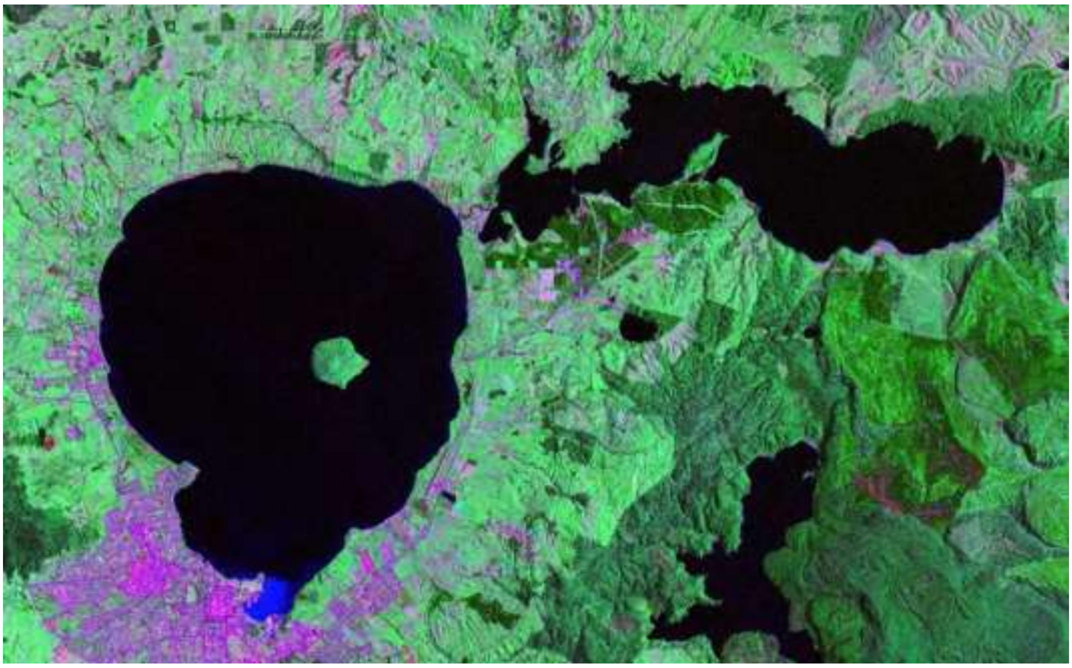
Nutrient budget for Lakes Rotoiti and Rotorua

Part I: Internal Nutrient Loads

CBER Contract Report 76

Report to the Lakes Water Quality Society

By David Hamilton, Will Alexander & David Burger



Centre for Biodiversity and Ecology Research
Department of Biological Sciences
School of Science and Engineering
The University of Waikato
Private Bag 3105
Hamilton, 3240
New Zealand

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Reviewed by:	*Approved for release by*:
C McBride	Brendan
Chris McBride	Brendan Hicks
University of Waikato	University of Waikato
Affiliation	Affiliation

Executive Summary

Time series data of temperature and concentrations of dissolved oxygen and nutrients (total nitrogen - TN, total phosphorus - TP, nitrate - NO_3, ammonium - NH_4 and filterable reactive phosphorus - FRP) were examined for trends in Lake Rotoiti for 1981-82 and 1990-2003 and in Lake Rotorua for 2002-03. TP, NO_3, NH_4 in bottom waters (40-60 m) of Lake Rotoiti, below the thermocline, were examined using regression equations to quantify seasonal transformations of TP, NO_3 and NH_4. There is a relatively linear build-up of phosphorus through time in the hypolimnion following the onset of stratification and initiation of deoxygenation until the time of winter mixing. When there were high frequency nutrient data for Lake Rotoiti (1981-82 at 60 m, 1991-94 at 60 m and 2001-02 at 40 m depth) the rate of seasonal phosphorus build-up could be quantified by regression to within a relatively narrow range of 111 mg TP m^{-3} yr^{-1} (0.304 mg TP m^{-3} day^{-1}) in 1992-93 to 136 mg TP m^{-3} yr^{-1} (0.373 mg TP m^{-3} day^{-1}) in 1981-82. Based on the first and last points of all the regression equations developed for TP increase, approximately 14.4 (1993-94) to 20.5 tonnes (2001-02) of phosphorus may be added to the hypolimnion of Lake Rotoiti during stratification. There is no evidence of a consistent interannual change in the seasonal build-up of phosphorus over the years considered and the reasons for interannual variations in the rates are not immediately apparent.

Changes in concentrations of inorganic nitrogen in the bottom waters of Lake Rotoiti are rapid and necessitate a higher frequency monitoring programme than currently used to confidently capture the abrupt peaks of nitrate and ammonium associated with different phases of stratification and deoxygenation. The onset of stratification is characterised by an initial peak of ammonium, followed by a peak of nitrate as the regenerated ammonium is oxidised by nitrification. This nitrate is rapidly removed as gaseous nitrogen through denitrification when dissolved oxygen disappears from the hypolimnion, and a second peak of ammonium then occurs near the end of the stratified period. Regression line fits to the different build-up phases of nitrate and ammonium through time are highly variable and do not provide any insight into interannual variations in processes controlling these nutrient species. The amount of nitrate that was denitrified ('lost') seasonally was estimated from the difference

between the peak nitrate concentration during the stratified period and the minimum concentration attained just prior to winter mixing, when oxygen was absent from the hypolimnion. This calculation did not differentiate the eventual fate of the nitrate (e.g. transformation to nitrite, nitrous oxide, nitrogen gas or ammonium). The large range, 40 tonnes (1995) to 83 tonnes (1982), reflects in some measure inability to adequately capture the abrupt seasonal peak of nitrate but is nevertheless indicative of a high proportion of total N in the lake that is denitrified.

While there was little to suggest that there has been a consistent trend in temperature, dissolved oxygen or nutrients in the contemporary data (1990-2003), there were some important differences between 1981-82 and 2002-03. The duration of temperature stratification was considerably greater in 2002-03 as it was initiated 2-3 months earlier than in 1981-82. Other studies of climate warming trends for lakes have found a shift in phytoplankton populations from diatoms to cyanobacteria as well as higher rates of sedimentation of particulate nutrients as the period of complete water column mixing is reduced with climate warming. The seasonal decline and the distribution of dissolved oxygen in the hypolimnion associated with temperature stratification in Lake Rotoiti appear to be little affected by the earlier formation of stratification in 2002-03, however, most likely because rates of oxygen consumption in the hypolimnion were slightly lower in 2002-03. One conspicuous feature of nutrient comparisons change in nitrate concentrations in surface waters during summer stratification, from consistently close to detection limits (around 5 mg m^{-3}) in 1981-82 and 2002-03, to more consistently around 20-30 around 5 mg m^{-3} in the early 1990s Ammonium concentrations remained low (< 10 mg m^{-3}) in both 1981-82 and 2002-03, though they commonly exceeded nitrate concentrations in summer of 2002-03. Concentrations of FRP in surface waters declined during stratification and varied little between 1981-82 and 2002-03. The net result, however, is that ratios of inorganic N (NO_3 + NH_4): P (FRP) remained very low (< 5) in surface waters of Lake Rotoiti in 2002-03 but underwent periodic large increases in 1981-82 at times of high NO3 concentrations and low FRP. Ratios of TN:TP differed little between 1981-82 and 2002-03 but evidence of nitrogen fixation by phytoplankton populations in mid to late summer of 2002-03 suggests that deficiencies in inorganic N may have been offset by N-fixation by the large population of cyanobacteria. Ratios of inorganic N:P and TN:TP in bottom waters (50-60 m depth) were lower in 2002-03 than in 1981-82, mostly due to higher concentrations of FRP, though NH4 concentrations were also higher in 2002-03 while NO3 concentrations were lower. In summary, the following features would be consistent with a switch from dominance of phytoplankton populations by diatoms to cyanobacteria:

In surface waters (2002-03 vs 1981-82):

- reduced inorganic N:P ratios;
- dominance of ammonium over nitrate concentrations;
- quantitative evidence of N-fixation in combination with large (heterocystous) populations of cyanobacteria;
- longer duration of stratification favouring buoyant species (cyanobacteria) over non-buoyant species (e.g. diatoms);
- low TN:TP ratios relative to requirements for balanced (N and P) growth by phytoplankton.

In bottom waters (also 2002-03 vs 1981-82):

- reduced and very low inorganic N:P ratios influenced strongly by inorganic N depletion through denitrification;
- increases in ammonium and FRP, and decrease in nitrate;
- very low TN:TP ratios (not plotted).

In the early period of 2003, Lake Rotorua underwent two periods of temperature stratification, both of which drove dissolved oxygen levels in bottom waters close to zero. Large increases in concentrations of TP and TN occurred in response to the presumed anoxic release of N and P, resulting in 308 tonnes of TN and 24 tonnes of TP released into the water column over the two events. The TN:TP ratio for the build-up of nutrients during stratification was 12.8:1; higher than for the period of seasonal anoxic release of nutrients from Lake Rotoiti, but within the range where dominance by cyanobacteria might still occur. Internal loading of nutrients plays an important role in the dynamics of both lakes Rotoiti and Rotorua and may have a strong influence on phytoplankton species composition and biomass in both lakes. Lakes that have progressed to the stage where internal loading is a prominent feature of the total nutrient load are not easily or quickly rehabilitated through conventional nutrient management measures and if strongly anoxic, may have low N:P ratios as a result sediment nutrient release and nitrogen loss through denitrification. This process may favour dominance of phytoplankton populations by cyanobacteria.

1. General introduction

This report was initiated in response to a request from the Lakes Water Quality Society to provide a contemporary evaluation of nutrient loads to Lakes Rotorua and Rotoiti, with a focus on loads arising from regeneration of nutrients from bottom sediments (i.e. internal loads). The request from the Lakes Water Quality Society stems from a prolonged bloom of blue-green algae (cyanobacteria) in Lake Rotoiti in the early part of 2003. The bloom resulted in closure of the main body of Lake Rotoiti for c.7 weeks and closure of other bays for a considerably longer period.

Nutrients sources to lakes are both external (catchment) and internal (bottom sediment). No study on the Rotorua lakes has focussed specifically on the relative contributions of nutrients from these two sources in both Lakes Rotorua and Rotoiti, though Rutherford *et al*. (1996, 2003) provide indirect assessments of internal loads that complement their comprehensive analysis of loads from the catchment of Lake Rotorua. Past studies of Lake Rotorua (Hoare, 1987) and Lake Rotoiti (Vincent *et al*. 1984; Gibbs 1992) as well as studies that encompassed both lakes (Fish, 1972; Burns *et al*. 1997) provide a good basis for assessments of nutrient loads from either external or internal sources, however, and Priscu *et al*. (1986) have examined in detail nutrient transformations that occur seasonally in the water column of Lake Rotoiti.

Lakes Rotoiti and Rotorua have undergone long-term deterioration in water quality associated with excess nutrients and eutrophication (Vincent *et al*. 1984; Rutherford *et al*. 1996, 2003; Hamilton, 2003). In Lake Rotorua the deterioration is considered to be associated mostly with nutrients added from direct inputs of wastewater to the lake, particularly between 1982 and 1989 when loads arising from wastewater were high (Rutherford *et al*. 1996). There is evidence more recently, however (Rutherford *et al*. 2003), that suggests that benefits of reductions in nitrate loads arising from removal of treated wastewater discharge to Lake Rotorua have been largely negated by increases other catchment inputs of nitrate. These inputs have been attributed to increasing nitrate levels in groundwater inputs to streams entering Lake Rotorua (Rutherford *et al*. 2003).

Some recent reports suggest that water quality in Lake Rotoiti has either stabilised or improved. For example, in their evaluation of the trophic status of seven of the Rotorua lakes, Burns *et al*. (1997) indicated that "Lakes Rotorua and Rotoiti have become less eutrophic". Gibbons-Davies (2002) also implicated the diversion of sewage effluent from Lake Rotorua as the likely reason for stabilisation of water quality in Lake Rotoiti from 1991-2002, though he noted no change in water quality of Lake Rotorua from 1990 to 2002. These evaluations suggest that the time scales for improvement of water quality in Lakes Rotorua and Rotoiti following diversion of sewage effluent from Lake Rotorua may have been more rapid than those forecast by Rutherford *et al*. (1996).

The model application by Rutherford *et al*. (1996) indicated that there would be a noticeable reduction in lake-water total phosphorus (TP) concentrations within 20 years, but only a slow reduction in TP in the bottom sediments. Two important assumptions in the model forecasts were that phosphorus loads to the lake approximated those of 1972-92 but without sewage inputs, and that a surficial layer of sediment of thickness 0.1 m was active in re-supplying the water column with nutrients. It is not possible to test the accuracy of these assumptions here but there is evidence that TP concentrations in Lake Rotorua tributaries have remained stable to the present point in time although nitrate concentrations are increasing (Rutherford *et al*. 2003). The depth of the sediment layer (0.1 m) in the Rutherford *et al*. (1996) model was based on data from Fish and Andrew (1980) and Lorenzen *et al*. (1996). The long-term recovery of Lake Rotorua would undoubtedly be sensitive to this parameter. However, it may be surmised that it would take many decades (195 years based on 0.1m sediment layer; Rutherford *et al*. 1996) for the sediments to fully recover from additions to the nutrient storage resulting from sewage inputs to the lake.

A study by Priscu *et al*. (1986) identified many of the processes leading to nutrient release from the bottom sediments and transformations in the water column of Lake Rotoiti, and these are directly pertinent to the present study. Dissolved forms of phosphorus and nitrogen are released by bottom sediments under varying conditions of stratification, temperature, oxidation-reduction potential and sediment composition. Release of nitrogen tends to be more difficult to quantify because of different forms of inorganic nitrogen, as well as potential for loss of nitrogen to the atmosphere as nitrous oxide and nitrogen gas (Priscu *et al*. 1986). A conceptual diagram of the processes influencing transformations of nitrogen and phosphorus is shown in Fig. 1. In general, the allophanic bottom sediments, which are rich in iron and manganese, will readily desorb phosphorus and release it to the sediment porewater and then into the water column when exposed to anoxic conditions. These occurrences may be associated first with manganese reduction ($Mn^{4+} \rightarrow Mn^{3+}$, Mn^{2+}), followed by iron reduction ($Fe^{3+} \rightarrow Fe^{2+}$). The latter reduction reaction occurs at lower redox potentials (Mortimer, 1941, 1942, 1971). The release of phosphorus associated with iron reduction may be largest due to higher concentrations of iron in the bottom sediments.

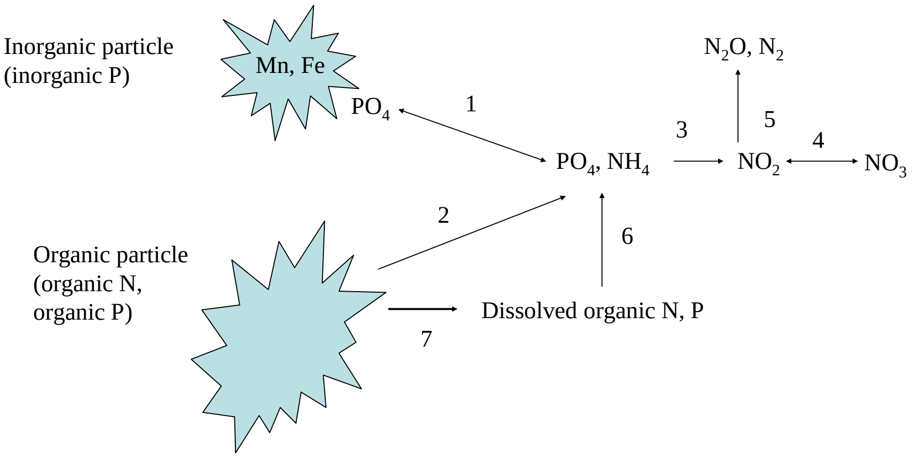


Figure 1. Major transformations of phosphorus and nitrogen in bottom sediments below the photic zone. 1 (right arrow/ left arrow) = adsorption/desorption of phosphorus from inorganic particles containing oxidised forms of iron and manganese, 2, 6, 7 = production of dissolved forms of phosphorus through breakdown of organic matter, 3, 4 (right arrow) = nitrification, 4 (left arrow), 5 = denitrification

While phosphorus release from sediments to the water column may be associated mostly with the effects on inorganic sediments of oxidation-reduction reactions, transformations of nitrogen are linked to particulate organic material rather than particulate inorganic material, though organic materials are often closely associated (coat) inorganic materials. The oxidation-reduction state of the sediments still plays a key role, however, and influences the rate of breakdown of organic nitrogen to ammonium, the subsequent oxidation of ammonium to nitrite and then nitrate, as well as the loss of nitrate through denitrification, which is associated with production of nitrous oxide and nitrogen gas.

Our study has deliberately simplified some of the processes that may act collectively or in isolation and which would otherwise make it difficult to assess nutrient loads from the bottom sediments. Only more detailed modelling studies and measurements will be able to assess what impacts our simplifications may have on the conclusions drawn from our assessments of internal loads. The main simplification is that the build-up of ammonium and total phosphorus in the bottom waters of the two lakes is due solely to release of these nutrients from the bottom sediments. This simplification ignores inputs of nutrients from the Ohau Channel, which at times insert into the bottom waters of Lake Rotoiti (Gibbs, 1992). Complexities of the Ohau Channel inflow, i.e. whether it inserts as an underflow, interflow or surface flow into Lake Rotoiti, have been ignored. We assume that sedimentation plays little role in the nutrient dynamics of bottom waters, since much of the total phosphorus and total nitrogen is in dissolved form (Priscu *et al*. 1986). It is our intention to more accurately quantify the impacts of these assumptions in later studies but the impacts of our simplifications are considered small relative to the signal arising from the net release of dissolved nutrients from the bottom sediments.

2. Lake Rotoiti Internal Nutrient Loads

2.1 Introduction - Lake Rotoiti

The estimates of internal nutrient loads to Lake Rotoiti given below are derived from water column nutrient profiles collected by Environment Bay of Plenty between 1991 and 2003. Data from Priscu *et al*. (1986) have been used to provide a comparison of an historical data set (1981-82) with the contemporary data.

2.2 Methodology - Lake Rotoiti

Data collected between 1991 and 2003 were used for the analyses of patterns of temperature, dissolved oxygen and nutrients presented in the results. Some of the data, particularly between 1994 and 2001, are sparse and this has influenced the accuracy of determinations where there are rapid transitions in nutrient concentrations. Data from two stations (referred to by Environment Bay of Plenty as stations 2 and 3) have been used in the analysis of data.

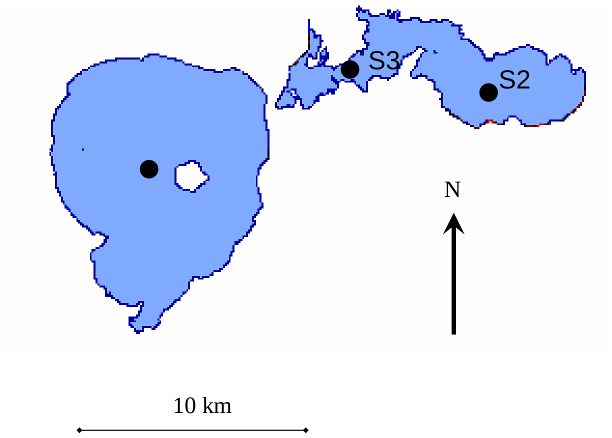


Figure 2 . Position of sampling stations on lakes Rotorua and Rotoiti marked as shaded circles.

2.3 *Results and discussion - Lake Rotoiti*

Temperature

Figure 3 is used here to illustrate the general spatial and temporal extent of temperature profiling at station 2 of Lake Rotoiti from 1990-2003. The data illustrate the repeated annual sequence of stratification in warmer months of the year and mixing in cooler months. The exception is in 2000-01 when no data were collected through the summer stratification period - hence the apparent absence of any thermal gradient. Data extend to c. 70 m depth where the grey shading to represent various water temperatures is replaced by the blackened region that shows absence of data.

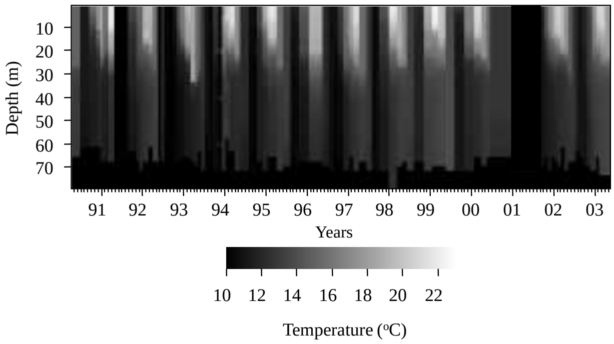


Figure 3. Grey-scale contour plot of temperature based on water column profiles from station 2 on Lake Rotoiti.

Figure 4 presents contour plots of temperature for three different time periods of relatively high-frequency sampling; (a) 1991-94, (b) 2001-03 and (c) 1981-82. Each record illustrates the well-developed seasonal stratification that is typical of a monomictic (single mixing per year) lake. A thermocline (highest temperature gradient with depth) forms around 20-30 m and generally migrates downward through summer. The thermocline disappears and the lake is fully mixed in May-June of each year. The thermocline separates the warm, well mixed surface waters (note the vertical isotherms) from the cooler waters of the hypolimnion.

One of the most conspicuous features of Fig. 4 is the difference in the length of the stratified period between 1981-82 and the other years. Climate is the major driver of variations in stratification and it seems likely that cooler weather and/or windy conditions may have delayed the onset of stratification and advanced destratification in 1981-82. There is conjecture that the Rotorua climate has become milder (warmer) in winter and an analysis of data from Rotorua Airport should be undertaken to establish whether 1981-82 was an exceptionally cold year or part of a more general trend of warming occurring at a regional or larger scale. Anecdotal evidence (N. Miller, pers. com.) suggests that 1981-82 was an exceptionally cool summer as part of a severe El Ninõ.

Most of the isotherms below the thermocline are vertically oriented, supporting (1982) findings that there is convective circulation in the deeper waters of the lake that is driven by geothermal heating, i.e. geothermal sediments heat the water making it buoyant and causing it to rise towards the base of the thermocline. Geothermal heating is confined to a relatively small area of Lake Rotoiti but its effect in mixing the bottom waters is widespread horizontally.

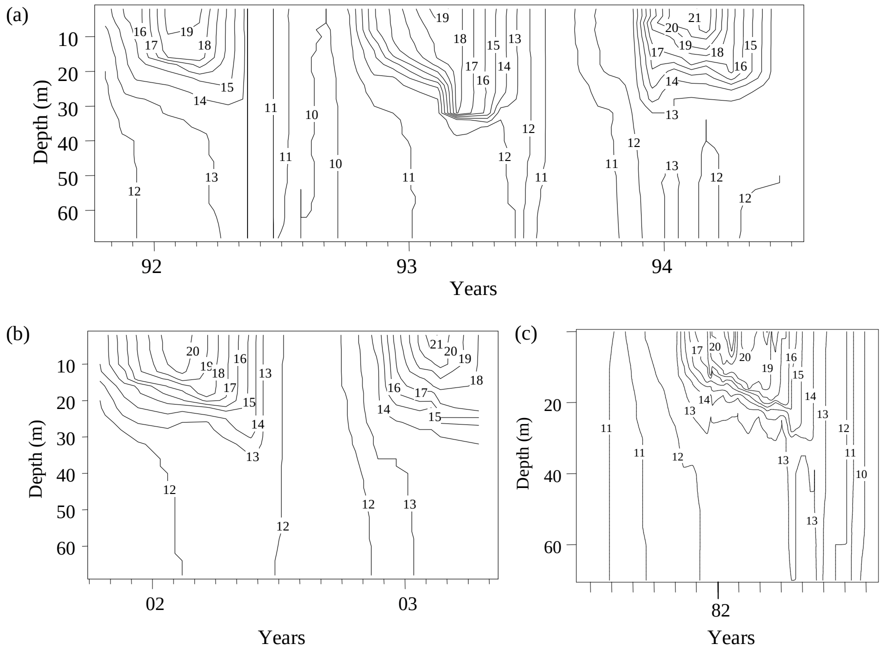


Figure 4. Temperature contours constructed from profiles at station 2 of Lake Rotoiti: (a) 1991-94, (b) 2001-02 and (c) 1981-82.

Figure 5 expands on Fig. 4 to show a direct comparison of temperature and the markedly different durations of stratification between 1981-82 and 2002-03.

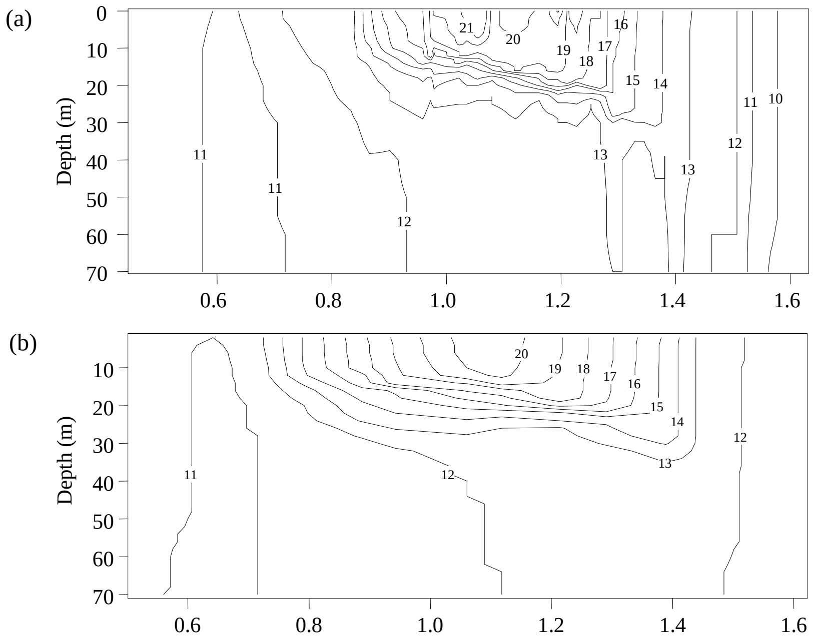


Figure 5. Temperature contours of Lake Rotoiti showing variations in isotherms with depth (vertical axis) and time (horizontal axis), where time is represented as fractions of a year and 1.0 represents the end/beginning of a year. (a) 1981-82 and (b) 2002-03.

The thermocline region in Lake Rotoiti is explored in more detail in Fig. 6 by examining the vertical gradient of temperature, i.e., the rate of change of temperature with depth. Conclusions drawn from this figure need careful consideration as the sampling interval was 1 m in 1981-82 compared with 2 m in 1990-2003. Despite these differences, however, it appears that the thermocline is more sharply defined in 1981-82 and that the seasonal deepening of the thermocline is also more marked than in other years. The first author has observed disruptions to linear time series of seasonal thermocline deepening in other lakes when inflows intrude into the thermocline region, though without more detailed modelling analysis it is not possible to establish what role inflows play in the vertical temperature gradients of Fig. 6.

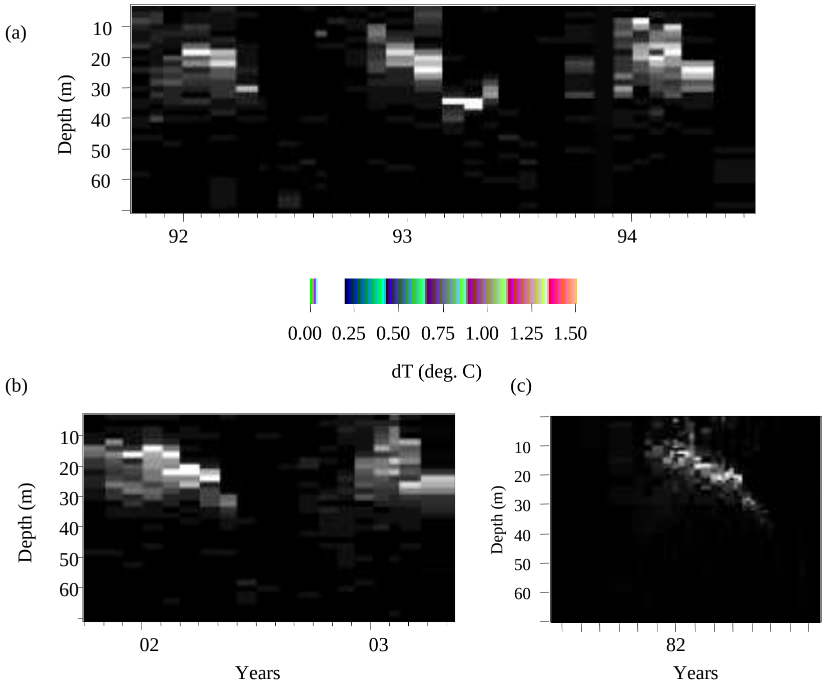


Figure 6. Grey-scale plots of vertical gradients of temperature in Lake Rotoiti for (a) 1991-94, (b) 2001-02 and (c) 1981-82.

2.2.1 Dissolved oxygen

Figure 7 shows the general spatial and temporal extent of profiles taken for dissolved oxygen at station 2 of Lake Rotoiti from 1990-2003. Darkened vertical blocks that extend from the top to the bottom of the water column represent periods when monitoring was too infrequent to enable accurate interpolation of data. Other periods of infrequent monitoring are evident where the width of the vertical profile is large. Despite these shortcomings it is clear that there is an annual sequence of oxygenation and deoxygenation of bottom waters. This sequence is tightly coupled with the temperature regime of stratification and destratification, whereby there is progressive depletion of oxygen in the bottom waters (hypolimnion) below the thermocline in summer, followed by replenishment of oxygen in winter with complete water column mixing.

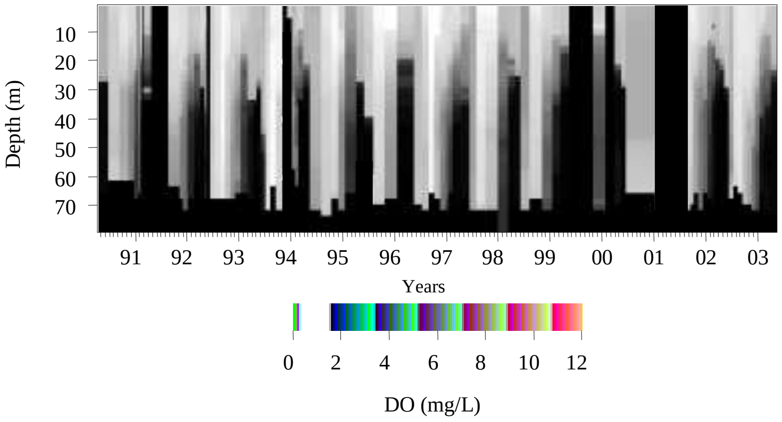


Figure 7. Grey-scale contour plot of dissolved oxygen based on water column profiles from station 2 on Lake Rotoiti.

Three periods with relatively high frequency of dissolved oxygen profiles are shown as contour plots in Fig. 8. Each year dissolved oxygen declines to zero for a period of two (1994) to six (1982) months. Clearly the duration of stratification (see Fig. 4) plays a role in the duration of anoxia (absence of oxygen) in the hypolimnion, but it is interesting to note that the relatively brief period of stratification in 1981-82 was associated with the longest duration of anoxia. The line denoting the concentration of 4 mg L^{-1} intersects the horizontal axis (70 m depth) around or just prior to the change of year in all years except 1981-82 when it intercepts the axis near the beginning of February. The delay in the onset of anoxia in 1981-82 is most likely related to the delay in onset of stratification in this summer. The distribution of dissolved oxygen in 1981-82 and 2002-03 is remarkably similar (Fig. 9), with the major exception being the more gradual progression of anoxia in 2002-03, which can be linked to the earlier onset of stratification and lower oxygen consumption rate. Data for 1981-82 are at higher temporal and spatial resolution and this is reflected in the closeness of contour distributions around 'events' (e.g. winter mixing around time 1.4 corresponding to the beginning of June).

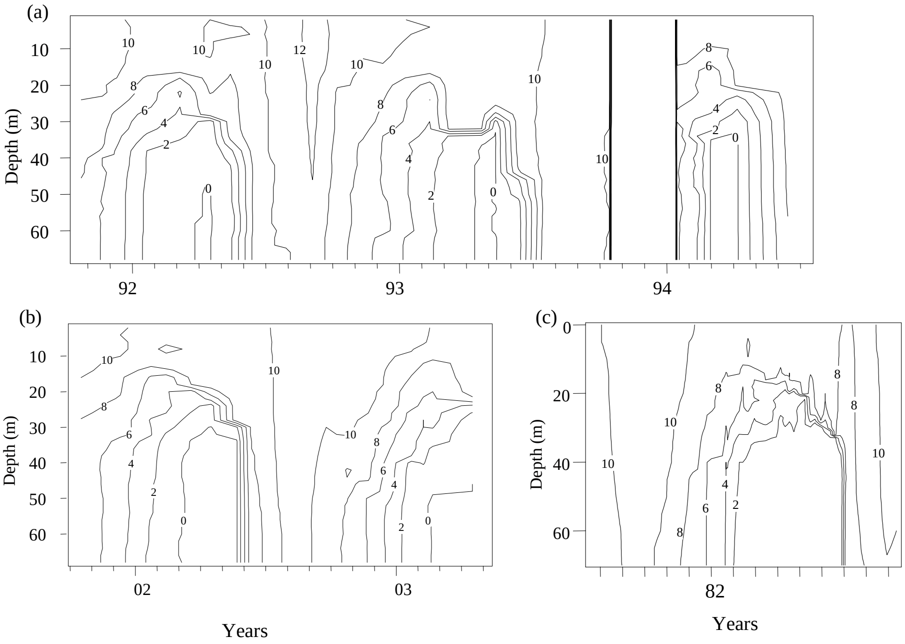


Figure 8. Dissolved oxygen contours constructed from profiles at station 2 of Lake Rotoiti: (a) 1991-94, (b) 2001-02 and (c) 1981-82. The 'zero' contour should not be used for comparative purposes.

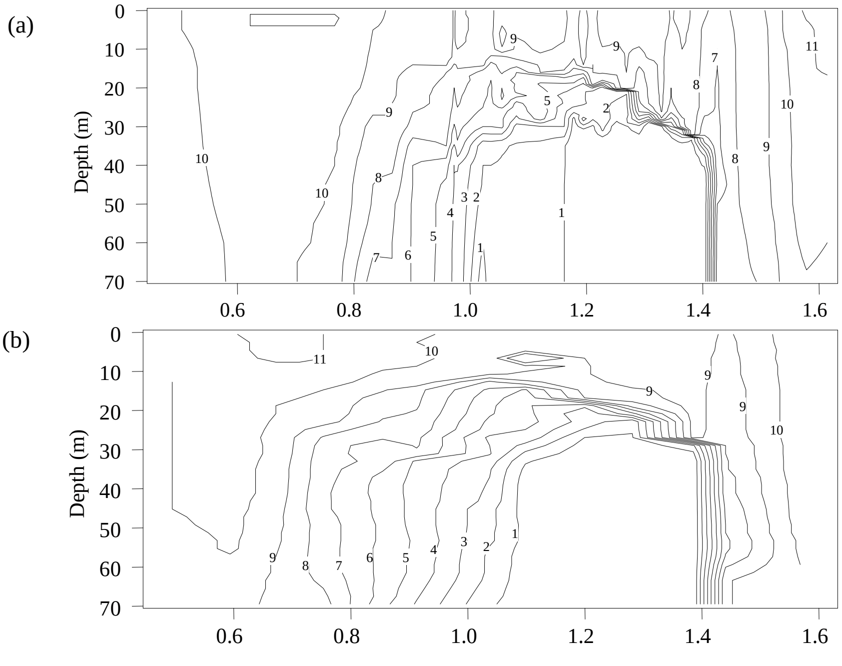


Figure 9. Dissolved oxygen contours of Lake Rotoiti showing variations in concentration with depth (vertical axis) and time (horizontal axis), where time is in fractions of a year (a) 1981-82 and (b) 2002-03.

Interannual variability of the factors driving dissolved oxygen depletion at depth is examined further and for an extended period in Fig. 10. The volumetric hypolimnetic oxygen demand (VHOD) is a measure of the rate at which oxygen is utilised in respiratory processes that are generally associated with aerobic decomposition of organic matter in the hypolimnion. The VHOD may also be affected, for example, by plunging inflows that insert oxygen-rich water into the hypolimnion (Vincent *et al*. 1991) or, in very clear waters, photosynthesis in the hypolimnion, both of which would offset the oxygen deficit to produce artificially low VHOD. The former case is very relevant to Lake Rotoiti, particularly early and late in the stratified period or even on a diurnal basis (Vincent *et al*. 1991) when the Ohau Channel inflow inserts as an underflow or interflow in Lake Rotoiti. In addition, the choice of the period selected for determination of the annual VHOD value is critical. Extending the period of oxygen decline too close to the mixing period or too close to the period when aerobic decomposition starts to become constrained by oxygen availability, produce artificially low values of VHOD. One such example is given in Fig. 11, which shows the effect of extending the VHOD calculation to one sampling period earlier than was actually used for the estimates of Fig. 10. The negative slope of the regression line fit to the decline in oxygen, which equates to the VHOD, is reduced artificially in 2003, from 0.077 to 0.049 g m^{-3} day^{-1}. Low frequency sampling will obviously increase the error associated with estimates of VHOD.

VHOD has clearly increased from the mid-1950s through the early 1970s to the 1980s. Any trend from the early 1980s to present is not clear, however, and the peak VHOD occurred in 1988. Based on historical oxygen profiles, values of VHOD above 0.06 g m^{-3} day^{-1} appear sufficient to induce anoxia at the end of the stratified period.

Figure 10. Volumetric hypolimnetic oxygen deficit based on oxygen profiles at deep sites in Lake Rotoiti.

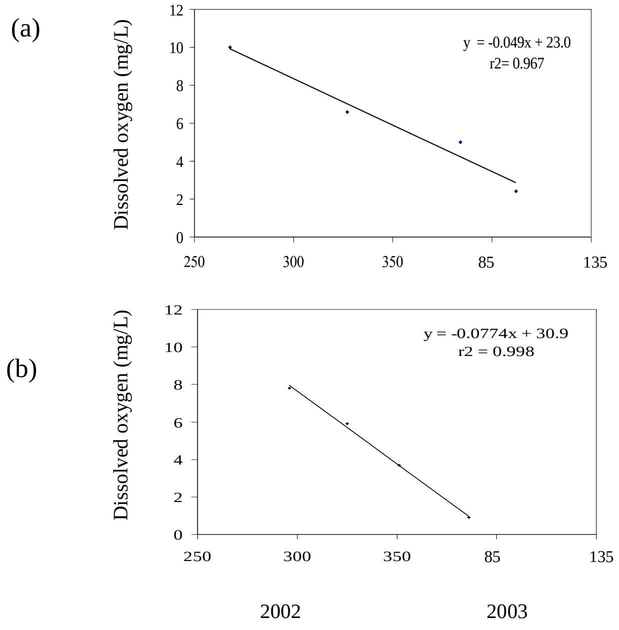


Figure 11. Volumetric hypolimnetic oxygen deficit given by the negative value of the slopes of regression fits to the oxygen data at 60 m depth, showing estimates based on (a) extending the regression to one sampling point earlier than the case in (b).

3.2.3 Time series of nutrients in bottom waters

Nutrient concentrations are highly variable over seasonal time scales in Lake Rotoiti. Under these circumstances and under a low frequency sample regime (sometimes > 8 weeks) it is difficult to derive nutrient budgets and clearly delineate the dominant processes affecting nutrient concentrations. Attempts have been made, however, to estimate internal (bottom sediment-derived) nutrient loads to Lake Rotoiti during the stratified period, when bottom waters are in effect isolated from surface waters, and there are clear signals of nutrient release from the sediments or transformations of nutrients. Some of the caveats applied to calculations of VHOD also apply to nutrient load calculations, however, and insertions of Ohau Channel water into the Lake Rotoiti hypolimnion will affect the accuracy of load estimates.

A relatively coherent time series of data for total phosphorus (TP) at 60 m from 1990-95 is shown in Figure 12a. Interpolation between measurements has been used to provide an uninterrupted sequence of data in both Figure 12a and b, but the data in Figure 12b (note: depth = 50 m) are taken at lower frequency and a period when there were no data (during part of 2000-01) was not part of the interpolated sequence. In addition, measurements were taken at two depths; 40 and 60 m from 1990-95, and provide a useful validation of trends in TP as well as a comparison between the two depths.

There is clear evidence of an increase in total phosphorus (TP), generally commencing just prior to the end of each year, and a decrease near the middle of the year, which coincides closely with the onset of winter mixing. Based on previous records of nutrients in Lake Rotoiti (Priscu *et al*. 1986), it can be assumed that the seasonal increase in total phosphorus is due mostly to an increase in filterable reactive phosphorus (FRP) as a result of a series of redox reactions that are initiated by the decline in oxygen. Desorption of phosphorus from iron and manganese in mineral-bound forms therefore takes place as described in the Introduction (section 2.0).

There is indeed little difference in TP between these the depths of 40 and 60 m (Fig. 12a), though the data at 60 m arguably suggest that peaks in TP may have been slightly larger and of slightly greater duration that at 40 m. Such a trend would be expected on the basis that the progressive anoxia of bottom waters would proceed from the bottom to the top of the thermocline, despite the relatively well mixed nature of this layer.

There is no clear evidence of a trend in TP concentrations since 1990, though levels exceeded 100 mg m^{-3} in 2001-02 and 2002-03. Data in 2001-02 were taken at relatively high frequency (~ 2 weeks) and as a result are likely to have captured more of the inherent variability in TP concentrations, though it is not known whether 2001-02 was anomalous in terms of the relatively high variability.

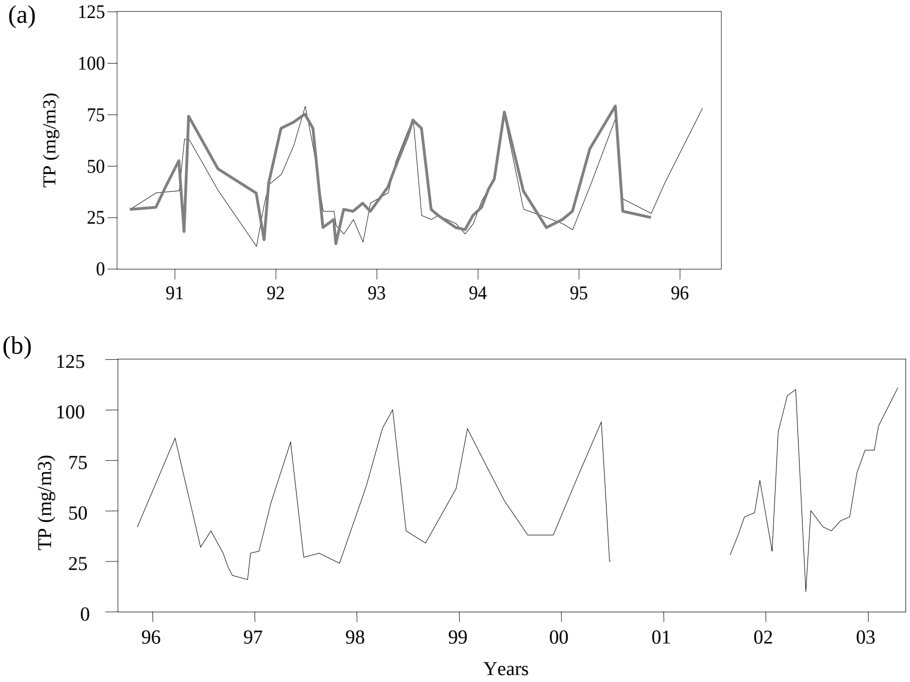


Figure 12. Time series of total phosphorus concentrations in Lake Rotoiti based on interpolation between measurements taken at (a) 60 m (grey line) and 40 m (black line), and (b) 50 m.

Figure 13 shows total nitrogen (TN) concentrations at depths corresponding to those of the TP plots of Figure 12. The relative variation of TN concentrations at 40 and 60 m are slightly greater than those of TP, but the general magnitude and timing of variations are mostly similar. Most peaks in TN occur soon after the beginning of each year, but no conclusions are drawn here about causal factors. The analytical measurement of TN draws together many different species (dissolved organic nitrogen, particulate organic nitrogen, nitrate, nitrite and ammonium), with potential for multiple transformations between these species and other nitrogen species as environmental conditions are altered. Some of these transformations are now explored in more detail in the analysis of the dissolved inorganic species, nitrate and ammonium. By contrast, measurements of dissolved species of phosphorus, specifically the analytical determination of filterable reactive phosphorus (FRP), were not undertaken routinely by Environment Bay of Plenty until recently and comparisons are restricted to 2002-03 and the earlier study by Priscu *et al*. (1986).

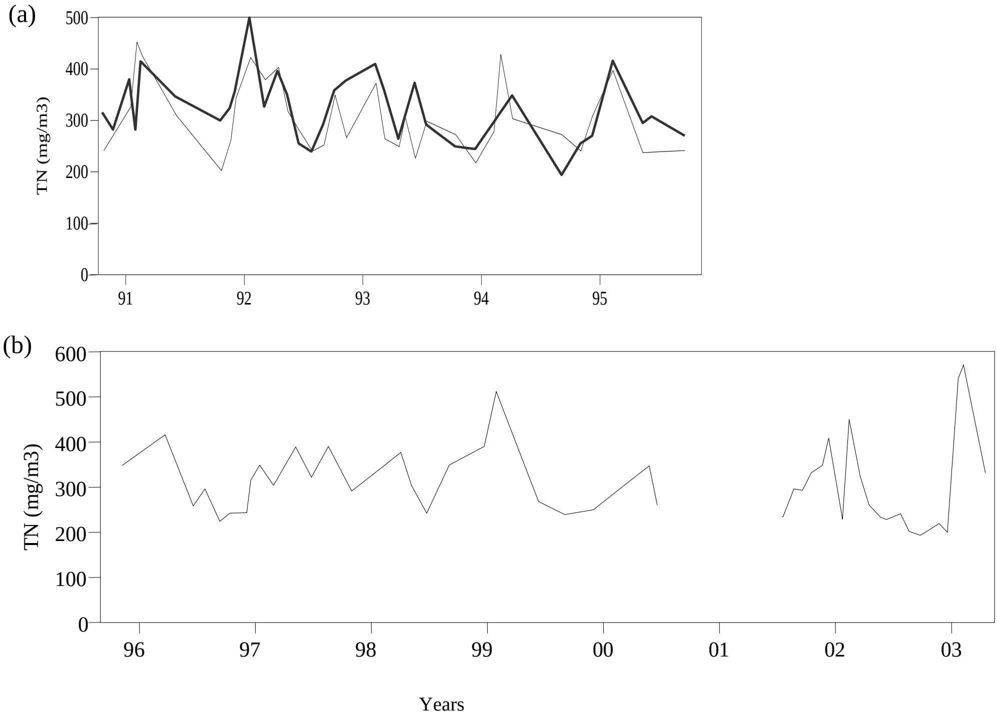


Figure 13. Time series of total nitrogen concentrations in Lake Rotoiti based on interpolation between measurements taken at (a) 60 m (grey line) and 40 m (black line), and (b) 50 m.

The analysis of seasonal changes in nitrate and ammonium concentrations commences with an examination of concentrations at 40 and 60 m between 1990 and 1995. Differences between the two depths are small with the exception of 1991 when there was an abrupt decline in ammonium early in the year and ammonium concentrations did not decrease to <20 mg m^{-3}, a concentration characteristic of the winter mixing phase of other years (Fig. 14).

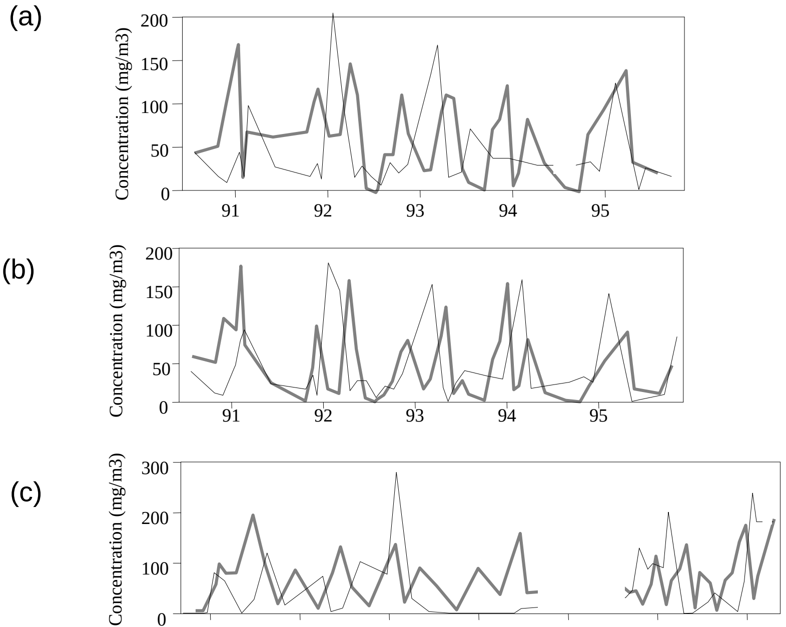


Figure 14. Time series of concentrations of nitrate (black line) and ammonium (grey line) for (a) 60 m and (b) 40 m for 1990-95 and (c) 50 m for 1996-2003. Note the different scale used for (c).

The most prominent feature relating to nitrate at 40 and 60 m (Fig. 14a and b) is the consistent peak in concentration that occurred soon after the beginning of the year. This peak was preceded in each year except 1994-95 by a rapid increase in ammonium concentration, and was followed by second rapid increase in ammonium. Nitrate concentrations were low between these two events, i.e. at the peak of the ammonium concentration. The reasons for the alternating peaks of ammonium and nitrate are presented conceptually for 1992 in Fig. 15. The loss of oxygen from the hypolimnion in mid-summer leads to loss of nitrate as denitrifying bacteria reduce oxygen from nitrate for their energy requirements. The consequences on nitrogen loads of this loss of oxygen are examined further below. The apparent lack of a second peak of ammonium in 1994 is most likely due to the low sampling frequency; no sample coincided with the low-point at the beginning of 1995. The highly dynamic regime of nitrate and ammonium over the summer requires a monitoring frequency of at least monthly or even more frequently to start to match the frequency on which peaks and troughs occur. The sequence of nitrification and denitrification is much less clear for the period 1996-2003. It is not known to what extent this was affected by the reduction in sampling frequency over much of this period compared with 1990-95 or whether there had been a systematic change in nitrogen dynamics in the hypolimnion over the 1990s. Data taken at relatively high frequency in summer 2001-02 would suggest the latter, but the data for 2002-03 suggest a more 'normal' pattern of two ammonium peaks and a single peak of nitrate.

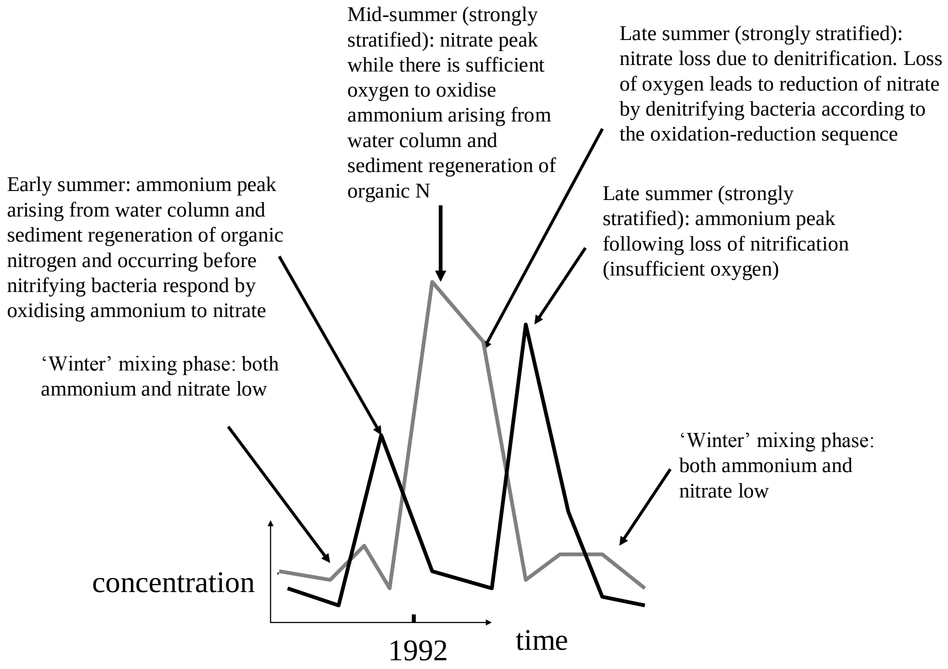


Figure 15. Nitrate and ammonium at 40 m in 1991-92 showing conceptually the processes regulating concentrations of each nutrient.

Figures 16 and 17 show differences in inorganic nitrogen concentrations between sites (stations 2 and 3) and between depths, respectively. Some of the data from Fig. 14 have been repeated, but only as an aid to comparisons between depths or stations.

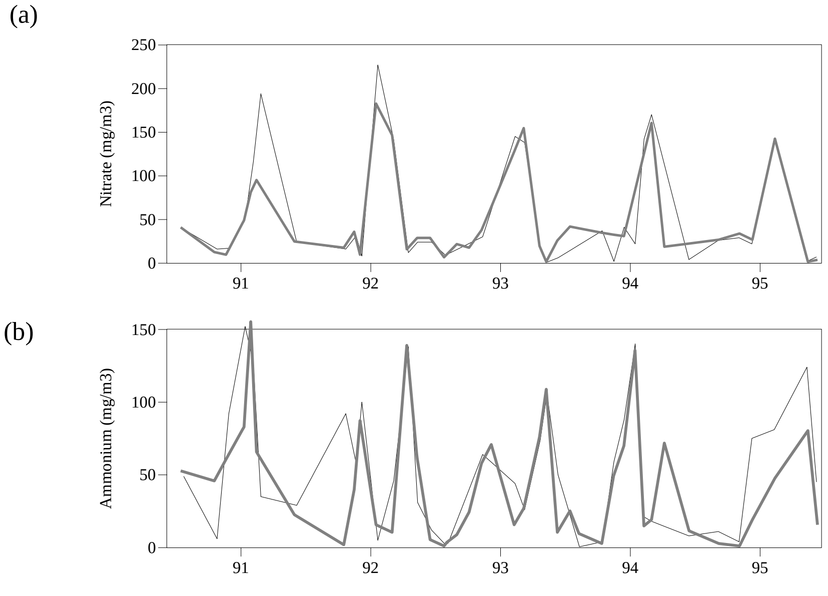


Figure 16. (a) Nitrate and (b) ammonium concentrations for 1990-95 at station 3 at 40 m depth (black line) and station 2 at depths ranging from 40-52 m (grey line).

Nitrate concentrations tend to be very similar between the two sites (note that there may be small anomalies in the depth at which the sample is taken at the two stations) though peaks in the deeper (central lake) station are higher in 1991 and 1992. The relative variation in ammonium concentrations between the two sites is greater than nitrate and ammonium concentrations at station 3 do not follow as clearly the regular twice-yearly peaks that are evident at the deeper site, station 2. Part of this variation may reflect greater proximity to the Ohau Channel and more variable concentrations of ammonium that arise from this inflow. There are relatively small variations in nitrate and ammonium with depth. Variations are greater for ammonium and concentrations at 60 m never declined to low levels in 1991 through winter even though the hypolimnion appeared to be fairly homogeneous based on temperature variations (Fig. 4a). Ammonium peaks at 60 m are mostly slightly higher than at 40 m perhaps reflecting greater organic nitrogen accumulation deeper in the water column, but differences are small.

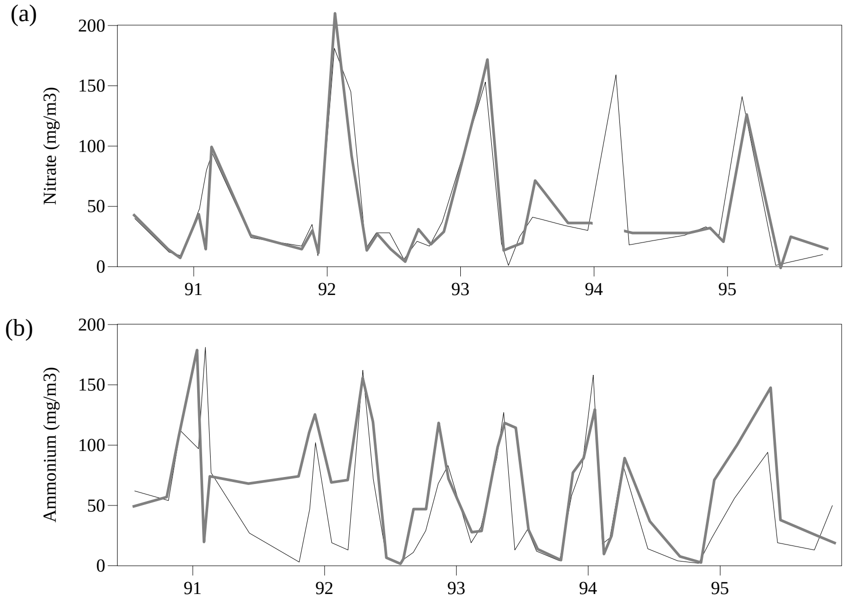
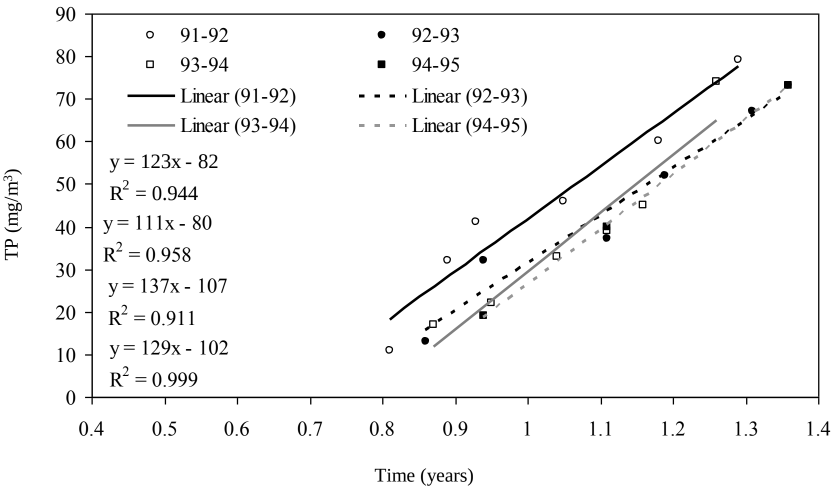


Figure 17. (a) Nitrate and (b) ammonium concentrations for 1990-95 at station 2 at 40 m (black line) and 60 m depth (grey line).

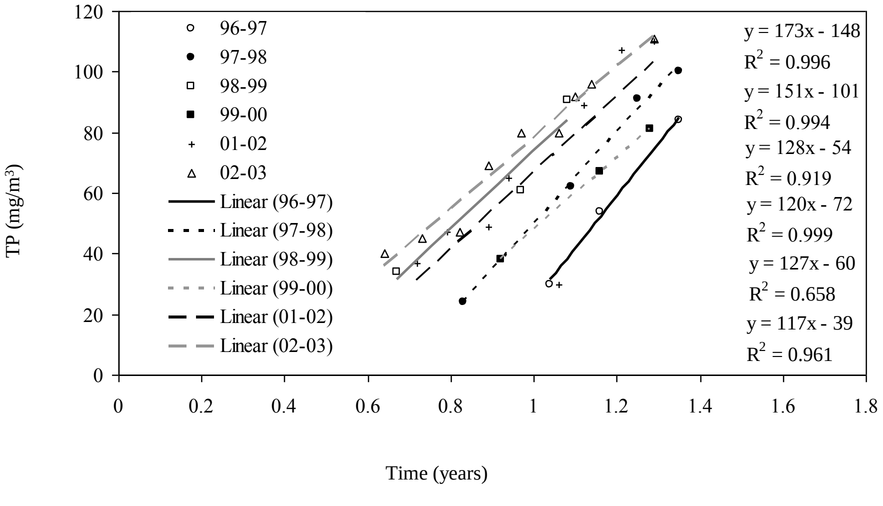
2.2.4 Time series of phosphorus in bottom waters during stratification

Because of the linearity of seasonal increases in concentrations of TP, it is possible to plot regression relationships that relate the increases to time of year (where the start of the first year is assigned a value of zero and the start of the second year is assigned 1). These regression relationships have not been tested for level of significance and in some cases there may be only a few points that are used to derive a relationship. The relationships should therefore be used as indicative only and not necessarily taken to be statistically significant. In addition, the selection of data is important. The first value in the annual time series was taken where the water column was clearly stratified but levels of dissolved oxygen were still high (> 6 mg L^{-1}) and there was no evidence that TP had commenced its seasonal increase, though this point did not deviate greatly from the regression line fit. The selection of the end-point is also important; low frequency sampling may not capture the true extent of the increase and therefore the peak concentrations shown in Fig. 18 may be conservative, though not necessarily affecting the slope of the regression equation. Nevertheless the relationships presented in Fig. 18 show coherent trends of seasonal increases in TP, which are analysed below.

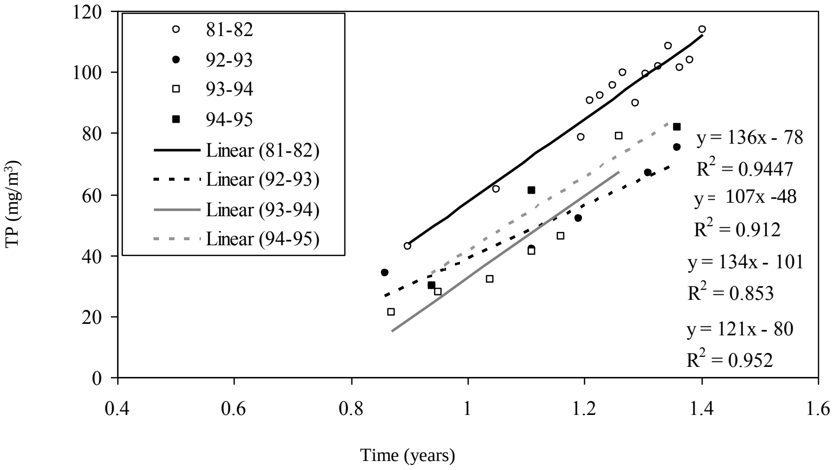
(a)



(b)



(c)



(d)

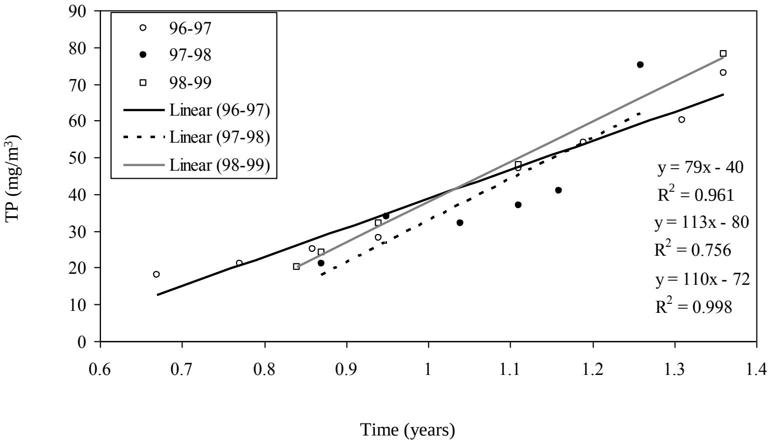


Figure 18. Time series of total phosphorus concentrations for the period of stratification when there was a general trend of increasing concentrations in bottom waters. Time is plotted commencing with the first year as zero and the second year as one. The order of the regression relationships for seasonal increases in TP on each plot corresponds sequentially with the years that are given. (a) 40 m in 1991-95, (b) around 50 m in 1996-2003, (c) 60 m in 1992-95, all from station 2, and (d) from around 50 m from station 3.

Table 1 summarises the regression line slopes presented in Fig. 18. Regression slopes are very consistent for data from 40 m at station 2 in 1991-95, but are more scattered for the remaining periods. Two values < 110 mg m^{-3} yr^{-1} stand out as exceptionally low and in both cases there may have been a strong influence of the selection of the start period for the regression. For example the 1992-93 regression slope at a depth of 60 m at station 2 changed from 107 to 130 mg m^{-3} yr^{-1} by removing the first sample point. Similarly the 1996-97 regression slope at a depth of ~50 m at station 3 changed from 79 to 98 mg m^{-3} yr^{-1} by removing the first three sample points. In both cases the removal of these points could be justified by lack of clear evidence that TP concentrations had begun to increase in the subsequent sample.

Regression slopes at station 3 show greater variability than those obtained for station 2 in the corresponding period. It may be speculated that the increased variability is due to proximity of station 3 to the Ohau Channel and the associated fluctuations produced by oxygen incursions as inter- and underflows pass this station.

Table 1. Slopes of regressions of total phosphorus concentrations on time for different stations (S2 and S3), depths and years in Lake Rotoiti. Values are in mg m^{-3} yr^{-1} with values in parentheses converted to mg m^{-3} day^{-1}.

Year	**40 m, S2**	**50 m, S2**	**~ 50 m, S3**	**60 m, S2**
1981-82	-	-	-	136 (0.373)
1991-92	123 (0.337)	-	-	-
1992-93	111 (0.304)	-	-	87 (0.238)
1993-94	137 (0.375)	-	-	134 (0.367)
1994-95	129 (0.353)	-	-	121 (0.332)
1995-96	-	-	-	-
1996-97	-	173 (0.474)	79 (0.216)	-
1997-98	-	151 (0.414)	113 (0.310)	-
1998-99	-	128 (0.351)	110 (0.301)	-
1999-2000	-	120 (0.329)	-	-
2000-01	-	-	-	-
2001-02	-	127 (0.348)	-	-
2002-03	-	117 (0.321)	-	-

It may be surmised from Table 1 that there do not appear to be consistent changes in the rate of increase of total phosphorus in the bottom waters in the presence of stratification. The regression slope at 60 m for 1981-82 falls well within the range of values for subsequent years and, by implication, suggests that there has not been a consistent change in rates at which phosphorus is released from the bottom sediments during stratification.

By choosing the first and the last points for each of the regressions of Fig. 18 and with knowledge of the volume of the lake below 30 m, it is possible to make a rough approximation of the increase in mass of TP in the hypolimnion over the stratified period. It should be emphasised that there are many approximations and assumptions involved in this approach; that the first and last points of the regression truly represent the start and end points respectively, for the mass calculation, and that the volume under consideration (i.e. depth < 30 m) remains constant, both between years and within seasons. The approximation of the increase in mass of TP in the hypolimnion over the stratified period is given in Table 2 with only station 2 used for the calculation. Once again the identification of consistent trends is difficult given the assumptions made and the variability of calculated mass for adjacent years. There is perhaps a hint that the mass (i.e. internal loading of phosphorus) may have been relatively low through the mid-1990s, but that it has since increased, with three of the largest seasonal loads occurring in the past 6 years. Selection of three widely spaced intervals in which there was high frequency of data collection (1981-82, 1991-92 and 2002-03), however, yields a range of TP mass of only 19.1-19.9 tonnes; once again the time scales of change in internal loading appear to be slow and may be influenced as much by the timing of stratification and mixing in Lake Rotoiti, as by any inherent changes in the rate at which phosphorus is released from the bottom sediments.

Table 2. Increase in mass of TP in the hypolimnion over the stratified period based on a hypolimnion depth of 30 m (equivalent water volume of 280,360,000 m^3) and start (start TP) and end (end TP) points for the calculations of mass taken from start and end points of the regressions for station 2 in Figure 18.

Year	***Depth (m)***	***Start TP (mg m^{-3})***	***End TP (mg m^{-3})***	***Mass (tonnes)***
1981-82	60	43	114	19.9
1991-92	40	11	79	19.1
1992-93	40	13	73	16.8
	60	21	79	16.3
1993-94	40	17	74	16.0
	60	30	82	14.6
1994-95	40	19	73	15.1
1996-97	50	30	84	15.1
1997-98	50	24	100	21.3
1998-99	50	34	91	16.0
1999-2000	50	38	94	15.7
2001-02	50	37	110	20.5
2002-03	50	40	111	19.9

Changes in dissolved inorganic nitrogen concentrations lend themselves to a similar analysis to those given above for TP. The result of plotting the first rise in ammonium concentrations at the onset of stratification (refer to Fig. 15) is shown in Fig. 19. The slopes of the regressions are highly variable (175-768 mg m^{-3} yr^{-1}) and would not add credence to any attempts to derive internal loading calculations associated with regeneration of ammonium from organic nitrogen in the sediments or the water column. Similarly the slopes of regressions for periods when nitrification (ammonium → nitrate) was dominant (Fig. 20) were highly variable (334-1433 mg m^{-3} yr^{-1}). No further analysis of a mass budget for inorganic nitrogen has been undertaken for these phases.

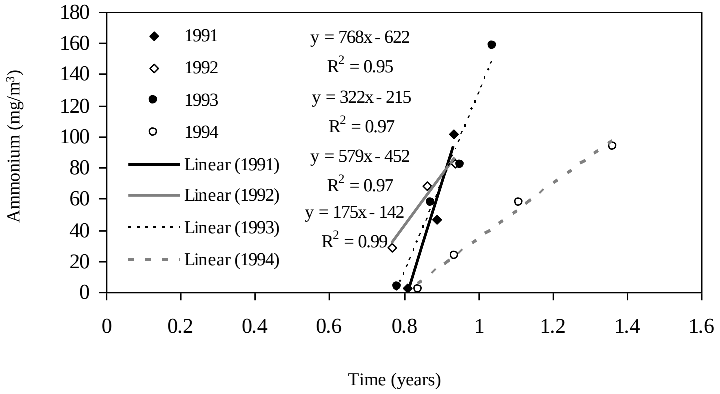


Figure 19. Ammonium concentrations at a depth of 40 m (station 2) corresponding to the period when ammonium initially increases at the onset of seasonal water column stratification, 1991-92 to 1994-95. Time commences with the first year zero. Order of regressions corresponds sequentially with the years that given.

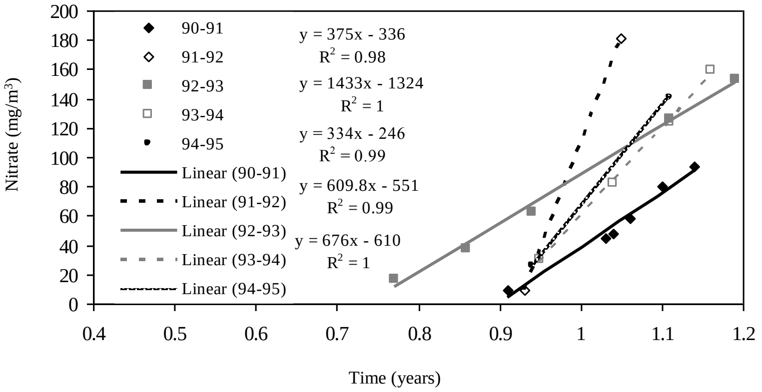


Figure 20. Nitrate concentrations at a depth of 40 m (station 2) corresponding to the mid-summer when nitrate concentrations increase and ammonium concentrations decline rapidly, 1991-92 to 1994-95. See Fig. 19 for further explanations of figure.

The final phase of increase of ammonium during the stratified period also produced highly variable regression slopes for observed increases as a function of time (Fig. 21). While some of the reasons for the high variability of Figs 19-21 are not well understood, it is clear that sampling frequency does not resolve adequately the rapid variations in inorganic N concentration that occur with changes in dissolved oxygen (and associated oxidation-reduction potential). In some cases just two points are available (Fig. 22) and hence regression line fits to these data are not truly applicable (yielding r^2 values of 1.0). When examining the very rapid decline in nitrate associated with seasonal dominance of denitrification (typically one month, mid to late summer), not more than 1-2 points were interspersed between the high and the low points of nitrate concentrations. From Figure 22 it is clear that application of regression relationships is inappropriate for estimating rates of denitrification, so mass of nitrate loss has been estimated from start and end points of the seasonal nitrate decline. Unlike previous calculations of changes in nitrate and ammonium (Figs 19-20), there can be some confidence that just one biochemical process (denitrification) acts to change nitrate concentrations when oxygen is absent from the hypolimnion (i.e. nitrification and denitrification act simultaneously in the presence of oxygen).

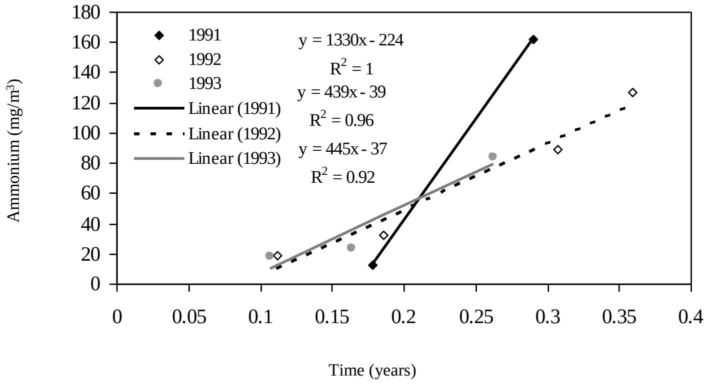


Figure 21. Ammonium concentrations at a depth of 40 m (station 2) corresponding to the late summer when nitrate disappears and ammonium increases rapidly, 1991-92 to 1993-94. See Fig. 19 for further explanations of figure.

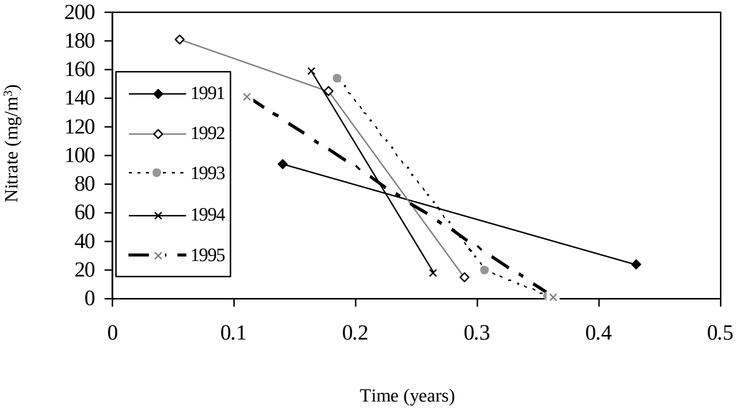


Figure 22. Nitrate concentrations at a depth of 40 m (station 2) corresponding to mid to late summer when nitrate disappears and ammonium increases rapidly, 1991 to 1995. See Fig. 19 for further explanations of figure.

It is highly likely that nitrate is removed entirely from the hypolimnion, though capturing the exact time when nitrate is removed as well as the analytical determination of 'zero' nitrate concentration makes capturing this phenomenon difficult. Table 3 gives estimates of the mass of nitrate lost through denitrification for years when it was possible to discern clearly both a mid-summer peak in nitrate concentration and a late summer minimum. The mass of nitrate represents a significant proportion of the total nitrogen budget for the lake based on calculations of loads from the Ohau Channel (to be presented as Part II of this study). There appear to be coherent variations between years, with relatively low loss rates of nitrate in the early 1990s and high losses in 1982 and in the past two years, 2002 and 2003. The variations are not likely to be attributable to differences in depth between years since concentrations of nitrate vary little over the depth range used in Table 3 (Fig. 17a).

Table 3. Decrease in mass of nitrate in the hypolimnion over the stratified period based on a hypolimnion depth of 30 m (equivalent water volume of 280,360,000 m^3) and start and end points for the calculations of mass taken from the mid-summer peak in nitrate and the late summer minimum, respectively (Mass 1). A second calculation of mass of nitrate lost (Mass 2) is also used based on an assumption of complete disappearance of nitrate from the hypolimnion.

Year	***Depth (m)***	***Start NO3 (mg m^{-3})***	***End NO3 (mg m^{-3})***	***Mass 1 (tonnes)***	***Mass 2 (tonnes)***
1982	60	298	0	83	83
1992	40	181	15	47	51
1993	40	153	1	43	43
1994	40	159	18	40	45
1995	40	141	1	39	40
2002	50	201	1	56	56
2003	50	239	-	-	67

3.2.5 Interannual comparisons of time series: 1981-82 versus contemporary data

While there is some evidence that oxygen levels have decreased and that rates of oxygen consumption have increased in the period from the mid-1950s to present, data presented above show little change in oxygen or nutrient levels in Lake Rotoiti since 1990. A more detailed comparison of 1981-82 nutrient data against contemporary data has therefore been undertaken below. Data used for the comparisons are from either 60 m (1981-82 and 1992-93) or 50 m (2002-03) and correspond to periods of relatively high frequency data collection. Total phosphorus concentrations at depth (50 or 60 m) are consistently higher in 2002-03 than in 1981-82 and even more so compared with 1992-93 (Fig. 23a). By contrast any consistent interannual variation in TN concentrations cannot be discerned from the inherent variation within each season (Fig. 23b). A further examination of phosphorus at depth has been carried out using filterable reactive phosphorus (FRP) concentrations for the only two period when this measurement was carried out (Fig. 24).

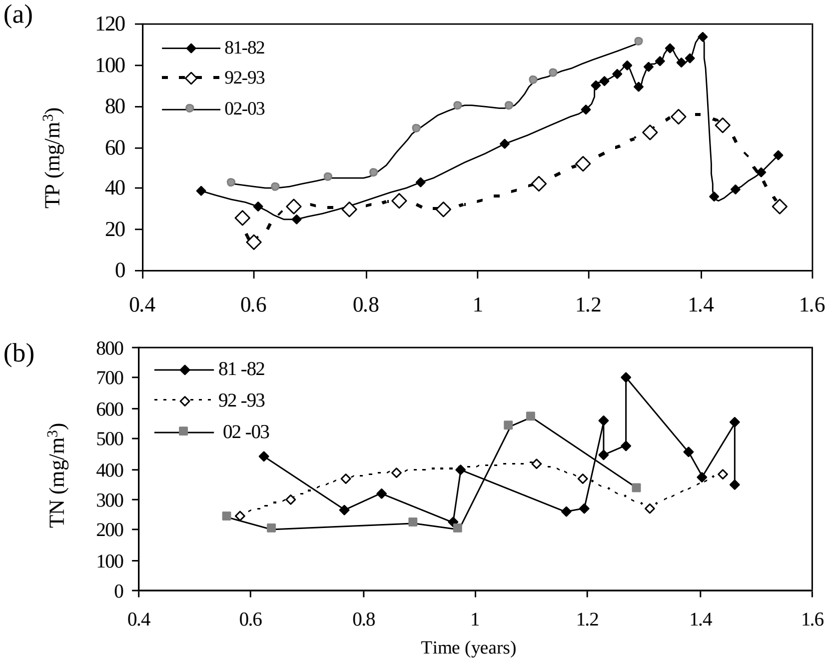


Figure 23. Interannual comparisons of total nitrogen at 50 m (2002-03) or 60 m (1981-82 and 1992-03), with time represented as fractions of a year commencing from the first year denoted in the legend.

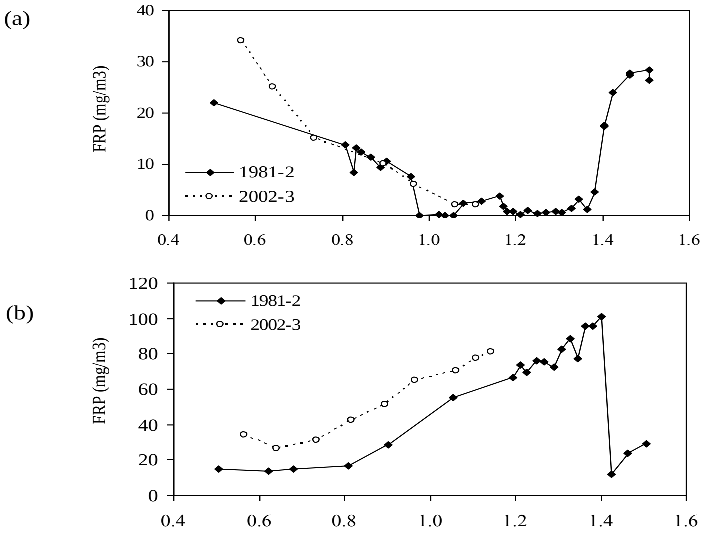


Figure 24. Interannual comparisons of filterable reactive phosphorus at (a) the water surface and (b) 50 m (2002-03) or 60 m (1981-82 and 1992-03), with time represented as fractions of a year commencing from the first year denoted in the legend.

FRP concentrations at 50 m are consistently higher in 2002-03 than in 1981-82 (Fig. 24b) though concentrations at the water surface are very similar. It is not known at this stage, however, whether concentrations in late summer of 2002-03 dropped to the very low values observed at the corresponding time in 1981-82. There is certainly an indication from Fig. 24a that FRP concentrations were higher in the winter mixing phase of 2002 than in 1981 for both surface and bottom waters.

Figure 25 shows total phosphorus and total nitrogen concentrations at the water surface for 1981-82, 1992-93 and 2002-03. Once again it is difficult to discern interannual variations from those occurring between adjacent samples, though there is evidently a period of relatively high levels of total nitrogen near the end of 2002 and the beginning of 2003. Hamilton and Silvester (unpubl. data) found direct evidence of nitrogen fixation by the prevailing cyanobacterial population (*Anabaena planktonica*) early in 2003, both in Okawa Bay and the central basin, and one could hypothesise that correspondingly high levels of TN were sourced atmospherically as nitrogen gas dissolved in the water column.

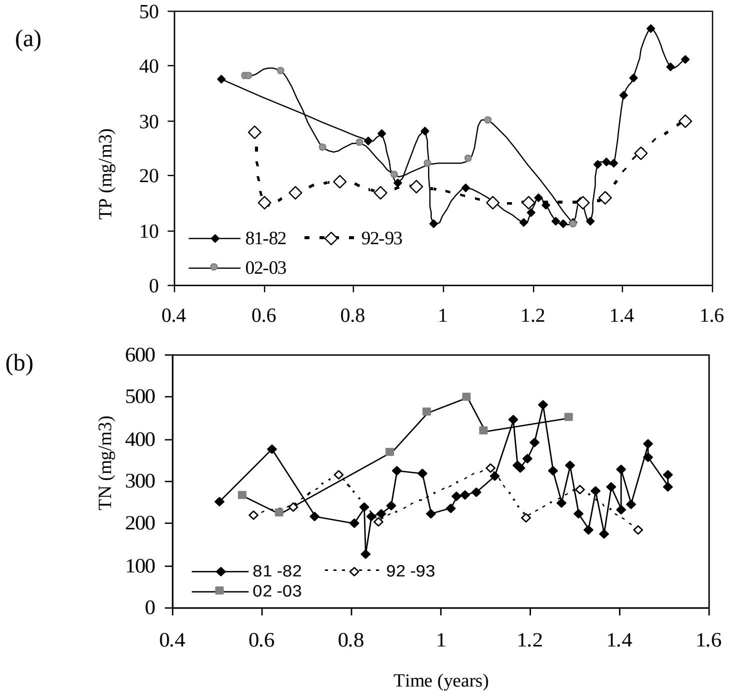


Figure 25. Interannual comparisons of (a) total phosphorus and (b) total nitrogen at the water surface for 1981-82, 1992-93 and 2002-03, with time represented as fractions of a year commencing from the first year denoted in the legend.

The interannual comparisons of phosphorus and nitrogen shown above (Figs 23-25) indicate that there may perhaps have been more available phosphorus to support phytoplankton production in 2002-03 than in the other two years. By contrast, however, there is clearly more inorganic nitrogen in 1992-93 than in 1981-82 or 2002-03 (Figs. 26 and 27). The reason why moderate levels of nitrate (i.e. not including nitrite) should persist in surface waters during 1992-93 (Fig. 26b) is not immediately clear but certainly suggests that inorganic nitrogen concentrations may have been sufficiently high to have satisfied phytoplankton growth requirements in that summer. Nitrate concentrations were higher in bottom waters in 1992-93 than in 2002-03 while conversely, ammonium concentrations were higher in 2002-03. It is not known whether these variations are significant or whether they have an impact on the availability of nitrogen for phytoplankton production. A preliminary conclusion is that there was greater regeneration of ammonium from mineralisation of organic nitrogen in 2002-03, and that nitrification may not have removed the additional ammonium as effectively. Whether this was a result of additional organic N supply or reduced oxygen available for nitrification is not known.

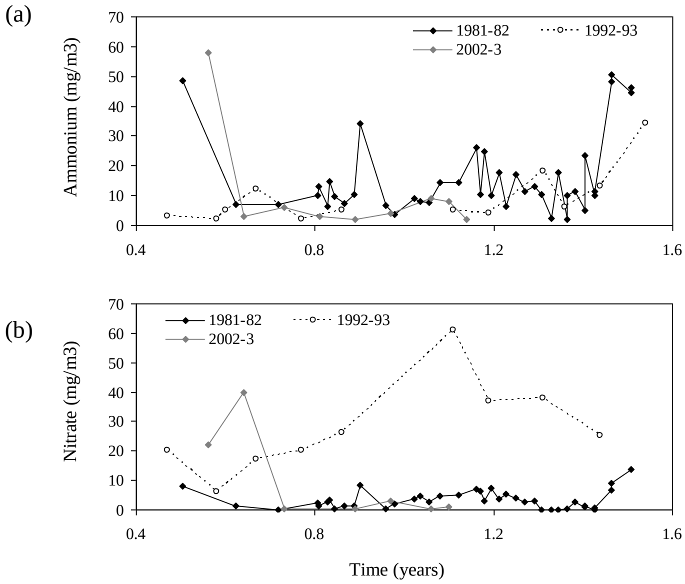


Figure 26. Interannual comparisons of (a) ammonium and (b) nitrate at the water surface for 1981-82, 1992-93 and 2002-03, with time represented as fractions of a year commencing from the first year denoted in the legend. An outlier of ammonium in 2002-03 that greatly exceeded any other recorded values has been removed.

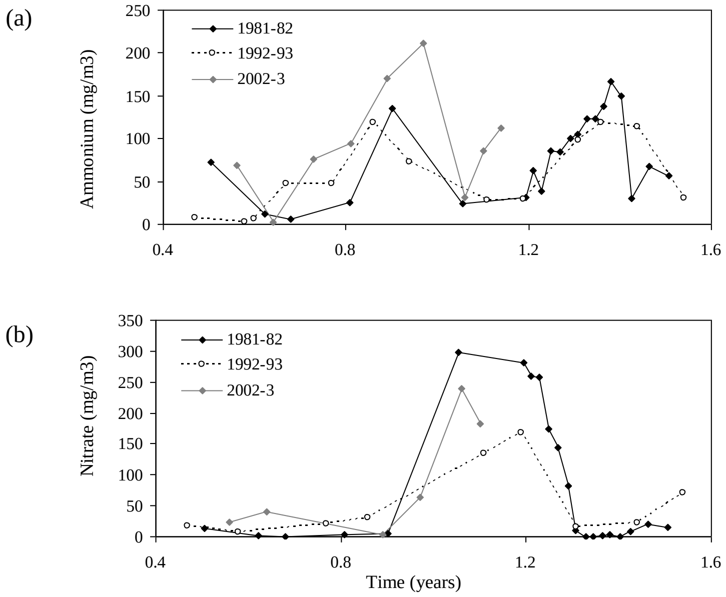


Figure 27. Interannual comparisons of (a) ammonium and (b) nitrate at 60 m (1981-82 and 1992-93) and at 50 m (2002-03), with time represented as fractions of a year commencing from the first year denoted in the legend.

3.2.6 Interannual comparisons of nutrient ratios: 1981-82 versus contemporary data

Ratios of total nitrogen to total phosphorus have previously been used to provide an indication of whether nitrogen or phosphorus might limit phytoplankton production. Smith (1983) suggested that a mass ratio of 13:1 is a critical point; ratios < 13:1 are likely to results in a significant competitive advantage for cyanobacteria which generally compete more effectively for N than other phytoplankton when concentrations of dissolved inorganic N are low and indeed some genera such as *Anabaena* may fix nitrogen under these conditions. Smith *et al*. (1995) and Havens *et al*. (2003) have used N:P ratios for assessing transitions in phytoplankton populations, with a very clear demonstration of their relationship with cyanobacterial biomass and dominance, though there is a suggestion that this ratio may be lower than the value of 13:1 originally suggested by Smith (1981).

Ratios of N:P are first examined in Lake Rotoiti using bottom waters. The results shown in Fig. 28 provide strong evidence of consistent differences between 1981-82 and 2002-03. Ratios of dissolved inorganic N (DIN: NO_3-N +NH_4-N) to P (FRP) may perhaps be higher during the winter mixing phase of 2002-03 though this outcome may be strongly biased by the small number of samples taken through winter of 2002. However, samples taken in the summer stratification period show consistently that DIN:FRP ratios are lower in 2002-03 than in 1981-82. One of the main drivers of this result is higher levels of FRP in bottom waters in 2002-03 compared with 1981-82 (Fig. 24b). By contrast, TN:TP ratios are higher in all samples in 1981-82 than in 2002-03, though again there are relatively few samples in 2002-03 that allow for comparison. The ratios of DIN:FRP illustrate clearly some of the important processes regulating inorganic nitrogen concentrations in bottom waters. The rapid increase in DIN:FRP around 0.8 years (i.e. Sept.-Nov.) corresponds to a period of ammonium regeneration at the onset of stratification. The rapid decrease in the ratio around 1.2 years (i.e. Feb.-Apr.) corresponds to a period of denitrification when nitrate is converted to gaseous form below the thermocline. Ratios of DIN:FRP and TN:TP are low by comparison to the critical ratio given by Smith (1981) for the transition between mixed phytoplankton populations and cyanobacteria.

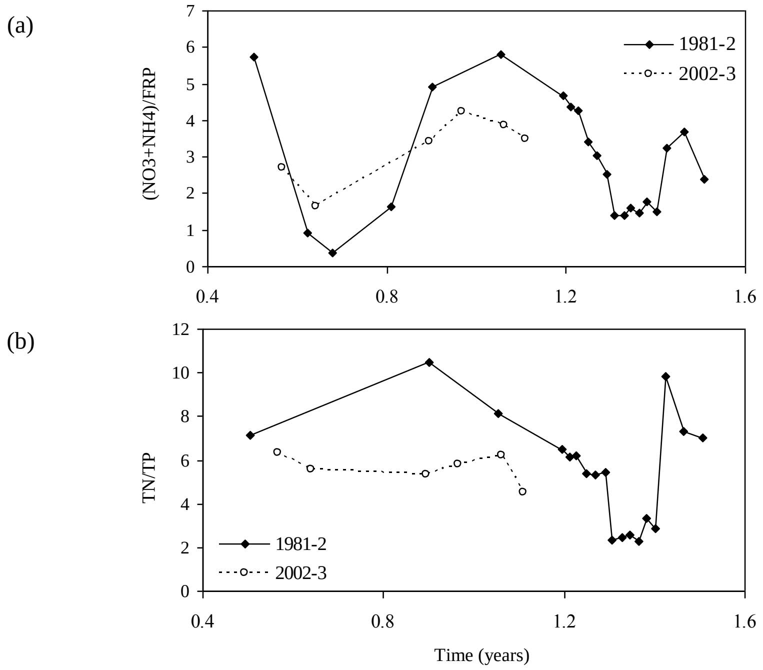


Figure 28. Comparisons of ratios of (a) inorganic N:P and (b) TN:TP for 60 m (1981-82) and 50 m (2002-03), with time represented as fractions of a year commencing from the first year denoted in the legend.

Surface water ratios of N:P are of greater significance to phytoplankton production and considerations of cyanobacterial dominance than bottom water ratios, though one should not underestimate the impact of inorganic N and P transitions in bottom waters for the resupply of nutrients for primary production. Ratios of DIN:FRP in surface waters were extremely low (< 5) in 2002-03 but were punctuated by large spikes in 1981-82, when FRP levels were extremely low (i.e. the denominator approaches zero; but values were expressed to lower values that for contemporary measurements) making the ratio extremely sensitive to any changes in either FRP or DIN concentrations. Total N: total P ratios are remarkably similar in 1981-82 and 2002-03, despite the changes in inorganic nutrient availability and ratios. It seems plausible that in 2002-03 nitrogen fixation by the dominant phytoplankton population, the cyanobacterium *Anabaena planktonica*, may have enabled this species to overcome the short supply of inorganic N and, because of its dominance in the population, may have maintained ratios of N:P close to those required for balanced growth (Redfield ratio; 7:1). There is a strong suggestion of phosphorus limitation of phytoplankton in 1991-92 and nitrogen limitation in 2002-03 which may have been at least partly negated in the latter period by nitrogen fixation by *Anabaena planktonica*, when this alga dominated the phytoplankton assemblage.

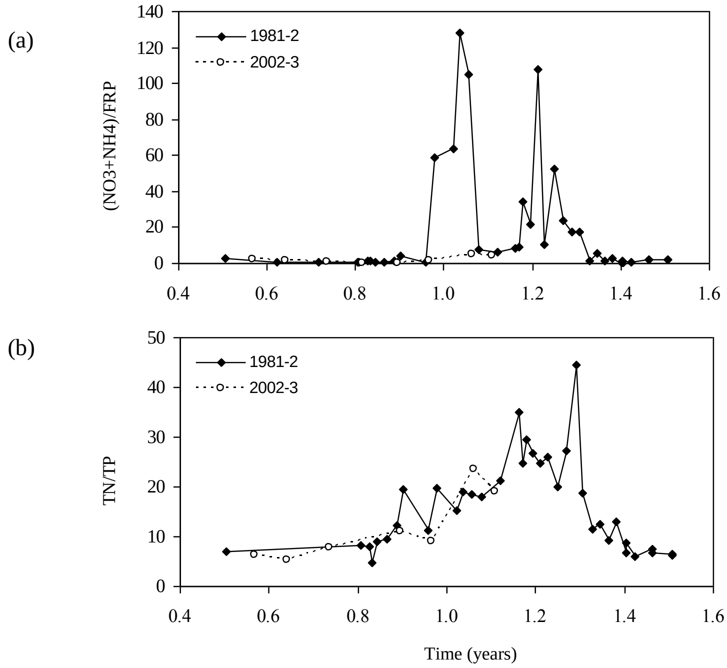


Figure 29. Comparisons of ratios of (a) inorganic N:P and (b) TN:TP for surface waters in 1981-82 and 2002-03, with time represented as fractions of a year commencing from the first year denoted in the legend.

3. Lake Rotorua Internal Nutrient Loads

3.1 Introduction - Lake Rotorua

Estimates of internal nutrient loads for Lake Rotorua are preliminary values for 2002-03 from David Burger's Ph.D. study, which is funded through a Hilary Jolly Scholarship at the University of Waikato. At this stage we request that values are treated as 'preliminary' as Burger intends to use a component of these data in the production of scientific papers and in his Ph.D. thesis.

3.2 Methods - Lake Rotorua

One central site (depth 20 m) at Lake Rotorua was sampled intensively over the summer of 2002-2003 to determine stratification-deoxygenation regimes and internal nutrient loading to Lake Rotorua. Temperature and dissolved oxygen profiles and grab samples of water were collected at depths 0-8, 12, 15.5 and 19 m at least twice a week between 13 January and 28 March 2003. Sampling frequency was increased during periods of calm weather when there was increased likelihood of stratification. *In situ* temperature (2 m depth intervals) and dissolved oxygen (15.5 and 19 m depth) loggers were also used to collect data every 5 minutes during this time. Plots of dissolved oxygen, temperature, total nitrogen (TN) and total phosphorus (TP) were used to calculate the sediment nutrient release rates for stratified periods. Dividing the lake into horizontal 'slabs' in which the volume and concentration of each slab were known produced the estimates of mass of N and P released, which are presented below. The volume of the horizontal slabs was found from the hypsographic curve (elevation vs. area vs. volume) for Lake Rotorua that was supplied by Environment Bay of Plenty. We used similar simplifying assumptions to those for Lake Rotoiti; that inflows contributed little to the observed changes in nutrient concentrations, and that the persistence of stratification, though only of brief duration in Lake Rotorua, maintained discrete horizontal layers with essentially no mixing between the layers.

3.3 Results and discussion - Lake Rotorua

Lake Rotorua showed distinct thermal stratification on two occasions for several days in summer 2002-03. Figure 30a shows these periods as clear separations of temperature at different depths. Concentrations of dissolved oxygen showed similar separations and were reduced to 0.33 mg L^{-1} at the bottom of the water column during one such event (Fig. 30b). The decline in dissolved oxygen also results in a net release of nutrients from the lake sediment.

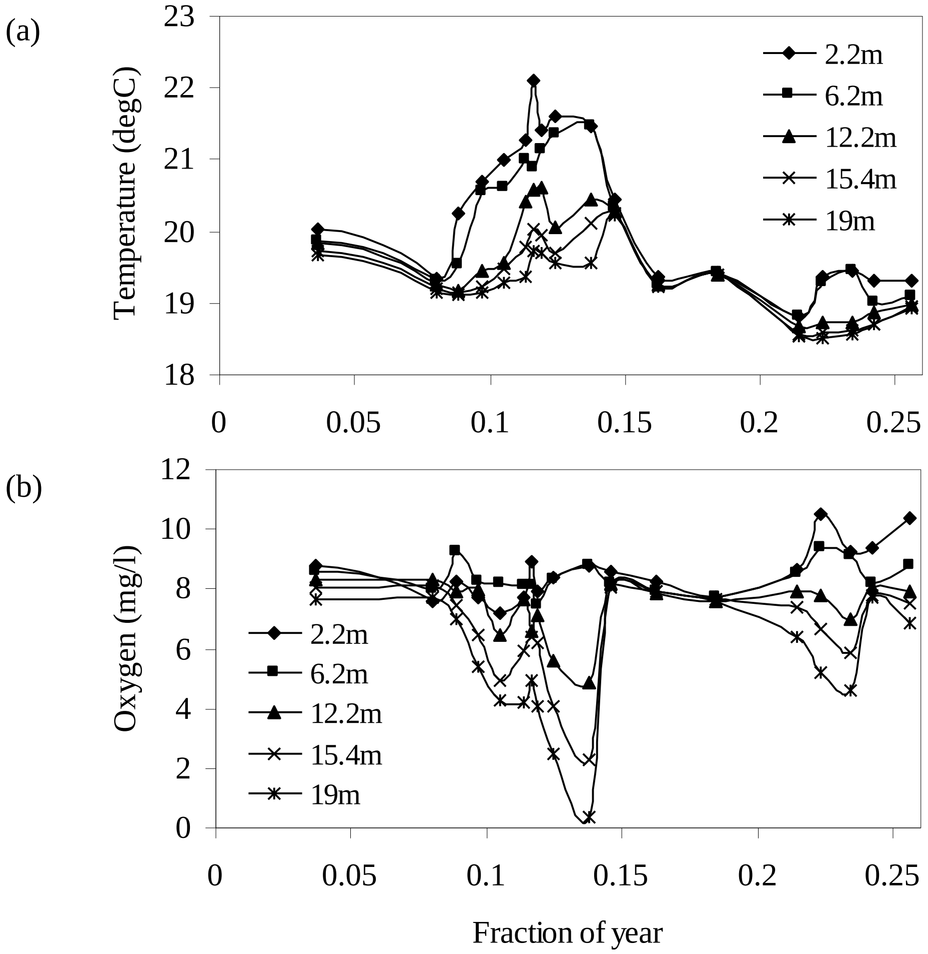


Figure 30. (a) Temperature and (b) dissolved oxygen in Lake Rotorua plotted against time as fraction of a year in 2003. The period plotted is 13 January – 28 March, 2003.

Figure 31a and b show the change in mass of TP and TN, respectively, for Lake Rotorua. Regressions of TP and TN on time are used to determine release rates and contributions of the bottom sediments to TP and TN loads. The details of each release event are given in Table 4, which shows that approximately 24 tonnes TP and 308 tonnes TN were released from the lake sediments in early 2003 in response to stratification/deoxygenation.

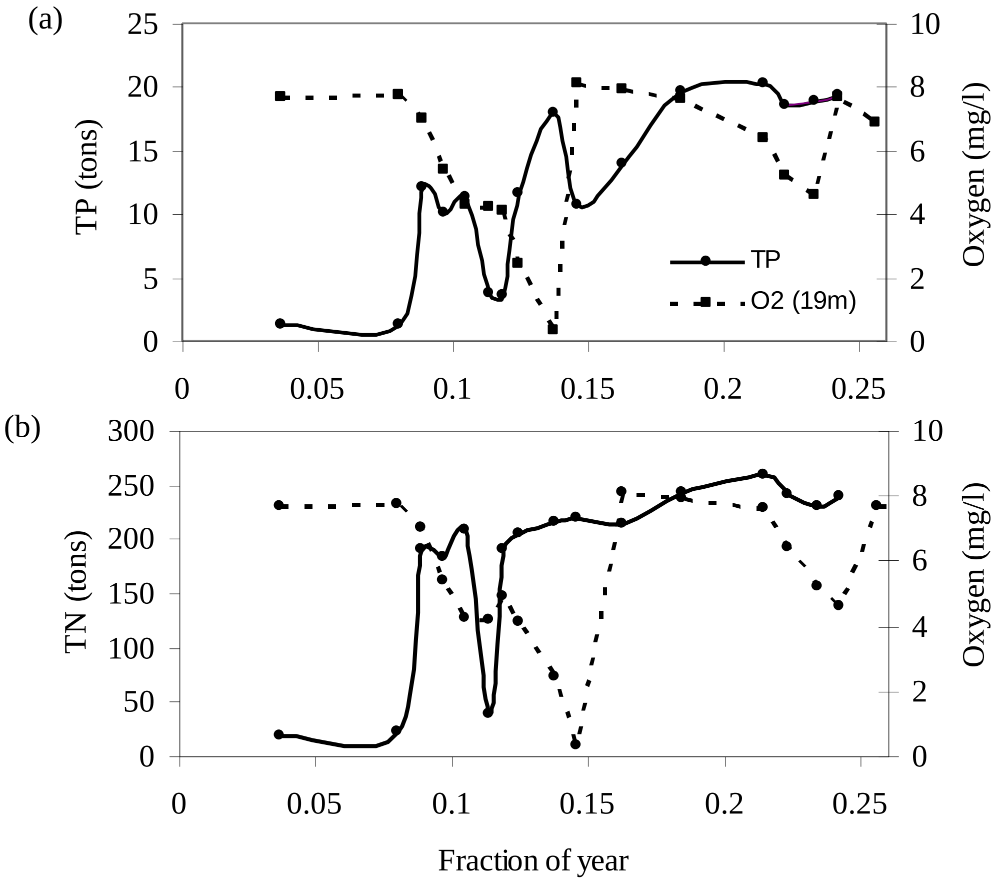


Figure 31. Dissolved oxygen (20 m depth) and (a) total phosphorus and (b) total nitrogen mass in Lake Rotorua plotted against time as fraction of a year in 2003. The period plotted is 13 January – 28 March, 2003.

The TN:TP ratio for this event is 12.8, similar to the ratio for catchment loads, but around the critical value for dominance by cyanobacteria. In terms of annual loads, our estimates for TN and TP are conservative and capture only two stratification-deoxygenation events, both near the beginning of 2003. The combination of nutrient loads from sediment resuspension together with a full description and modelling of anoxia-driven sediment nutrient release over an annual cycle (2003) will provide a more complete picture of the importance of internal loads to nutrients in Lake Rotorua. On the basis of the conservative estimates presented here, it might be expected that contributions from bottom sediments will play a major role in the overall loads to Lake Rotorua in 2003. More work is required to understand whether 2003 is typical or atypical in terms of frequency of anoxia- and wind-driven nutrient release.

Table 4. Changes in mass of total phosphorus and total nitrogen in response to two periods of stratification in Lake Rotorua in 2003.

Nutrient	**Stratification period in 2003**	**Duration (days)**	**Release rate (tonnes day^{-1})**	**Total released (tonnes)**	**Total released (tonnes)**
	29 Jan - 7 Feb	10	0.9	8.4	23.9
TP	10 Feb - 19 Feb	10	1.7	15.5	
	29 Jan - 7 Feb	10	18.2	164.2	307.7
TN	10 Feb - 19 Feb	10	16.2	143.5	

4. Management implications

The comparison of 1981-82 against 2002-03 revealed a major depletion of inorganic nitrogen from surface waters of Lake Rotoiti in 2002-03 and a greater relative abundance of ammonium through the lake. Further evaluations are required to determine if this occurrence is symptomatic of a general temporal trend or specific to the years of investigation, but the incomplete record for the intervening period indicates that it may be part of a general trend for Lake Rotoiti. Increased stratification may lead to more severe nitrogen limitation in surface waters, and significant competitive advantages for N-efficient/N-fixing cyanobacteria. In simple terms, there is not enough nitrate or there is too much phosphorus. One could facetiously ask whether the recent increases in streamflow nitrate to Lake Rotorua (Rutherford *et al*. 2003) might be of benefit in increasing N:P ratios and reducing likelihood of dominance by cyanobacteria? Calcium nitrate, for example, has even been used by ploughing it into the sediments of strongly P-limited lakes, in order to increase the oxidation status of the sediments, bind P with calcium, and ensure that the lake remains strongly P-limited (Cooke *et al*. 1983). This is a high risk option for the Rotorua lakes in view of the sometimes delicately balanced ratio of N:P.

Control of phosphorus is essential in view of the widespread nature of cyanobacterial blooms through the central volcanic plateau of New Zealand in summer 2002-03. As we question whether there is a consistent change (decrease) with time in levels of inorganic N in Lake Rotoiti, we must also question whether the widespread geographical nature of cyanobacterial blooms in summer 2002-03 is symptomatic of a regional trend of low inorganic N or perhaps high inorganic P availability, either of which would have reduced inorganic N:P ratios.

It is also important to acknowledge the importance of anoxia-driven denitrification in bottom waters, particularly as trophic status of lakes increases, deoxygenation takes place, and nitrate follows sequentially in the oxidation-reduction sequence. Where internal nutrient loads are significant in the overall load of nutrients to a lake, it is likely that N:P ratios will decrease with anoxia. Clearly it is essential to enact strong management controls on nitrogen and phosphorus inputs from the catchment where there is an interannual trend of decreasing oxygen in bottom waters, alleviating the resultant internal nutrient load and ensuring that N:P ratios are influenced primarily by catchment sources and not regeneration of nutrients within the lake. This will ensure that by sound management of nutrients in catchments, some control can still be exerted on biomass and species composition of phytoplankton populations.

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