and the decompositiontal derivatives (Wetzel, 2001). In Lake Rotoroa, it is possible that dissolved organic compounds sourced from the peat lake bed results in reduced levels of nitrification.

1.18 Summary

Since the collapse of charophytes in 1990 there was a recovery from 1998 to 2005, probably due to improvement in water clarity allowing sufficient light to the bottom sediments. Recovery of macrophytes is expected to have contributed to reduction of suspended sediment resulting in an improvement in total phosphorus concentrations. This proceeded to reduce chlorophyll *a* levels improving water clarity allowing macrophytes to grow at greater depth. These improvements in phosphorus, chlorophyll *a* and Secchi depth lead to shift of the trophic level since 1992 from a supertrophic to a eutrophic state. A subsequent collapses of 2009 to 2011 is suspected to be due to grazing and disturbance by exotic fish, with disturbance observed by divers during vegetation surveys. Additional stresses such as blooming cyanobacteria producing microcystin also may have had an effect by contributing additional stresses on the charophyte beds. Unlike the past vegetation declines the current decline, appear not to have been affected by the nutrients levels with continued decline and phosphorus and chlorophyll *a*. However we may have just seen a shift to turbid state with the reduction in water clarity in 2011 which may be followed by a decrease in the TLI. Improvements in water quality are significant and further reductions should be strived for. Further research on the possible stresses causing the macrophyte needs to be completed before conclusions can be made on the exact cause.

Chapter 4: Conclusions

The aim of this thesis develop a general understanding of the fluctuations in water quality and macrophyte community of Lake Rotoroa associated with introduction of exotic species into the ecosystem. This was completed by summarising the changes in fish assemblage, water quality and macrophyte coverage.

1.19 Fish assemblage

Lake Rotoroa has a diverse assemblage of fish species with four original natives and six exotic species. The native fish assemblage is dominated by shortfin eel which are present in high numbers, common bullies are present but their secretive nature limits their capture. Longfin eel were last caught in 1994, it is unknown if there are still individuals present in the lake. Smelt are no longer present in the lake due predation from perch as water quality was still high in Graynoth's survey's 1976-1978. Perch and catfish are the most relative abundant exotic species in Lake Rotoroa with large populations recorded since their introduction in 1907 and 1977, respectively. Perch populations are now dominated by large numbers of small individuals due to the lack of a large predator fish species. Catfish show a stable population, dominated by large individuals with a lack of young fish. Length-weight analysis shows that there has been a decrease in catfish condition, with individuals becoming longer thinner over time. Rudd, the most abundant species in 1990, population has decreases, with low numbers caught since 2003. Released illegally between 1978 and 1989, they have been contributed to the collapse of macrophytes and because of a lack of predators reach a large size. Tench released legally in 1990 to provide coarse fishing opportunities in Hamilton, show a small, stable population of large fish that dominate the biomass.

Once thought to have limited breeding potential due to water temperature and wind exposure at breeding sites, tench have shown strong recruitment with young individuals caught in 2001 between 2005. Evidence suggests that their breeding may have been limited by a collapse of macrophytes between 1990 - 1998 and 2010 – 2012. Goldfish show a small population with limited fish captures since 2005, population size may be limited due to interspecific competition. Comparison between fish surveys is limited by different methods used resulting in a bias towards certain species based on size and selectivity.

1.20 Macrophytes

In 1973 Lake Rotoroa had a diverse range of native macrophytes including three species of charophytes, two native *Potamogeton* species and *Glossostigma submersum* (table 1.1). The introduction of *Lagarosiphon major* in the late 1950 and *Egeria densa* in 1977 resulted in a change in species dominance with the exotic species outcompeting the natives, which grew to a lower height. To control the exotic species that were limiting recreational use of the lake, sodium arsenite was applied to in 1959, and Diquat applied at regular intervals since this period. In 1990 the macrophyte community collapsed causing the lake to switch to a turbid state. Since 1998, the charophyte community has recovered, with maximum coverage of 30% of lake bed in 2005. *Egeria densa* was first observed again in 2004 with hand weeding and two applications of Diquat limiting its spread. A second collapse of the macrophyte community occurred in 2009 with only small clumps of macrophytes present in 2011. Fish disturbance directly by herbivorous rudd and indirectly by benthic feeding fish disturbing sediment and reducing water clarity have been proposed as the leading cause of macrophyte collapse.

Enclosure experiments have showed that when fish are excluded macrophytes can reach high densities (de Winton et al., 2002, Dugdale et al., 2006). Other disturbances cannot be excluded as it is expected that fish disturbance alone would not result in the complete macrophyte collapse experienced. This could include effects such as microcystin producing cyanobacteria, increased turbulence or reduced water clarity.

1.21 Water quality

Water quality in Lake Rotoroa has shown an improving trend since 1992 with a decrease in trophic level from a supertrophic 5.3 in 1992 to the eutrophic 4.3 in 2010. This improvement has been driven by decrease in total phosphorus concentrations limiting phytoplankton biomass and increasing Secchi depth. This improved water clarity allowed macrophytes to re-establish promoting further increase in water quality as resuspension of sediments and release of nutrients decreased. There has been no decreasing trend in total nitrogen as seen with other key TLI variables. Total nitrogen has undergone periods of extreme concentrations, the cause of this is unknown but it does not seem to have had long lasting effects on water quality. Since 2009 the macrophyte community has collapsed with an associated by a decrease in water clarity. This has not been associated with an increase in phosphorus or chlorophyll *a* which continued to decline. Lake Rotoroa has been sheltered from a trend of decreasing water quality in Waikato peak lakes due to its urban catchment with small external nutrient inputs into the lake.

1.22 Recommendations for future research

The effect of tench introductions on Lake Rotoroa needs to be quantified. This species has recently been released with little information available on the true effects of the large bodied species on the ecosystem. Overseas, tench have been implicated in environmental changes including reduced invertebrate density, reduced macrophytes, and reduced water clarity in shallow lakes (Rowe, 2004).

Comparisons between boat electrofishing and the passive gill and fyke netting used up to 2001, would allow better estimate of change in fish assemblage over time as currently only comparative and relative estimates were able to be made. The effort and time required to undertake netting surveys makes repeating theses intensive surveys restricted on funding, whereas the relatively small amount of time and personnel needed to undertake a boat electrofishing survey allows for continual repetitive sampling over greater lengths of time, allowing changes in fish assemblage associated with changes in environmental variables to be quantified more easily. Despite the extra resources needed, it is clear that another netting survey using the same methods as Wise (1990), Kane (1995), and Roberts (2002) should be undertaken as soon as possible.

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