Predicting the effects of nutrient loads, management regimes and climate change on water quality of Lake Rotorua

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Executive Summary

Lake Rotorua is a large body of water (area = 80 km^2, mean depth = 10 m) which is of great cultural significance to the Te Arawa people, and is a focal point for the city of Rotorua's tourism, recreational activity such as boating, kayaking and swimming, as well as an important trout fishery. Water quality has declined progressively in Lake Rotorua since the 1960s in response to influences including inputs of treated wastewater, and land use change from forest to pasture. The input of treated wastewater to the lake from the urban settlement ceased in 1991, but there has been ongoing pastoral expansion and intensification since that time, particularly expansion of dairy and increasing nitrogen-based fertilizer applications. Most of the conversion of forest to pastoral land was from 1970 to 1990. Current challenges to managing water quality of Lake Rotorua include not only nutrient loads both from current land use, but also legacy effects as the dominant cold-water spring inflows to the lake have long response times that reflect gradual enrichment of nitrate in the groundwater sources of these springs in association with changes in land use. Furthermore, there are large nutrient pools in the lake sediments and, with increasing trophic status and intermittent deoxygenation of bottom waters during periods of temperature stratification in the lake, there is an associated high internal (bottom-sediment) nutrient load. Because of the complexity of both spatial and temporal variations in catchment nutrient loads, and the way in which Lake Rotorua responds to these loads, a modelling approach is the best way to allow for quantitative assessments of the water quality effects of different land uses, climates and in-lake actions for the purposes of managing lake water quality. For water managers, quantitative statistics are required in order to clearly demonstrate the effects of land use change on water quality. These statistics must be science-based, defensible from the perspective of policy implementation, and open to challenge by affected groups. We linked together different system-specific models to simulate time-varying water quality responses which were expressed primarily in terms of the trophic state of the lake, based on a trophic level index (TLI). The models used in this study include:

- SimCLIM: a model developed in the International Global Change Institute at the University of Waikato, that downscales data from atmosphere-ocean general circulation models used by the International Panel for Climate Change (IPCC) to hindcast past climates and predict a future climate, up to 2100. For the present case SimCLIM was used to generate past and future air temperature and rainfall within the Rotorua catchment at a spatial scale of 25 m. This model output was then used as rainfall input for ROTAN (see below) and as the rainfall and air temperature input for DYRESM-CAEDYM (see below).

• ROTAN: the ROtorua TAupo Nitrogen model, developed by the National Institute of Water and Atmospheric Sciences (NIWA). ROTAN is a rainfall-runoff-groundwater model that produces streamflows for sub-catchments of Lake Rotorua. Nitrate is included as a solute transported between surface water and groundwater compartments of ROTAN, with provision of non-conservative processes due to biogeochemical fluxes of nitrate. Outputs of stream discharge and nitrate concentrations were then used as input data for the lake model simulations.

• CLUES: the Catchment Land Use for Environmental Sustainability (CLUES) model developed by NIWA. This model produces annual average total phosphorus loads in streams, which were adjusted to stream phosphorus concentrations using discharges simulated with ROTAN, to provide daily input concentrations suitable for the lake model.

• DYRESM-CAEDYM is a coupled hydrodynamic-ecological model developed by the Centre for Water Research, University of Western Australia, that simulates the one-dimensional (vertical) distribution of water temperature and chemical and biological constituents of water that are relevant to lake trophic state. Raw data output from DYRESM-CAEDYM was transformed to obtain indices of direct relevance to the assessment of trophic state in the lake, such as the Trophic Level Index (TLI).

The results showed that it would be necessary to remove at least 350 tonnes of nitrogen per year (t y^{-1}) and a similarly large relative fraction of phosphorus load from the lake catchment nutrient load in order to firstly arrest the present trend of declining water quality, reduce the duration of bottom water anoxia and frequency of cyanobacterial blooms, and subsequently to achieve a sustained reduction of TLI at or near the target value of 4.2. This goal may necessitate land use change as well as improved management of nutrients from all sources, i.e., potentially reversing some of the land use trends that have taken place until the mid 2000s. Climate change would make the lake water quality targets more difficult to attain because of increases in stratification duration and intensity in this polymictic lake which has intermittent periods of stratification in calm weather interspersed with mixed periods at other times. Without improvements in water quality, increased stratification will extend the duration and extent of anoxia in bottom waters and increase internal releases of nutrients (i.e., those arising from the bottom sediments). The modelling also considered mechanisms to reduce internal loading of nutrients, specifically using chemical flocculants in one-off dosing of sediments across a large area of the lake. Our empirical data suggest that the duration of effective sediment capping may be of the order of four years. We simulated the lake water quality using nutrient release rates from bottom sediments that we altered in order to approximate the effects of a flocculation/sediment capping treatment. This treatment hastened the effects of land use change and offset a deterioration in water quality resulting from climate change, but it must still be considered a risky option if catchment nutrient loads are not attenuated effectively first, and will likely have limited duration of effectiveness. In summary the model results indicate that only a sustained and pro-active nutrient management strategy to reduce catchment nutrient loads will make it possible to reverse the present buildup of nitrate in groundwater aquifers within the catchment and to confer resilience of lake water quality to a changing climate, thereby sustainably meeting the TLI targets set for improvements in water quality of Lake Rotorua.

Acknowledgements

We thank the following people and organisations for assistance in making the report possible: Kit Rutherford and Chris Palliser from NIWA for providing ROTAN simulation output, Sandy Elliot for assistance with CLUES, Uwe Morgenstern from GNS for information and discussion of groundwater ages and Simon Park from Headway Ltd for nutrient loads associated with different land use scenarios. We thank Andy Bruere, Rob Donald and Paul Scholes for comments received in the preparation of this report, as well as Glenn Ellery (Bay of Plenty Regional Council) and John McIntosh (Lochmoigh Consultants) for provision of data. The model DYRESM-CAEDYM was obtained from the Centre for Water Research, University of Western Australia.

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1. Introduction

Lake Rotorua is a major asset to the city of Rotorua. It is a lake of great cultural significance to Māori, is an important trout fishery and provides important recreational and tourism opportunities. However, water quality of Lake Rotorua has declined progressively over the past four decades (e.g. Fish 1969; Rutherford et al. 1996; Burger et al. 2008) to the extent that some of these activities have been jeopardised and the mauri (life force) of the lake is threatened. As a result an Action Plan has been developed by Bay of Plenty Regional Council (BoPRC) to improve water quality of the lake as part of a restoration initiative with its partners the Te Arawa Lakes Trust and the Rotorua District Council. Key parts of this Action Plan are considerations of land use change and artificial nutrient controls on stream inflows and lake bottom-sediments.

The primary tool used by BoPRC to assess and report on lake water quality is the Trophic Level Index (TLI). This index combines individual trophic level values assigned to concentrations of chlorophyll *a*, total phosphorus and total nitrogen, and transparency assessed by Secchi disk depth (Burns et al. 1999). It is generally acknowledged that water quality of Lake Rotorua began to show marked signs of deterioration in the 1960s; a TLI target of 4.2, corresponding to the TLI value around the 1960s, has been set as a goal by BoPRC to restore water quality of the lake (Figure 1). Therefore, the objective of undertaking the present modelling exercise was to determine what combination of external nutrient load reductions, together with any in-lake actions, may be used to reduce to the present TLI to a value of 4.2 or less on a long-term basis.

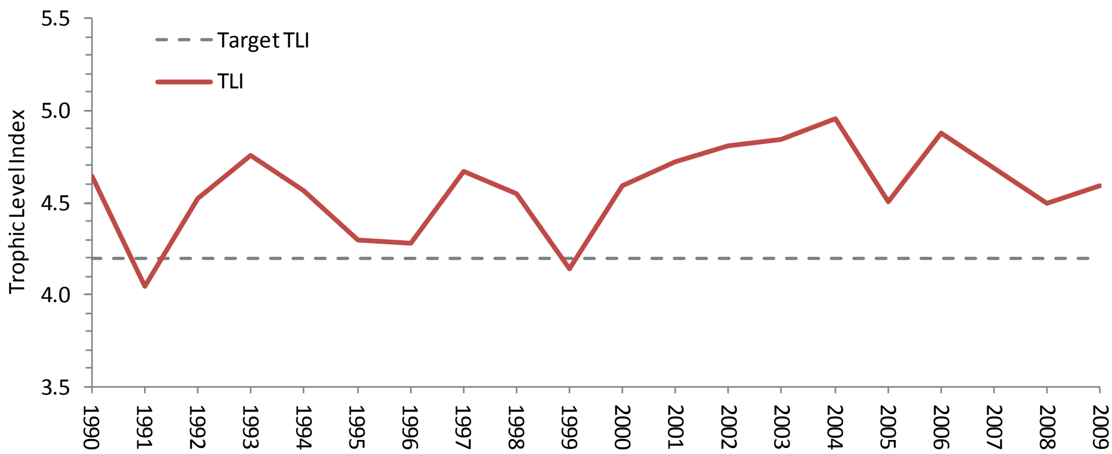


Figure 1. Estimated Trophic Level Index (Burns et al. 1999) of Lake Rotorua for calendar years between 1990 and 2009. The Lake Rotorua target TLI (BoPRC 2009) is shown by the dashed grey line.

In 1991 discharge of treated effluent to Lake Rotorua ceased with the implementation of land-based effluent polishing in an area of the Whakarewarewa forest in the Puarenga sub-catchment of the lake. A resource consent for 'sewage-derived' loads to Lake Rotorua permits an upper limit of 30 t y^{-1} for total nitrogen (N) and 3 t y^{-1} for total phosphorus (P) from this source. The P limit has generally been met, while the recent N load has been approximately 35 t y^{-1} even with modifications of treatment and discharge practices by Rotorua District Council (Park and Holst 2009). The Puarenga Stream is also showing a trend of increasing nitrate concentrations which were, however, evident even before land-based effluent disposal in this sub-catchment (Rutherford et al. 2008). The expected improvements in lake water quality as a result of wastewater diversion from the lake were predicated upon catchment nutrient loads of approximately 34 t P y^{-1} and 405 t N y^{-1} (excluding atmospheric deposition of N and P), typical of those in the 1960s. Coupled with implementation of the Kaituna Catchment Control Scheme in the 1980s, which appeared to have largely arrested increases in particulate P loads from earlier pastoral conversions (Williamson et al., 1996), water quality of Lake Rotorua was expected to stabilise and improve. There were noticeable improvements in water clarity through the period 1993-1995, following land-based effluent discharge in 1991 (Rutherford et al. 1996), but the TLI increased dramatically in the early 2000s and although TLI has decreased slightly between c. 2007 to 2009, for the past decade Lake Rotorua has been characterised by frequent blooms of cyanobacteria (blue-green algae) through summer-autumn as well as more prolonged bottom-water anoxia (loss of dissolved oxygen) during periods of intermittent stratification of lake water in warmer months of the year (Burger et al. 2007a).

The expected improvements in lake water quality in response to removing wastewater discharges to the lake have been offset by two main factors. First is that bottom sediments of Lake Rotorua have been strongly enriched in nitrogen and phosphorus as a result of many years of eutrophication (Trolle et al. 2010) and releases of these nutrients are greatly enhanced as levels of dissolved oxygen decline when the water column stratifies, which can occur for periods of up to one month (Vant 1987). On an annual basis, loads contributed from this internal release mechanism are of similar magnitude to those arising from the lake catchment (see Burger et al. 2008; Bay of Plenty Regional Council 2009).

The second influence related to the ongoing eutrophication of Lake Rotorua has been a trend of increasing nitrate concentrations observed in many of the stream inflows to the lake (1968-2003; Rutherford et al. 2011). This trend has been attributed to gradual nitrate enrichment of aquifers draining agricultural land in the catchment, with changes in nitrate concentration reflecting a time history of agricultural development and intensification. The duration over which nitrate is retained within the sub-catchment aquifer can be many decades for some of the largest aquifers, such as those associated with the Hamurana and Awahou Streams.

The restoration of Lake Rotorua is a long-term goal of the Rotorua Te Arawa Lakes Strategy Group, which is the overarching lake management group made up of representatives from Bay of Plenty Regional Council, Te Arawa Trust Board and Rotorua District Council. Restoration of the lake can be expected to have far-reaching benefits that extend beyond the immediate lake and catchment areas. For example, the construction of the $10 million Ohau Channel diversion wall was deemed necessary to protect water quality in Lake Rotoiti from the effects of the nutrient-enriched inflow arising from Lake Rotorua. The projected time-frame for the diversion wall to be in place is approximately 50 years; a duration on which nitrogen loads to Lake Rotorua and through the Ohau Channel might reasonably be expected to decline in response to management actions in the Rotorua catchment. Furthermore, problems with algal blooms in the Kaituna River would likely be alleviated as water quality in Lake Rotorua is improved.

Because of the complexity of both spatial and temporal variations in catchment nutrient loads, and the way in which Lake Rotorua responds to these loads, a modelling approach is the best way to provide quantitative assessments of the water quality effects of different land uses, climates and in-lake actions for the purposes of managing lake water quality. For water managers, quantitative statistics are required to be able to clearly demonstrate the effects of land use change on water quality. These statistics must be rigorous and science-based, defensible from perspectives of policy implementation, and open to challenge by affected groups. No single model is capable of coupling climate, nutrient loads and lake water quality. For this reason, different system-specific models have been linked together through their respective model inputs and outputs, in order to simulate time-varying TLI responses to changes in nutrient loads and climate. The models used in this study include:

- SimCLIM: a model that downscales data from atmosphere-ocean general circulation models to hindcast past climates and predict climate at some point in the future, up to 2100 (Warrick et al. 2005). For the present case SimCLIM was used to generate past and future air temperature and rainfall within the Rotorua catchment at a spatial scale of 25 m. This model output was then used as rainfall input for ROTAN (see below) and as the rainfall and air temperature input for DYRESM-CAEDYM (see below).

- ROTAN: the ROtorua TAupo Nitrogen model. ROTAN is a rainfall-runoff-groundwater model that produces streamflows for sub-catchments of Lake Rotorua (Rutherford et al. 2009, 2011). Nitrate is included as a solute transported between surface water and groundwater compartments of ROTAN, with provision of non-conservative processes due to biogeochemical fluxes of nitrate. Outputs of stream discharge and nitrate concentrations were then used as input data for DYRESM-CAEDYM simulations in the present study.

- CLUES: the Catchment Land Use for Environmental Sustainability (CLUES) model. This model produces annual average total nitrogen, total phosphorus, *E. coli* and sediment loads in streams at a variety of spatial scales from individual stream reaches and subcatchments up to a whole-nation scale, based on land use, geology and land morphology. It is a GIS-based modelling system and specifically targets long-term land use change effects on catchment nutrient fields. For the purposes of the present study CLUES was used to generate total phosphorus loads in individual stream reaches of Lake Rotorua corresponding to those used in ROTAN. Annual average P loads were adjusted to stream P concentrations using discharges simulated with ROTAN, to provide input data suitable for DYRESM-CAEDYM.

- DYRESM-CAEDYM is a coupled hydrodynamic-ecological model that has been applied to lakes and reservoirs around the world to simulate the one-dimensional (vertical) distribution of water temperature and chemical and biological constituents of water that are relevant to lake trophic state (Hipsey and Hamilton 2008). In a review of internationally-available lake water quality models it was considered to be the leading model of its type for water quality predictions (Trolle et al. 2011b). Raw data output from DYRESM-CAEDYM has also been transformed in this study to obtain indices of direct relevance to the assessment of trophic state in the lake, such as TLI.

This report presents results from DYRESM-CAEDYM simulations and assesses these results in relation to the goal of attaining a TLI value of ≤ 4.2 to meet the objectives for water quality in Lake Rotorua defined in the Bay of Plenty Regional Council (BoPRC) Regional Water and Land Plan. As background, a detailed description is given of the derivation of input data for DYRESM-CAEDYM, calibration of the model and comparison against measured data. Outcomes are presented for a series of simulated scenarios designed to test for the effects of different external nutrient loadings corresponding to changes in land use, management interventions related to artificially modifying internal (bottom sediment-derived) nutrient loads or stream inflows, and climate change to better understand the implications of a warming climate that is predicted to occur in responses to increasing atmospheric carbon dioxide.

2. Methods

2.1 Study site – Lake Rotorua

The Lake Rotorua catchment has an area of approximately 425 km^2 (Figure 2). The hydrogeology of the catchment is complex with permeable pumiceous tephra overlying massive deep rhyolite and ignimbrite unconfined aquifers that retain groundwater for long and variable periods. Rutherford et al. (2011) produced a series of land use maps that indicate how land use of the Rotorua catchment has changed between 1940 and 2010. In summary, the urban area has expanded substantially during this time but with progressive reticulation of septic tanks into the centralised wastewater treatment plant within this area. Relatively low-intensity pastoral land cover comprising mostly sheep and beef farms covered a considerable area of the lower catchment as early as 1940 but also expanded rapidly around the lake margin and then progressively into the upper catchment. Dairy farming has more recently replaced both forest and sheep and beef farming in the north-west of the catchment, mostly in and around the Mamaku Plateau, particularly over the past 3-4 decades. In general there has been a gradual loss of forest with the expansion of pasture in the Rotorua catchment to the point where pasture now makes up about 50% of the total catchment area.

Phytoplankton biomass and production in Lake Rotorua may be limited by nitrogen and/or phosphorus, as well as other environmental factors depending on time of year and the location within the lake (Burger et al. 2007b). Ratios of N:P are low by comparison with many other lakes in New Zealand and particularly overseas, and, based on Redfield ratios (see Abell et al. 2010) suggest that either nutrient could potentially limit phytoplankton productivity. White et al. (1977) found using laboratory based bioassays that N consistently limited algal biomass. Burger et al. (2007b) ran similar but *in situ* bioassays and also used a model to show that in different seasons co-limitation by N and P was the norm. Bioassays carried out in large-scale mesocosms in Lake Rotorua in summer 2009-10 also indicated that co-limitation was the most frequent case, as opposed to limitation by either N or P (Hamilton, unpubl. data). For these reasons, the Technical Advisory Group for the Rotorua lakes has consistently advocated (EBoP 2004) for maintaining controls on loads of both N and P.

Reducing solely N or P would little constrain phytoplankton biomass during periods when phytoplankton would otherwise be co-limited, or limited by the alternate nutrient species. Furthermore, attempting to reduce phytoplankton production by P-limitation via P load reduction may be difficult to achieve on a sustainable basis, given the natural geological enrichment of phosphorus in aquifers within the Rotorua catchment (Timperly 1986). Focus solely on N load reduction could potentially increase the risk of cyanobacterial blooms if there were periods of adequate P, and N-limitation sufficient to induce proliferations of heterocystous species that could fix atmospheric N to meet the shortfall in their nutritional requirements for N (Wood et al. 2010).

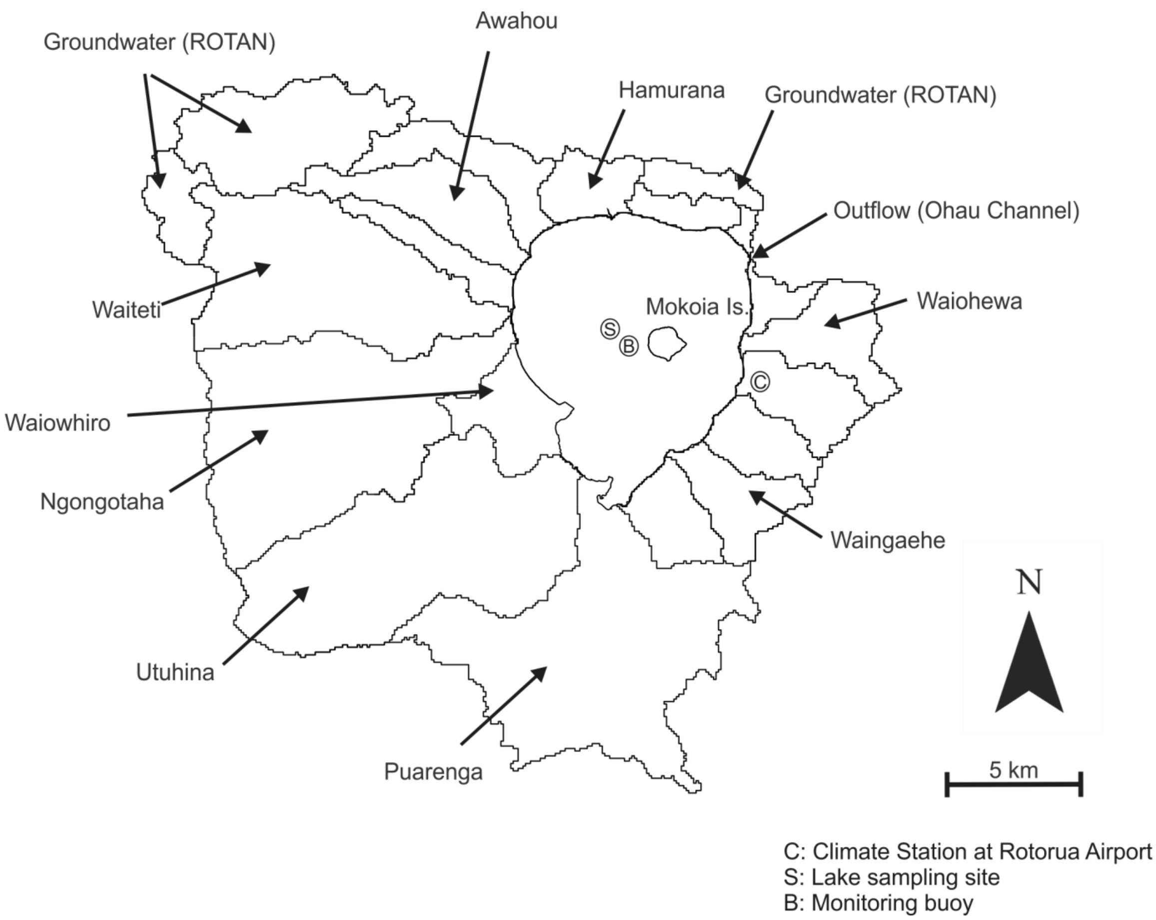


Figure 2: Map of the Lake Rotorua catchment with details of ROTAN sub-catchments (from Rutherford et al. 2011), and showing the BoPRC sampling site, the high-frequency monitoring buoy and the Rotorua airport weather station. Mokoia Island is the closed circle in the centre of the lake, and unnamed subcatchments are those without permanently flowing surface streams. Unnamed sub-catchments have no permanently flowing surface stream.

2.2 Lake model description

In this study, the one-dimensional (1D) hydrodynamic model DYRESM (version 3.1.0-03) was coupled with the aquatic ecological model CAEDYM (version 3.1.0-06), both developed at the Centre for Water Research, The University of Western Australia, to simulate water quality in Lake Rotorua. DYRESM resolves the vertical distribution of temperature, salinity, and density, and the vertical mixing processes in lakes and reservoirs. CAEDYM simulates time-varying fluxes that regulate biogeochemical variables (e.g., nutrient species, phytoplankton biomass). The model includes comprehensive process representations for carbon (C), nitrogen (N), phosphorus (P), and dissolved oxygen (DO) cycles, and several size classes of inorganic suspended solids. Several applications have been made of DYRESM-CAEDYM to different lakes (e.g., Bruce et al., 2006; Burger et al., 2007a; Trolle et al., 2008; Gal et al., 2009; Özkundakci et al., 2011) and these publications have detailed descriptions of the model equations.

The biogeochemical variables in CAEDYM may be configured according to the goals of the model application and availability of data. In this study, three groups of phytoplankton were included in CAEDYM, representing generically cyanophytes (without N-fixation), diatoms, and a combined group termed chlorophytes. Particular focus was placed on cyanobacteria because of their capability to form blooms, with model output interpreted to provide a frequency distribution of different cyanobacteria levels represented by their contribution to chlorophyll *a*. The interactions between phytoplankton growth and losses, sediment mineralisation and decomposition of particulate organic matter influence N and P cycling in the model as shown in the conceptual model in Figure 3. Fluxes of dissolved inorganic and organic nutrients from the bottom sediments are dependent on temperature and nitrate and DO concentrations of the water layer immediately above the sediment surface. Parameters are calibrated (i.e., specific) for each new application but with an extensive parameter library now available from the large number of studies undertaken with the model.

2.3 Model timesteps and baseline simulation period

In this study, DYRESM-CAEDYM was run at hourly time steps between July 2001 and June 2009, with daily averaged input data and daily output data at 0900 h. The period July 2005 to June 2009 was used for calibration of the model, and the validation period was July 2001 to June 2005.

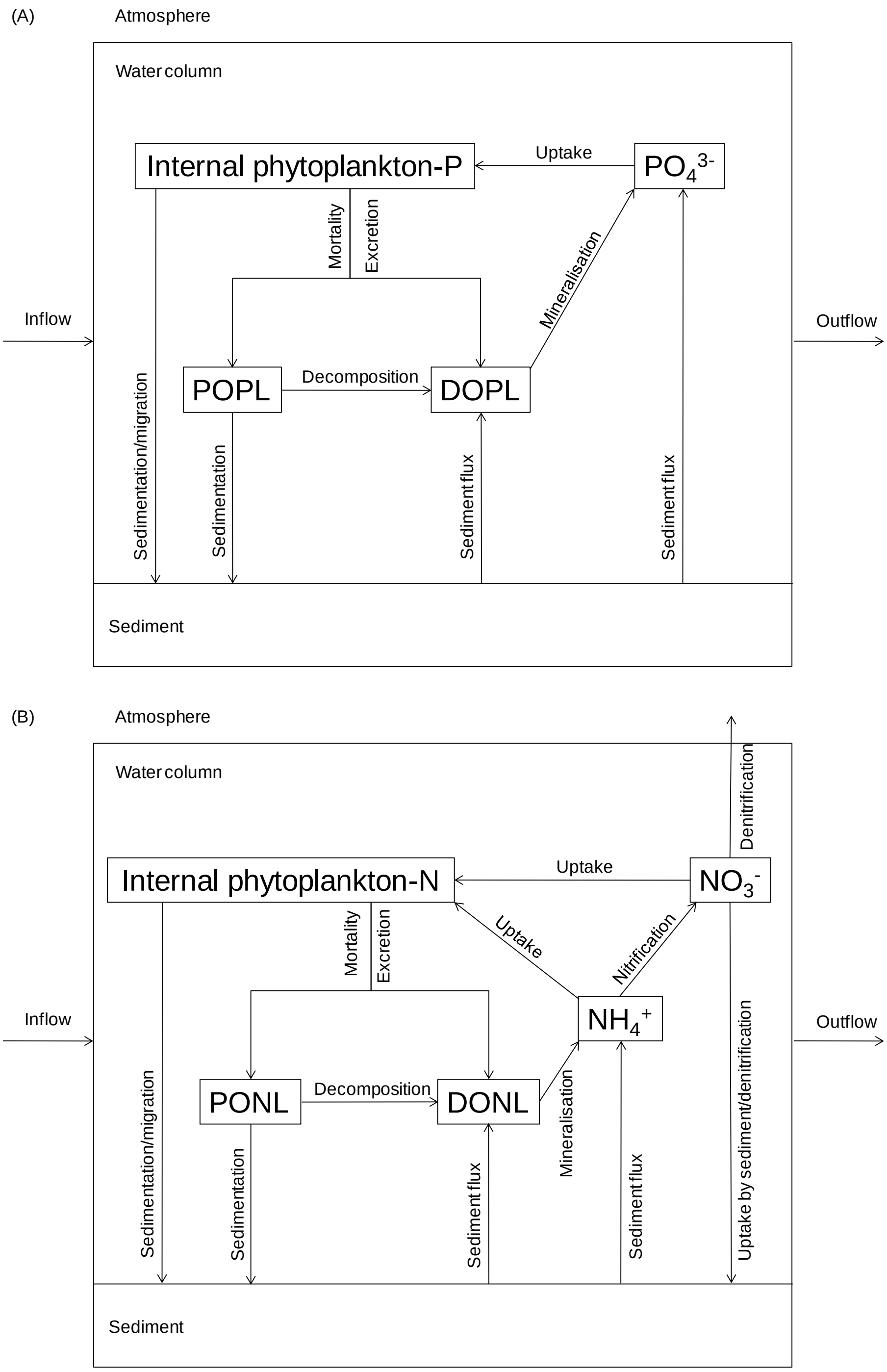


Figure 3: Conceptual model of the (A) phosphorus and (B) nitrogen cycles represented in DYRESM-CAEDYM for the present study. POPL, PONL, DOPL and DONL represent particulate labile organic phosphorus and nitrogen, and dissolved labile organic phosphorus and nitrogen, respectively.

2.4 Meteorology

Meteorological data required for the simulation period were obtained from the National Climate Data Base for the Rotorua Airport climate station c. 50 m from the Lake Rotorua shoreline. The data included air temperature (°C), shortwave radiation (W m^{-2}), cloud cover (fraction of whole sky), vapour pressure (hPa), wind speed (m s^{-1}) and rainfall (m) (Figure 4). Data are collected at Rotorua airport at various time intervals from one hour to whole-day, and for the purposes of the model input were standardised to daily average values except for rainfall, which was provided as a daily total value.

2.5 Water balance

Surface inflow discharges to the lake were obtained from output of the ROtorua TAupo Nitrogen model (ROTAN; Rutherford et al., 2011). Flows for nine streams of the major Lake Rotorua sub-catchments were included, along with a tenth stream inflow representing the sum of all minor surface flows from around the lake. Each of these surface flows accounted for both stream and groundwater inputs from the respective sub-catchments. Rainfall was removed from the meteorological input to the model, and instead daily rainfall directly on the lake was represented as a surface inflow in order to account for atmospheric deposition of N and P which would not otherwise be accounted for in the rainfall input in the present model version.

Change in lake storage (ΔS) was calculated from water level recorder data provided by BoPRC, multiplied by the water level-dependent lake area derived from hypsographic curves provided by BoPRC.

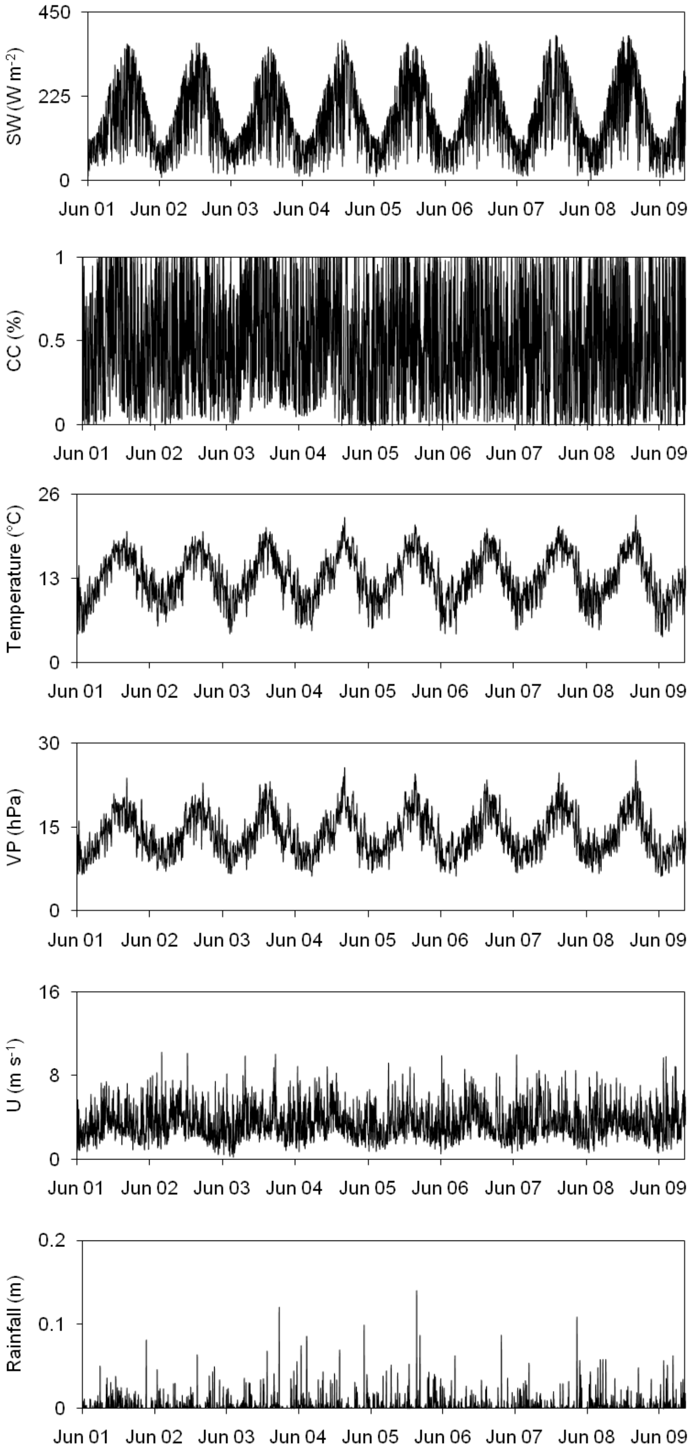


Figure 4. Meteorological data used as input to the DYRESM model (July 2001 – June 2009), including short wave radiation (SW), cloud cover (CC), air temperature (T), vapour pressure (VP), wind speed (U), and rainfall. Data were obtained from the Rotorua Airport climate station (see text for details of transforming measured values to daily input data for model input).

Evaporation from the lake was calculated as a function of wind speed and air vapour pressure from the daily average evaporative heat flux (Fischer et al., 1979 eqn 6.20) using meteorological input data and water temperature:

$$Q_{lh} = minimum\left(0, \frac{0.622}{P} C_L \rho_A L_E U_A (e_A - e_S(T_S))\Delta t\right) \qquad (1)$$

where:

Q_{lh} is the evaporative heat flux in J m^{-2} s^{-1}

P is atmospheric pressure in hPa

C_L is the latent heat transfer coefficient for wind speed at a height of 10 m (1.3 x 10^{-3})

ρ_A is the density of air in kg m^{-3}

L_E is the latent heat evaporation of water (2.453 x 10^6) in J kg^{-1}

U_a is the wind speed in at 10 m height above ground level in m s^{-1}

$e_s(T_s)$ the saturation vapour pressure at the water surface temperature in hPa

e_a is the vapour pressure of the air in hPa

The condition that $Q_{lh} \geq 0$ excludes condensation effects. For the purposes of determination of water evaporated from the lake surface, a surface lake water temperature was estimated from a fifth-order polynomial fit of surface water temperature versus time of year using measured data from 2002-2009. The saturated vapour pressure $e_s(T_s)$ is calculated via the Magnus-Tetens formula (TVA 1972, eqn 4.1):

$$e_s(T_S) = exp\left(2.3026\left(\frac{7.5T_S}{T_S+237.3} + 0.7858\right)\right) \qquad (2)$$

where:

T_s is the water surface temperature in °C

The change in mass in the surface layer (layer N) due to latent heat flux is calculated as

$$\Delta M_N^{lh} = \frac{-Q_{lh}A_N}{L_V} \qquad (3)$$

where:

ΔM_N^{lh} is the change in mass in kg s^{-1} (L s^{-1})

A_N is the surface area of the lake in m^2

L_V is the latent heat of vaporisation for water (2.258 x 10^6) in J kg^{-1}

The result of this calculation was multiplied by 86.4 to give daily evaporation (E_L) in m^3 d^{-1}.

Daily values for the outflow volume were calculated as a residual term of a water balance for the simulation period:

$$Outflow = \sum(surface\ inflows) + rainfall - E_L - \Delta S \tag{4}$$

where:
E_L is evaporation in $m^3\ d^{-1}$
ΔS is change in storage in $m^3\ d^{-1}$

The resulting flow was averaged over 15 days to remove any negative values. Derived outflow was used for a DYRESM simulation over the period 2001–2009, and the lake level output was compared to BoPRC water level recorder data. Due to a slight gradual decrease in simulated versus observed water level, a small proportion of outflow (c. 3%) was removed for each day. This small discrepancy may be partly attributable to differences between the estimated surface lake water temperature used to derive evaporation for the water balance (Eqn 2), and the surface water temperature simulated within DYRESM which was ultimately used to derive the water level. Multi-day averaging of the outflow may have contributed additional uncertainty. Lake water level recorded by the BoPRC water level gauge, and simulated in the final water balance as water level output from DYRESM, were closely matched (Figure 54).

The estimated outflow to close the calculated water balance and the observed Ohau Channel discharge were very similar (Figure 6). Given that derived outflow was based on ROTAN-modelled inflows (as opposed to recorded stream flows), the match was considered to be highly satisfactory. Differences between simulated and observed outflow (e.g. 2004 – 2006) are likely due to minor differences between sub-catchment surface flows as modelled by ROTAN, and real-world lake inflow volumes, as well as minor differences in evaporation as described above.

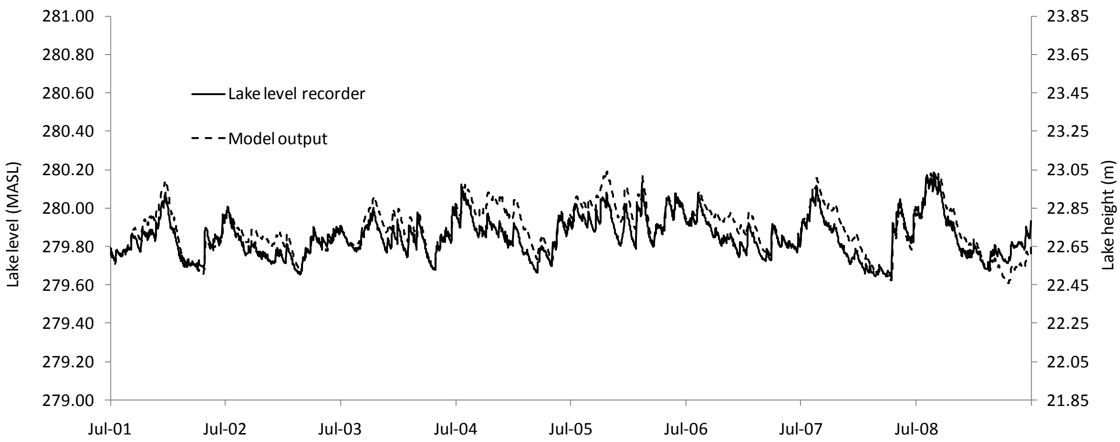


Figure 5. Simulated Lake Rotorua water level, and observed BoPRC water level recorder data.

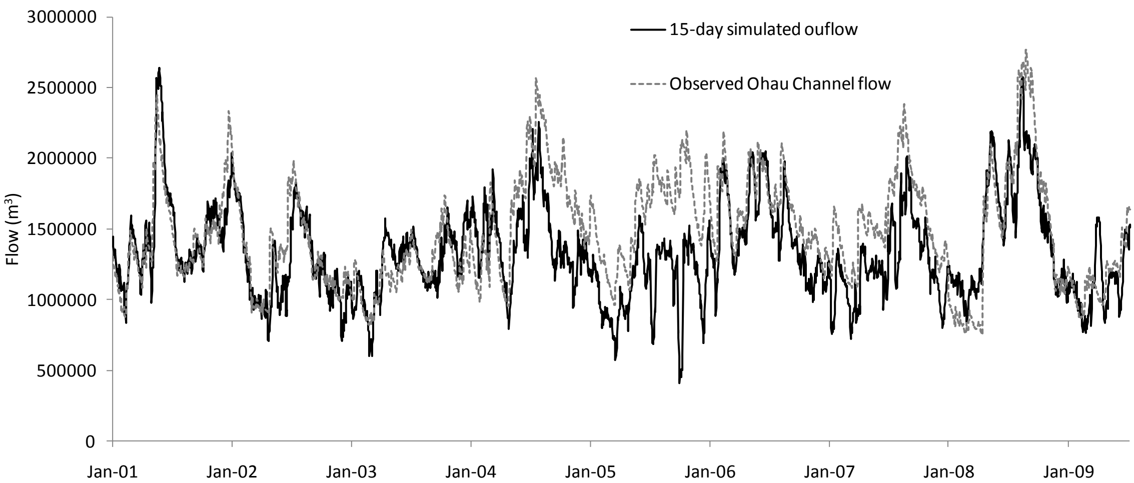


Figure 6. Simulated 15-day average Lake Rotorua outflow derived from the lake water balance, and observed Ohau Channel flow.

2.6 Surface inflow, groundwater and rainfall parameterisation

A total of 11 inflows to Lake Rotorua were simulated, including nine major streams, an inflow representing the sum of all inflows from minor sub-catchments, and another representing direct rainfall to the lake surface. All non-rain inflows were assumed to represent both surface and groundwater inputs to the lake, in order to be consistent with the modelling approach within ROTAN (Rutherford et al. 2011).

2.6.1 Temperature

Surface inflow temperatures for all major inflows other than Hamurana were estimated using the method described in Mohseni et al. (1998):

$$T_s = \frac{\alpha}{1+e^{\gamma(\beta-T_a)}} \tag{5}$$

where:

T_s is the estimated stream temperature
T_a is the measured air temperature
α is the coefficient for the estimated maximum stream temperature
γ is a measure of the steepest slope of the function
β represents the air temperature at the inflection point

To estimate surface temperature of Hamurana Springs and minor cold inflows, a cosine variation was applied to describe variations within each year:

$$T_s = A\cos(\omega t + \sigma) + T_0 \tag{6}$$

where:

T_s is the derived water temperature in ^oC
A is the amplitude in m
ω is the angular frequency (2π/365)
σ is the phase angle
T_0 is the mean water temperature in ^oC
t is time in days

For both temperature models, quality of fit was defined by the difference between modelled water temperature and available *in situ* BoPRC measurements for each stream. Model parameters were adjusted in order to minimise the root-mean-square error (RMSE) and maximise the Pearson correlation co-efficient, using *Microsoft Excel Solver.*

Temperature of the rainfall inflow was set to be equal to the surface lake water temperature defined in the water balance, so that rainfall was always inserted into the uppermost water layer of the model.

2.6.2 Dissolved oxygen

Dissolved oxygen concentrations of all inflows were estimated as a function of water temperature (Mortimer 1981) based on data from Benson and Krause (1980):

$$DO = exp(7.71 - 1.31 ln(T + 45.93)) \quad (7)$$

where:

DO is dissolved oxygen in mg L^{-1}
T is water temperature in $^{\circ}C$

2.6.3 Nutrients

Daily nitrate (NO_3-N) concentrations for all major inflows were derived from interpolation of weekly flow and nitrate load output from ROTAN. Ammonium (NH_4-N) and phosphate (PO_4-P) concentrations were derived by linear interpolation between monthly samples from BoPRC stream monitoring data to obtain daily values, as outlined for other model applications to Te Arawa lakes (Burger et al., 2007; Trolle et al., 2010; Özkundakci et al., 2011). As most streams monitoring is undertaken by Bay of Plenty Regional Council under baseflow conditions, the interpolation method to fill nutrient concentrations for days between monthly sampling may potentially underestimate total nutrient loads; less so for nitrogen but particularly for particulate forms of phosphorus (J. Abell, University of Waikato, unpubl. data). Labile organic nitrogen and phosphorus concentrations (ONL and OPL, respectively) were calculated from BoPRC monthly stream nutrient measurements and were evenly divided into dissolved (D) and particulate (P) fractions using the equations:

$$\text{DONL or PONL} = (TN - NH_4\text{-N} - NO_3\text{-N}) / 2 \quad (8)$$

$$\text{DOPL or POPL} = (TP - PO_4\text{-P}) / 2 \quad (9)$$

Labile dissolved and particulate organic carbon concentrations were calculated using inflow labile organic nitrogen concentrations and a 'Redfield' molar ratio of 106:16 for C:N, where:

$$\text{DOCL or POCL} = (\text{DONL or PONL}*106*M(C)) / (16*M(N)) \quad (10)$$

where:

M(C) is the molar mass of carbon

M(N) is the molar mass of nitrogen.

Nitrogen and phosphorus concentrations for the 'cold minor' inflow in the model were calculated using similar methods to those for the major inflows, but using a volumetric average for concentrations across all available minor inflow data.

Rainfall nitrate and phosphorus were set to constant concentrations of 0.285 and 0.013 g m^{-3}, respectively, which with consideration of rainfall amount, equated to concentrations typical of the Taupo Volcanic Zone (Hamilton 2005). No particulate or dissolved organic nitrogen, phosphorus or carbon fractions were prescribed to the rainfall inflow representing atmospheric deposition (Table 1).

Table 1. Flow and nutrient loads for all inflows over the calibration/validation period, 2001 – 2009. Symbols represented by NH_4, NO_3 and PO_4 are dissolved nutrient concentrations represented by NH_4-N, NO_3-N and PO_4-P, respectively.

Inflow load for scenario	NH_4	NO_3	Organic N	PO_4	Organic P	Flow
			t yr^{-1}			m^3 yr^{-1}
Hamurana	0.8	56.3	5.1	6.2	0.2	7.494×10^7
Waiteti	0.4	54.7	3.6	3.3	0.1	4.758×10^7
Awahou	0.7	67.7	4.4	1.5	0.3	3.870×10^7
Ngongotaha	1.0	66.3	10.2	2.2	0.9	6.049×10^7
Waiowhiro	0.4	16.0	1.8	0.6	0.1	1.430×10^7
Utuhina	2.6	61.2	10.2	2.7	1.4	6.630×10^7
Puarenga	4.1	106.3	14.1	2.9	1.8	6.002×10^7
Waingaehe	0.1	12.4	1.5	0.8	0.2	8.780×10^6
Waiohewa	16.7	51.3	2.6	0.3	0.5	1.171×10^7
Minor	54.8	143.4	0.4	2.9	1.5	4.881×10^7
Rainfall	0.0	28.8	0.0	1.3	0.0	1.009×10^8
Total	81.7	664.4	53.8	24.8	7.0	532562204

2.6.4 Phytoplankton

Those inflows not dominated by groundwater springs were prescribed a 'seeding' concentration of 0.1 µg L^{-1} chlorophyll *a* for the assigned cyanophyte and diatom groups though the chlorophytes group was assigned zero concentration based on mostly lower concentration in Lake Rotorua. The Hamurana, Awahou and rainfall inflows were given a concentration of zero for each of the three phytoplankton groups.

2.7 Model calibration and validation

DYRESM-CAEDYM was calibrated against field data for a four-year period between July 2005 and June 2009 for variables of temperature, DO, PO_4-P, TP, NH_4-N, NO_3-N and TN at the water surface (0 m), at a depth of 15 m, and near the bottom (depth of 19 m) in the lake, using monthly samples collected by BoPRC. Nutrient samples were analysed by BoPRC using standard analytical methods (APHA, 2005). The three simulated phytoplankton groups collectively contributed to a total simulated chlorophyll *a* concentration, but with cyanophytes dominating during summer and diatoms/chlorophytes during winter and early spring, in a sequence similar to what has been observed previously in Lakes Rotorua (Paul et al. 2012) and Rotoiti (Von Westernhagen et al. 2010). The sum of the chlorophyll concentrations for all three groups was calibrated against surface chlorophyll *a* measured using an acetone extraction procedure (Arar and Collins, 1997) carried out by NIWA (on contract to BoPRC). Model parameters were adjusted manually using a trial and error approach with values set to within literature ranges (e.g., Schladow and Hamilton, 1997; Trolle et al., 2008). The model error, represented by the root-mean-square-error (RMSE) and Pearson correlation coefficient (R) for each output variable, was quantified after each simulation for which model parameter values were adjusted. Calibration continued until there was negligible improvement in RMSE and R values with repeated model simulations. Root-mean-square-error and R values were also compared to modelling studies in the literature to assess an acceptable model error for prediction purposes.

A TLI value was calculated for each year of the simulation period. The relevant equations for determination of the TLI are:

$$TL_{Chla} = 2.22 + 2.54 log(Chla) \quad (11)$$

$$TL_{SD} = 5.1 + 2.27 log\left(\frac{1}{SD} - \frac{1}{40}\right) \quad (12)$$

$$TL_{TP} = 0.218 + 2.92 log(TP) \quad (13)$$

$$TL_{TN} = -3.61 + 3.01 log(TN) \quad (14)$$

$$TLI = \frac{1}{4}\sum(TL_{Chla}, TL_{SD}, TL_{TP}, TL_{TN}) \quad (15)$$

where:

TL_{Chla}, TL_{SD}, TL_{TP} and TL_{TN} represent the individual level trophic level indices for the individual variables of chlorophyll *a*, Secchi depth, total phosphorus and total nitrogen.

As Secchi depth is not explicitly included in the model, this variable was derived from a model-predicted attenuation coefficient as:

$$z_{SD} = \frac{\alpha}{K_d} \tag{16}$$

where:

z_{SD} is the Secchi depth (m)
K_d is the diffuse attenuation coefficient (m^{-1})
α is a constant (1.85 in this study) determined by comparing field measurements of Secchi depth with corresponding values of K_d (RMSE = 0.859 m).

In CAEDYM, K_d is calculated as:

$$\mathrm{K_{d(CAEDYM)}} = \mathrm{K_w} + \sum(\mathrm{K_a chl}a) + \mathrm{K_{POC}POC} + \mathrm{K_{DOC}DOC} \tag{17}$$

where:

K_w is the background extinction coefficient
K_a is the specific attenuation coefficient for chl *a* for each simulated phytoplankton group (a)
K_{POC} and K_{DOC} are the specific attenuation coefficients for particulate (POC) and dissolved organic matter (DOC), respectively.

TLI output from the model was compared with observed data and calibration of parameters was undertaken in DYRESM-CAEDYM until a satisfactory match was achieved. We aimed to calibrate the model TLI within ± 0.1 TLI units of the measured TLI. The final model parameters from the calibration were then fixed for model validation over the period July 2001 – June 2005.

2.8 Scenarios

2.8.1 Historical and future simulations based on past land use and no modification to future land use (R0)

Separate scenarios with different climates were run for eight-year periods at intervals between 1921 and 2099 including the calibration and validation period of 2001–2009 (Table 2). These scenarios used nutrient loads commensurate with observed land use up to 2001–2009 and then maintained nutrient loads at 2001–2009 until 2099 (Table 2). Meteorological data used in the simulations of two historical (1921–1929, 1971–1979) and three future (2031–2039, 2061-2069, 2091–2099) climates used measured data from 2001–2009 except for air temperature and rainfall, which were perturbed based on the 'A1B' climate change scenario (International Panel for Climate Change (IPCC): http://www.ipcc.ch/ipccreports/tar/wg1/029.htm#storya1). The global climate model (GCM) used to generate the A1B greenhouse gas climate scenario was the CSIRO MK3 version. To assess the impact of climate change on the water quality of Lake Rotorua, a variation of the baseline 'R0' scenario was simulated, without the DARLAM/CLIMPACTS5 A1B perturbation of the baseline climate data (i.e., 'no climate change' in which 2001-2009 meteorological data were applied to each of the five time periods (1921–1929, 1971–1979, 2031–2039, 2061-2069 and 2091–2099). The simulations of historical scenarios did not include lake level management (i.e. Ohau Channel capacity increased 1972, gabions placed at inlet of Ohau Channel 1974, Okere Gates installed October 1982, Ohau stoplog weir installed September 1989). Instead, we assumed that the change in lake storage volume, ΔS, would be the same for all six 8-year periods.

Globally-averaged temperature in the A1B scenario is expected to warm by 2.8 °C (range 1.7–4.4 °C) by the end of the 21st century under an assumption that there will be rapid economic growth, reliance on technological controls for control of climate change, and limited environmental quality initiatives. The A1B scenario produces an increase in mean annual air temperature of 2.27 °C for the Bay of Plenty region by 2100. This scenario is generally acknowledged to provide a mid-range projection of a future climate. The perturbation of the measured (2001–2009) air temperature and rainfall data involved an assumption that the 2001–2009 climate was not substantially different from that of 1961–1990; a period normally taken to be a 'baseline' for Global Climate Model (GCM) simulations. Whilst globally there have been obvious warming trends from the 1960s to 2010 (see Hansen et al. 2010) there appears to have been little change in air and water temperature of Rotorua and the lake, respectively, at least in the period of 1967–2002 (Hamilton et al. 2005; Hamilton et al. in press) and therefore changes between 1960 and 2010 are small compared with those that are predicted to occur later in the 21st century.

The output of the CSIRO MK3 GCM is in 0.5 degree latitude x longitude grids. These data were interpolated and downscaled with the MAGICC/SCENGEN model (Wigley, 2008). MAGICC was applied using the output from the CSIRO MK3 GCM A1B scenario as input, to interpolate data from 1920 to 2100. SCENGEN was then applied to produce spatial change patterns, expressed for air temperature as change per 1 ^{o}C change in global-mean temperature. In its latest version, the MAGICC/SCENGEN model provides the latest climate change information used in IPCC AR4 assessment. For any location, a future climate change scenario (i.e., the modified temperature) can be expressed as:

$$T_{year}(x,y) = T_{base}(x,y) + T_{GCM}(x,y) \times \Delta T_{year} \tag{18}$$

$$P_{year}(x,y) = P_{base}(x,y) \times (1 + P_{GCM}(x,y) \times \Delta T_{year}) \tag{19}$$

$$RH_{year}(x,y) = RH_{base}(x,y) \times (1 + RH_{GCM}(x,y) \times \Delta T_{year}) \tag{20}$$

where:

x and y are the longitude and latitude of the location
year is the future climate change year (up to 2100)
base indicates the baseline value
GCM is the normalized change
T is temperature
P is rainfall
RH is relative humidity
ΔT is global temperature change.

The pattern scaling method was used in constructing the future climate change projection for Lake Rotorua. The method assumes that the global mean responses of a GCM can be represented accurately by a simple climate model, even though the response may be non-linear (Raper et al., 2001), and a wide range of climate variables simulated by a GCM can be represented by linear functions of the global mean annual mean temperature change from GCMs, at different spatial and/or temporal scales (Mitchell, 2003). In order to create a 'business as usual' projection for lake Rotorua, a mid-range GCM projection for Lake Rotorua was simulated with CSIRO_30 and a mid-range greenhouse gas emission scenario, i.e. SRES A1B under mid-climate sensitivity was used to generate the projections for the selected climate variables for the selected time periods.

Rainfall followed a similar methodology to air temperature in order to create a combined modified air temperature and rainfall input to DYRESM-CAEDYM for each time period. The loads of N and P arising from atmospheric deposition were held constant for all time periods (1921–1929, 1971–1979, 2031–2039, 2061-2069 and 2091–2099). In order to maintain constant loads, however, concentrations of nitrate and phosphate in rainfall were modified for each period as rainfall amount varied for each time period.

Surface inflow temperature was estimated using the method described in equations 5 and 6, using the appropriate air temperature data for each simulation period and parameters obtained from the calibration. Surface inflow volumes and nitrate concentrations of all inflows for all six periods between 1921 and 2099 were obtained from output of ROTAN (Rutherford et al., 2011). Inflow concentrations of ammonium, phosphate and organic N and P for all six periods were the same linearly-interpolated BoPRC observations as those used in the 2001 – 2009 period. This assumption was based on analysis of data for this period, which showed no significant change in concentrations for nutrient species other than nitrate in the Lake Rotorua catchment.

Table 2. Summary of model input adjustments for the six simulation periods under the baseline 'R0' model scenario.

Period	Climate data	Flow	Temperature	DO	NO3-N	NH4-N	Organic N	PO4-P	Organic P
1921-1929	Rotorua airport station 2001-2009	ROTAN output 1921-1929	Estimated from air temperature	Equation 6	ROTAN output 1921-1929	Linear interpolation of BoPRC data 2001-2009	As for 2001-2009	Linear interpolation of BoPRC data 2001-2009	Equation 7.2
1971-1979	Rotorua airport station 2001-2009	ROTAN output 1971-1979	Estimated from air temperature	Equation 6	ROTAN output 1971-1979	Linear interpolation of BoPRC data 2001-2009	As for 2001-2009	Linear interpolation of BoPRC data 2001-2009	Equation 7.2
2001-2009	Rotorua airport station 2001-2009	ROTAN output 2001-2009	Estimated from air temperature	Equation 6	ROTAN output 2001-2009	Linear interpolation of BoPRC data 2001-2009	Equation 7.1	Linear interpolation of BoPRC data 2001-2009	Equation 7.2
2031-2039	DARLAM adjusted	ROTAN output 2031-2039	Estimated from air temperature	Equation 6	ROTAN output 2031-2039	Linear interpolation of BoPRC data 2001-2009	As for 2001-2009	Linear interpolation of BoPRC data 2001-2009	Equation 7.2
2061-2069	DARLAM adjusted	ROTAN output 2061-2069	Estimated from air temperature	Equation 6	ROTAN output 2061-2069	Linear interpolation of BoPRC data 2001-2009	As for 2001-2009	Linear interpolation of BoPRC data 2001-2009	Equation 7.2
2091-2099	DARLAM adjusted	ROTAN output 2091-2099	Estimated from air temperature	Equation 6	ROTAN output 2091-2099	Linear interpolation of BoPRC data 2001-2009	As for 2001-2009	Linear interpolation of BoPRC data 2001-2009	Equation 7.2

2.9 Sediment parameterisation

The ecological model CAEDYM has a relatively simple process representation of sediment nutrient dynamics, but has nevertheless proven remarkably successful in reproducing seasonal changes in concentrations of nutrients in lake bottom waters induced from sediment releases (e.g. Burger et al., 2008; Trolle et al., 2008; Özkundakci et al., 2011; Trolle et al. 2011a). The model regulates the sediment phosphate and ammonium releases according to concentrations of dissolved oxygen and temperature in the overlying water layer. Changes in sediment nutrient releases in the model thus reflect only the relevant water layer variables and not any changes in sediment N or P content or other processes that might affect releases. Changes in external nutrient loading, however, will likely feed back to changes in sediment oxygen demand and nutrient releases, leading to a new equilibrium between external and internal loading (Søndergaard et al., 2003). In the case of Lake Rotorua, Rutherford et al. (1996) calculated a time for near-complete equilibrium of sediments of up to 195 years but predicted quite rapid and substantial reductions in sediment nutrient releases because of reduced loading and deoxygenation in the water column with rapid water column responses to changes. The lack of such a feedback between the water column and the bottom sediment composition in the current version of CAEDYM has been described previously but has never been adequately accounted for (Trolle et al., 2010; Özkundakci et al., 2011). In this model application, and based on the considerations of Rutherford et al. (1996), we sought to account for the effect of external nutrient loads on sediment oxygen demand and nutrient release by adjusting relevant sediment parameters in CAEDYM based on changes in total lake external N loading derived from the ROTAN model (Figure 7). Our approach is further validated by results from Trolle et al. (2010) who noted moderate reductions in nutrient concentrations of bottom sediments in Lake Rotorua between 1995 and 2006.

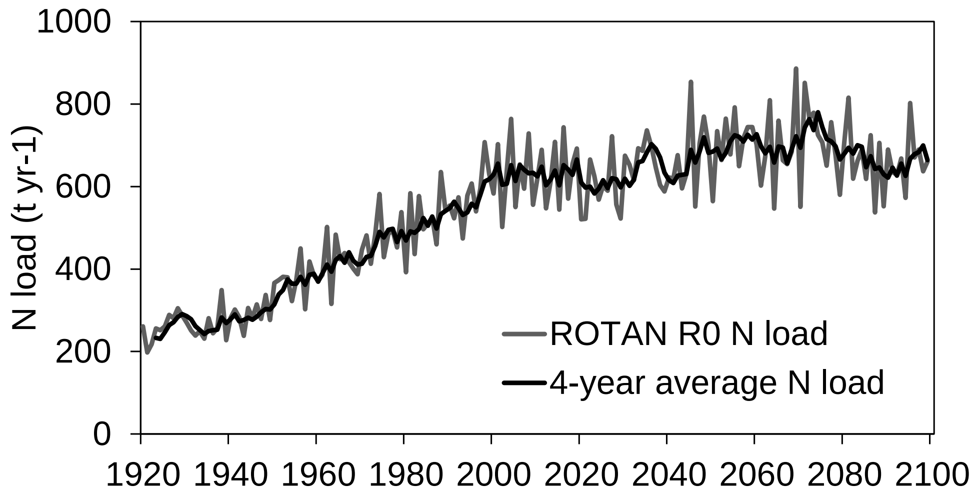


Figure 7. Annual N load and 4-year average N load (t yr-1) to Lake Rotorua based on ROTAN R0 (present land use scenario) output.

We used empirical relationships between N loads and measured values of volumetric hypolimnetic oxygen demand (VHOD; Vant, 1987). Figure 8A shows a linear and an exponential relationship between measured VHOD values (Vant, 1987) and the preceding four-year average annual N load to Lake Rotorua. The exponential function was used to describe the relationship between these two variables based on the assumption that the VHOD would never be zero for any given N load > 0 t yr^{-1}. However, this model would likely overestimate VHOD with increasing N loads. Therefore, the linear model was adopted to predict VHODs for N loads > 390 t yr^{-1} (i.e., an N load corresponding to the intercept of the linear model on the x-axis) and the exponential model to predict VHODs for N load ≤ 390 t yr^{-1}. We assumed that the sediment oxygen demand (SOD) parameter in CAEDYM was proportional to VHOD as modelled using the empirical relationship described above. Figure 8B shows that the preceding 4-year average N load is linearly related to the measured sediment N concentration, indicating that four years provides a duration on which the sediments may respond to changes in external loading.

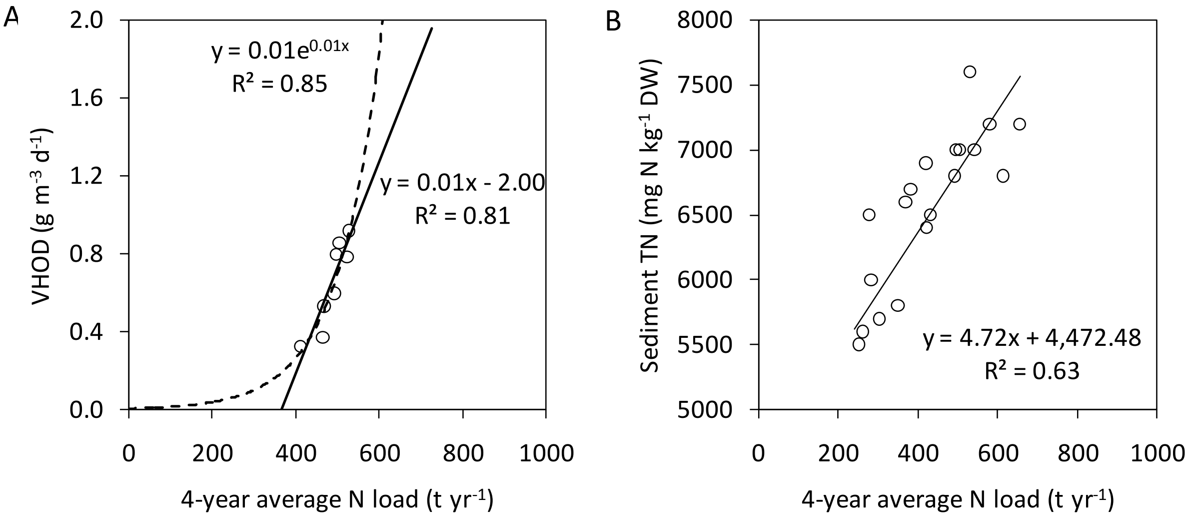


Figure 8. (A) Linear and exponential relationships between VHOD and 4-year average of annual N load to Lake Rotorua. (B) Linear relationship between sediment N concentrations and 4-year average N loads.

Sediment N concentrations for earlier time periods were derived from measured particulate N concentrations of horizontal layers of dried cores (see Trolle et al. 2010). The age of the layer was estimated in terms of the time since the Tarawera eruption (1886), based on the layer height above the tephra divided by the total sediment depth above the tephra, and then multiplied by the total number of years since the eruption. It was then possible to compare the sediment total N concentration against the 4-year average N load for that time period, and derive a linear relationship which could be used to estimate sediment TN concentrations from 4-year average N load for each year simulated in DYRESM-CAEDYM. We assumed that the maximum potential ammonium release rate would be directly proportional to the predicted sediment N concentration (Figure 9), which in turn was related to the predicted 4-year average N load (Figure 8). Phosphate release rates were calculated relative to ammonium releases, assuming that the ratio of nitrogen and phosphorus in lake sediments would remain constant over the entire period 1921 – 2099 (Figure 10).

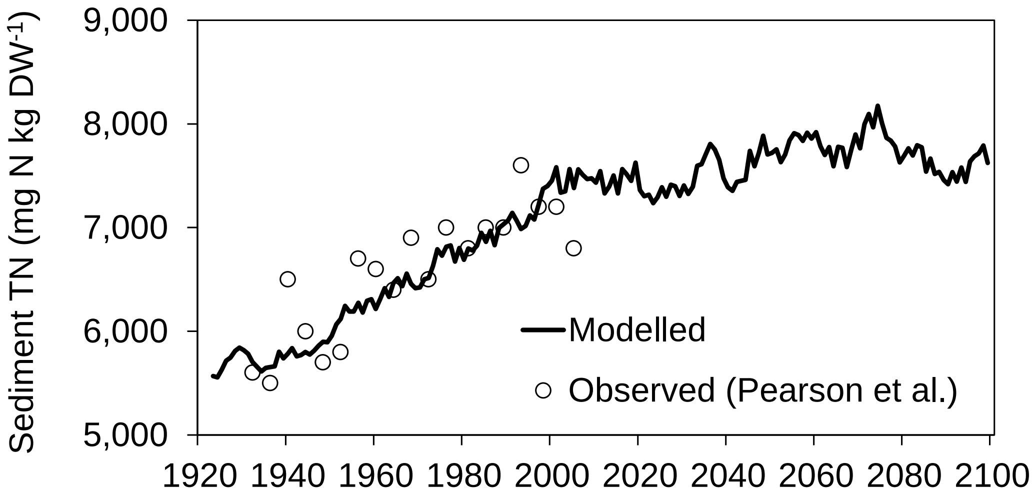


Figure 9. Time series of sediment N concentrations (modelled) and observed values from Pearson (2007).

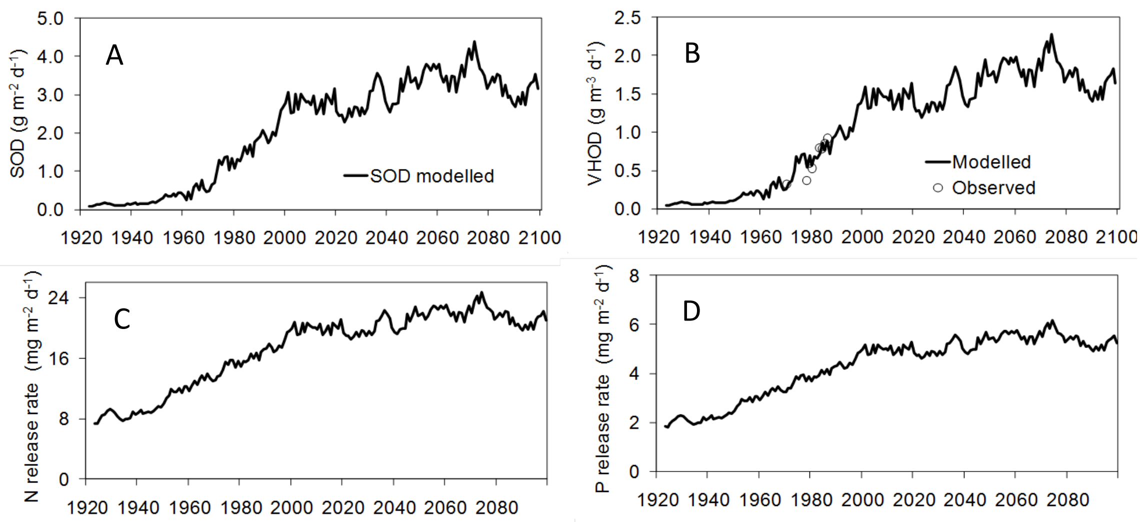


Figure 10. Time series of derived (A) sediment oxygen demand (SOD), (B) volumetric hypolimnetic oxygen demand (VHOD), (C) P release and (D) N release using empirical relationships between measured values of VHOD, sediment N concentrations and 4-average N load. Detailed descriptions of derivation methods are given in the text.

2.10 Lake restoration scenarios

Lake water quality was modelled for three scenarios of land use change over the future simulation periods between 2031 and 2099. These land use changes were assumed to take effect in 2015 and hence external nutrient load to the lake as well as internal load via sediment oxygen demand and nutrient releases, were assumed to have reached equilibrium by the beginning of the first simulation period in 2031. Water quality was quantified by using model output to calculate TLI for each year simulation, as described in equations 11 to 15.

The three scenarios simulated were termed R250, R300 and R350. In each case, the number represents the reduction in external nitrate load in t yr^{-1} on the land, with associated loads to the lake as predicted by ROTAN. Inflow volumes also changed under each land use scenario simulated with ROTAN. Details of the methods used to derive flow rates and stream inflow nitrate loadings can be found in Rutherford et al. (2011). Ammonium and organic N were assumed to remain constant for all periods simulated as there is no clear evidence that these variables are changing in the dominant groundwater-dominated coldwater inflows to Lake Rotorua (Rutherford et al., 2009). A separate water balance and, therefore, lake outflow were calculated for each scenario in order to maintain an identical lake level regime to that observed between 2001 and 2009. The control gates operated by BoPRC at the lake outlet at Ohau Channel exert the primary control on water levels in Lake Rotoiti.

2.11 Catchment Land Use for Environmental Sustainability (CLUES)

Changes in external phosphorus loads to Lake Rotorua were simulated using the CLUES model. CLUES is a tool for assessing the effects of land use on stream water quality including nitrogen and phosphorus, at a minimum scale of subcatchment. Modified land use scenarios identical to those applied in ROTAN, were input to CLUES, which then produced output of total phosphorus loading in t yr^{-1} for the entire catchment. The phosphorus load for each scenario was expressed as a fraction of the R0 scenario, and this was applied as a multiplier to the R0 phosphorus load, in order to obtain the load appropriate for each of the three land use scenarios (R250, R300 and R350, Table 3). For all land use change scenarios, the ratio of phosphate to total phosphorus under the R0 scenario was held constant for each sub-catchment. A more detailed specific-specific model of phosphorus dynamics than currently exists in CLUES would be required to adopt a different approach to phosphorus speciation, and was beyond the scope of the present study.

Table 3. Annual mean nutrient loads and discharge for the period 2031 – 2039 (following land use change) for different land use scenarios (Ro, R250,R300 and R350; see text).

Inflow load for scenario	NH_4	NO_3	Organic N	PO_4	Organic P	Flow
			t yr^{-1}			m^3 yr^{-1}
R0 (2031 - 2039)	78.6	706.6	51.1	23.7	6.7	5.146E+08
R250 (2031 - 2039)	74.7	570.1	48.9	20.4	5.8	4.986E+08
R300 (2031 - 2039)	75.8	555.1	49.1	21.5	6.1	5.012E+08
R350 (2031 - 2039)	72.6	515.9	48.0	17.8	5.1	4.903E+08

2.12 Additional scenarios

2.12.1 Hamurana spring diversion

A scenario was simulated that included diversion of the Hamurana inflow away from the lake and directly into the outflow at the Ohau Channel. In this case all flow from Hamurana was removed and added to the lake outflow. Therefore, the nutrient load contributed from Hamurana was removed entirely. This diversion was assumed to be implemented in 2015, along with any imposed land use change.

2.12.2 Sediment management

A scenario of sediment management in addition to the R350 land use changes was simulated to assess the synergistic effects of internal nutrient load management alongside external load reduction. In this scenario, a single application of a phosphorus binding sediment 'capping' agent on 01 July 2031 was simulated by adjusting internal phosphate release from lake sediments within CAEDYM. The efficacy of capping agents in reducing phosphorus release has been shown to decline on time-scales between days to years (Gibbs et al., 2010). We developed a relatively simple sediment capping decay function which describes the decay of sediment capping efficacy:

$$E_{(t)} = E_{(0)}(1-r)^t \qquad (20)$$

where:

E(t) is the remaining sediment capping efficacy at time t
E(o) is the initial sediment capping efficacy
r is decay constant

The decay rate constant was derived from an assumed half-life:

$$t_{1/2} = \frac{ln2}{r} \qquad (21)$$

In CAEDYM, sediment phosphorus release is modelled as maximum potential release rate which is controlled by changes in temperature and oxygen concentration in the bottom-water layer adjacent to the sediments. Figure 11 shows the decay of P uptake efficacy of the sediments following a single sediment capping application, carried out on 1 July 2031 and assumed to initially block 100% of the sediment P release. Three simulations of half life are shown, whereby the uptake capacity of a sediment capping agent would be exhausted: (1) after 4 years, (2) after many years and (3) after 117 days. The method described here assumes that no phosphate from the sediment below the sediment capping layer could be released in the future. It does, however, take into account the effect of slow burial with a sediment containing particulate P which could be mobilised under anoxic conditions (Max Gibbs, pers. comm.). In order to simulate this decay function, the model was run for individual years within the period 2031-2039 using the adjusted release rate corresponding to reduced phosphorus retention efficacy by the bottom sediments (Figure 12). The model was re-initialised for each year using model output from the last day of the previous year's simulation for all parameters. Based on likely organic matter deposition rates (Burger 2006), the sediment retention efficacy was assumed to be exhausted after four years. Therefore, no ongoing suppression of phosphate release was included in the subsequent simulation period of 2061 – 2069 or beyond.

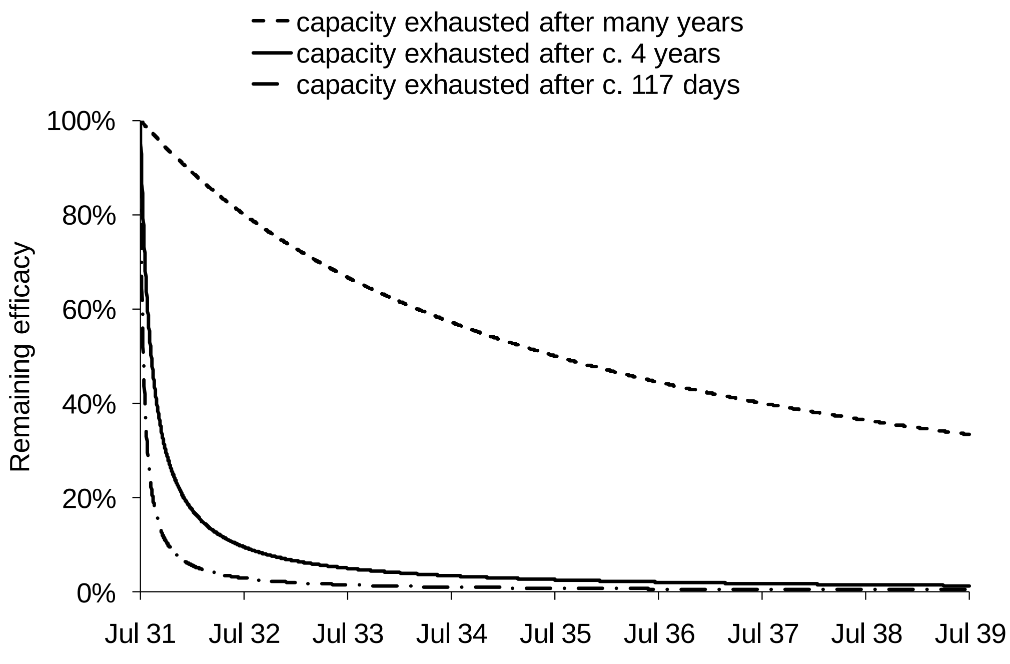


Figure 11. Simulation results for sediment capping efficacy decay rates based on three different half lives. The "capacity exhausted" refers to a time scale on which the retention efficiency for phosphorus of the treated sediments is largely exhausted.

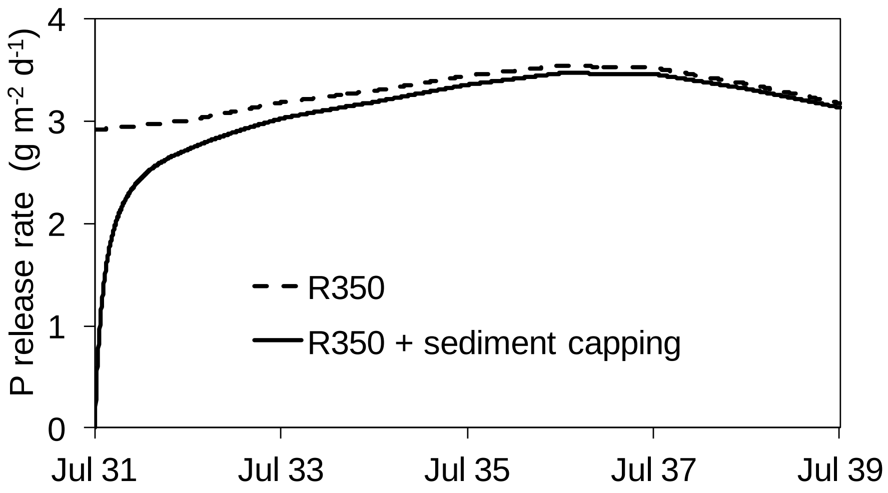


Figure 12. Maximum potential sediment phosphate release rate following sediment capping application on 01 July 2031 (solid line). The dashed line shows phosphate release rate for the R350 scenario without sediment capping, for comparison. Changes in the dashed line reflect the effects of land use change in sediment phosphate release rates.

2.12.3 Additional land use phosphorus mitigation scenarios

Additional scenarios were undertaken based on calculation of phosphorus mitigation scenarios (see Appendix 1). Based on the assigned export coefficients for land use, various P loads for the catchment were determined. For each P-mitigation scenario, the percentage P load reduction was calculated relative to the baseline (R0) case. This percentage was then applied to the P load for the R0 model scenario (Tables 4 & 5). Sediment P release rates were also adjusted in proportion to the percentage adjustments of the original nutrient management scenarios. The model simulations assumed that there was equilibrium between the (mitigated) external P loads and lake sediment P concentration/release.

Table 4. Phosphorus external loads for various scenarios, including additional P mitigation (refer to text). Loads are given as both absolute (upper section) values as well as relative to the R0 baseline case (lower section). 50-50-50 and 50-30-30 represents percentage P load mitigation for Dairy-Drystock-Lifestyle land uses respectively.

Scenario	R0	R-P	R250	R300	R350
	t P yr^{-1}	t P yr^{-1}	t P yr^{-1}	t P yr^{-1}	t P yr^{-1}
CLUES	30.4	n/a	26.2	27.6	22.8
SP 50-50-50	39.5	28.9	27.4	27.9	26.6
SP 50-30-30	39.5	31.8	29.6	30.7	28.8
	% of R0	% of R0	% of R0	% of R0	% of R0
CLUES	100	n/a	86	91	75
SP 50-30-30	-	80	75	78	73
SP 50-50-50	-	73	69	71	67

Table 5. Phosphorus external loads and sediment nutrient release rates for the R0 and R350 scenarios, and for two scenarios of additional on-land P mitigation that were selected for analysis (refer to section 2.12.3). 50-50-50 and 50-30-30 represents percentage P load mitigation for Dairy-Drystock-Lifestyle land uses respectively.

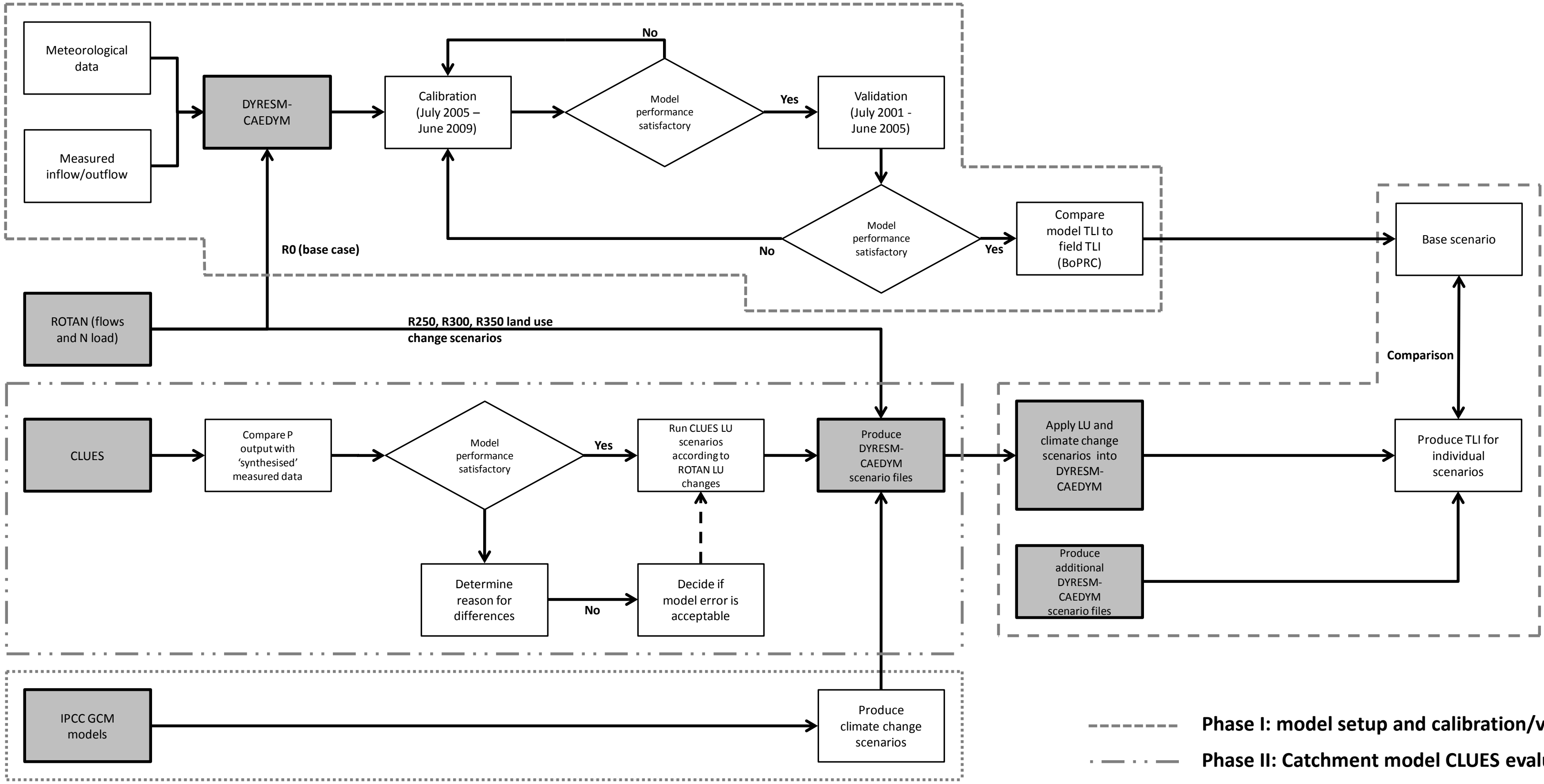
Scenario	TP load	PO4-P release	TN load	NH4-N release
	t yr^{-1}	g m^{-2} d^{-1}	t yr^{-1}	g m^{-2} d^{-1}
R0	30.4	0.0521	836.3	0.2081
R0 SP 50-50-50	22.2	0.0303	636.4	0.1309
R350	22.8	0.0327	636.4	0.1309
R350 SP-50-50-50	20.4	0.0257	836.3	0.2081

2.13 Comparison of scenarios

TLI values were calculated for all scenarios and compared with baseline TLI values from the R0 scenario. Frequency of deoxygenation events and cyanobacterial blooms was assessed for each scenario of land use, climate change, inflow diversion and/or sediment management.

Table 6. Model parameters used in CAEDYM for each simulated period.

Period	SOD (g m^{-3} d^{-1})					Maximum potential PO_4-P release rate (mg m^{-2} d^{-1})					Maximum potential NH_4-N release rate (mg m^{-2} d^{-1})				
	R0 (CC)	R0 (noCC)	R250	R300	R350	R0 (CC)	R0 (no CC)	R250	R300	R350	R0 (CC)	R0 (no CC)	R250	R300	R350
Land use change only															
1921-1929	0.133	0.134	0.133	0.133	0.133	20.56	20.64	20.55	20.55	20.54	82.26	82.55	82.19	82.20	82.17
1971-1979	1.098	1.098	1.098	1.098	1.098	37.20	37.20	37.20	37.20	37.20	148.82	148.82	148.82	148.82	148.82
2001-2009	2.808	2.808	2.808	2.808	2.808	50.00	50.00	50.00	50.00	50.00	200.00	200.00	200.00	200.00	200.00
2031-2039	3.078	3.475	1.211	0.933	0.570	52.02	54.99	38.05	35.97	32.73	208.09	219.97	152.21	143.88	130.93
2061-2069	3.400	3.532	1.176	0.829	0.394	54.43	55.42	37.79	35.19	31.47	217.73	221.68	151.17	140.76	125.88
2091-2099	3.111	3.122	0.859	0.522	0.348	52.26	52.35	35.41	32.72	29.02	209.06	209.39	141.66	130.87	116.07
Land use change including Hamurana diversion															
1921-1929	0.133	0.134	0.133	0.133	0.133	20.56	20.64	20.55	20.55	20.54	82.26	82.55	82.19	82.20	82.17
1971-1979	1.098	1.098	1.098	1.098	1.098	37.20	37.20	37.20	37.20	37.20	148.82	148.82	148.82	148.8	148.82
2001-2009	2.808	2.808	2.808	2.808	2.808	50.00	50.00	50.00	50.00	50.00	200.00	200.00	200.00	200.0	200.00
2031-2039	2.503	2.828	0.722	0.478	0.346	47.72	50.15	34.39	32.34	29.39	190.89	200.59	137.56	129.4	117.56
2061-2069	2.599	2.702	0.565	0.351	0.308	48.44	49.21	33.22	30.86	27.54	193.75	196.82	132.88	123.4	110.17
2091-2099	2.391	2.375	0.427	0.350	0.248	46.88	46.76	31.49	29.05	25.75	187.51	187.04	125.97	116.2	103.02



Phase I: model setup and calibration/validation
Phase II: Catchment model CLUES evaluation
Phase III: Climate change modelling
Phase IV: Scenario simulations

Figure 13. Conceptual schematic of model coupling and scenario calculation methods.

3. Results

3.1 Calibration and validation

The model parameters adjusted during the calibration of DYRESM are presented in Table 7 and for CAEDYM in Table 8. Parameter values were assigned mostly within the range found in the literature (e.g., Schladow and Hamilton 1997; Trolle et al. 2008, Özkundakci et al., 2011). The overall model performance was assessed statistically using R and RMSE values (Table 9), and these values suggest that the model was generally able to satisfactorily reproduce the magnitude and dynamics of field measurements. The statistical evaluation of the current Lake Rotorua model shows that the model performance has improved compared to an earlier model application (Burger et al., 2008). Visual comparisons of modelled temperature, dissolved oxygen, TN and TP concentrations with field measurements are shown in Figure 14 and comparisons of phosphate, ammonium, nitrate and chlorophyll *a* concentrations are shown in Figure 15.

Table 7. Assigned values for parameters used in DYRESM.

Parameter	Unit	Calibrated value	Reference/remarks
Critical wind speed	$m\ s^{-1}$	4.5	Spigel et al. (1986)
Emissivity of water surface	-	0.96	Imberger & Patterson (1981)
Mean albedo of water	-	0.12	Patten et al. (1975)
Potential energy mixing efficiency	-	0.25	Spigel et al. (1986)
Shear production efficiency	-	0.28	Spigel et al. (1986)
Vertical mixing coefficient	-	900	Yeates & Imberger (2003)
Wind stirring efficiency	-	0.5	Spigel et al. (1986)
Effective surface area coefficient	m^{-2}	1.0×10^{7}	Calibrated

Table 8. Assigned values for parameters used in CAEDYM for Lake Rotorua; DOPL and DONL are dissolved organic phosphorus and nitrogen, respectively.

Parameter	Unit	Calibrated value	Reference source
Sediment parameters			
Sediment oxygen demand	g m^{-2} d^{-1}	2.8	Schladow & Hamilton (1997)
Half-saturation coefficient for sediment oxygen demand	mg L^{-1}	0.4	Schladow & Hamilton (1997)
Maximum potential PO_4 release rate	g m^{-2} d^{-1}	0.05	
Oxygen and nitrate half-saturation for release of phosphate from bottom sediments	g m^{-3}	2.0	
Maximum potential NH_4 release rate	g m^{-2} d^{-1}	0.20	
Oxygen half-saturation constant for release of ammonium from bottom sediments	g m^{-3}	3.0	
Maximum potential NO_3 release rate	g m^{-2} d^{-1}	-0.1	
Oxygen half-saturation constant for release of nitrate from bottom sediments	g m^{-3}	0.5	
Temperature multiplier for nutrient release	-	1.05	Robson & Hamilton (2004)
Nutrient parameters			
Decomposition rate of POPL to DOPL	d^{-1}	0.001	Schladow & Hamilton (1997)
Mineralisation rate of DOPL to PO_4	d^{-1}	0.1	Schladow & Hamilton (1997)
Decomposition rate of PONL to DONL	d^{-1}	0.02	Schladow & Hamilton (1997)
Mineralisation rate of DONL to NH_4	d^{-1}	0.01	Schladow & Hamilton (1997)
Denitrification rate coefficient	d^{-1}	1.5	
Oxygen half-saturation constant for denitrification	mg L^{-1}	2.5	
Temperature multiplier for denitrification	-	1.08	
Nitrification rate coefficient	d^{-1}	0.1	
Nitrification half-saturation constant for oxygen	mg L^{-1}	1.0	
Temperature multiplier for nitrification	-	1.08	

Phytoplankton parameters		Diatoms, chlorophytes, cyanophytes	
Maximum potential growth rate at 20°C	d^{-1}	1.5, 1.0, 0.55	Robson & Hamilton (2004)
Irradiance parameter non-photoinhibited growth	µmol m^{-2} s^{-1}	20, 50, 380	Robson & Hamilton (2004)
Half saturation constant for phosphorus uptake	mg L^{-1}	0.004, 0.005, 0.006	Trolle et al. (2008)
Half saturation constant for nitrogen uptake	mg L^{-1}	0.018, 0.012, 0.01	Trolle et al. (2008)
Minimum internal nitrogen concentration	mg N (mg chl *a*)$^{-1}$	4.0, 3.0, 4.5	Schladow & Hamilton (1997)
Maximum internal nitrogen concentration	mg N (mg chl *a*)$^{-1}$	12.0, 12.0, 14.0	Schladow & Hamilton (1997)
Maximum rate of nitrogen uptake	mg N (mg chl *a*)$^{-1}$ d^{-1}	3.0, 3.5, 4.5	Schladow & Hamilton (1997)
Minimum internal phosphorus concentration	mg P (mg chl *a*)$^{-1}$	0.2, 0.15, 0.3	Schladow & Hamilton (1997)
Maximum internal phosphorus concentration	mg P (mg chl *a*)$^{-1}$	1.3, 1.5, 4.0	Schladow & Hamilton (1997)
Maximum rate of phosphorus uptake	mg P (mg chl *a*)$^{-1}$ d^{-1}	0.7, 0.35, 0.2	Schladow & Hamilton (1997)
Temperature multiplier for growth limitation	-	1.04, 1.04, 1.07	Schladow & Hamilton (1997)
Standard temperature for growth	°C	7.0, 10.0, 20.0	Gal et al. (2009)
Optimum temperature for growth	°C	8.0, 20.0, 22.0	Gal et al. (2009)
Maximum temperature for growth	°C	9.5, 35.0, 33.0	Gal et al. (2009)
Respiration rate coefficient	d^{-1}	0.07, 0.06, 0.051	Schladow & Hamilton (1997)
Temperature multiplier for respiration	-	1.08, 1.08, 1.04	Schladow & Hamilton (1997)
Fraction of respiration relative to total metabolic loss rate	-	0.9, 0.9, 0.6	
Fraction of metabolic loss rate that goes to DOM	-	0.05, 0.1, 0.01	
Constant settling velocity	m s^{-1}	-0.3×10^{-5}, -0.23×10^{-6}, 0.25×10^{-4}	Burger et al. (2007a)

Table 9. Statistical comparison between model simulations and field data (monthly measurements) of surface (0 m), 15 m depth and bottom (20 m) waters in Lake Rotorua using root-mean-square-error (RMSE), which has the same unit as the variable estimated and Pearson correlation coefficient (R) for each variable.

	Calibration (2005-2009)						Validation period (2002-2005)					
	Surface (0 m)		15 m		Bottom (20 m)		Surface (0 m)		15 m		Bottom (20 m)	
	R	RMSE	R	RMSE	R	RMSE	R	RMSE	R	RMSE	R	RMSE
Temperature (°C)	0.994	0.661	0.994	0.822	0.994	0.783	0.991	0.666	0.992	0.789	0.991	0.828
Dissolved oxygen (mg L^{-1})	0.603	1.638	0.879	1.295	0.994	1.166	0.784	1.869	0.763	1.921	0.741	1.988
Phosphate (mg L^{-1})	-0.051	0.005	0.309	0.008	0.302	0.013	-0.119	0.010	0.274	0.018	0.126	0.020
Ammonium (mg L^{-1})	0.124	0.029	0.403	0.049	0.341	0.110	0.158	0.032	0.011	0.104	0.181	0.135
Nitrate (mg L^{-1})	0.002	0.011	0.222	0.010	0.114	0.010	0.598	0.014	0.552	0.014	0.550	0.019
Total phosphorus (mg L^{-1})	0.412	0.016	0.302	0.016	0.282	0.024	0.289	0.018	0.421	0.024	0.187	0.027
Total nitrogen (mg L^{-1})	0.392	0.123	0.576	0.103	0.148	0.453	0.297	0.155	0.276	0.157	0.191	0.237
Chlorophyll *a* (µg L^{-1})	0.404	12.909	-	-	-	-	0.105	16.042	-	-	-	-

Table 10. Statistical comparison between model simulations and field data (buoy data) of surface (0 m) and bottom (20 m) waters in Lake Rotorua using root-mean-square-error (RMSE), which has the same unit as the variable estimated and Pearson correlation coefficient (R) for each variable.

	Calibration (2005-2009)			
	Surface (0 m)		Bottom (20 m)	
	R	RMSE	R	RMSE
Temperature (°C)	1.00	0.552	0.99	0.967
Dissolved oxygen (mg L^{-1})	0.76	0.2726	0.90	1.521

There were occasional discrepancies between model output and field measurements which were evident as low R and high RMSE values, respectively. For example, fit of the model to phosphate dynamics in surface waters was not as good as for other parameters, as reflected in low R values for both calibration and validation period. However, the RMSE values for both periods are relatively low, indicating that the magnitude of concentrations was captured reasonably well. A very similar pattern was found for the model output of surface nitrate concentrations. The R value for nitrate surface concentrations was relatively low (0.002) for the calibration period, but was as high as 0.589 for the validation period, while the RMSE values were low for both periods. The difference in R values is due to relatively high observed nitrate concentrations during the validation period, which were captured well by the model, but did not occur during the calibration period. The R values for chlorophyll *a*, TN and TP were generally higher and the corresponding RMSE values lower during the calibration period compared to the validation period.

Comparison of model output with high frequency data for temperature and dissolved oxygen concentration obtained from the monitoring buoy (Table 10, Figure 16) enabled the calibration of DYRESM-CAEDYM parameters according to the polymictic nature of Lake Rotorua. The adjustment of the physical parameters appreciably enhanced the model performance. Relatively short-lived stratification events followed by oxygen depletion in the bottom waters were captured more accurately compared to the calibration using monthly measurements only. These improvements supported a more accurate calibration of seasonal transitions of stratification, deoxygenation of bottom waters and subsequent build-up of bottom water concentrations of phosphate and ammonium. Parameter values for sediment processes in this study were similar to those of Burger et al. (2008), while parameter values for phytoplankton dynamics were slightly different. These differences were likely due to the increased complexity of the conceptual model (i.e. three phytoplankton groups) in this study. Chlorophyll *a* concentrations observed during late summer and early spring were represented in the model largely as contributions from the 'chlorophyte' group using parameters within literature ranges (Schladow and Hamilton, 1997; Trolle et al., 2008) while 'diatoms' were predominantly present during winter. The magnitude and timing of chlorophyll *a* concentrations during summer were reproduced by allowing 'cyanophytes' luxury uptake of P (Reynolds, 2006) which is reflected by a relatively high maximum internal P concentration (4.0 mg P mg^{-1} chl *a*) as observed in field studies in Lake Rotorua (Burger et al., 2007b). The model performance of reproducing chl *a* concentration based on statistical values was comparable to or better than other modelling studies (e.g., Arhonditsis and Brett 2004; Burger et al. 2007a; Trolle et al., 2010; Özkundakci et al., 2011).

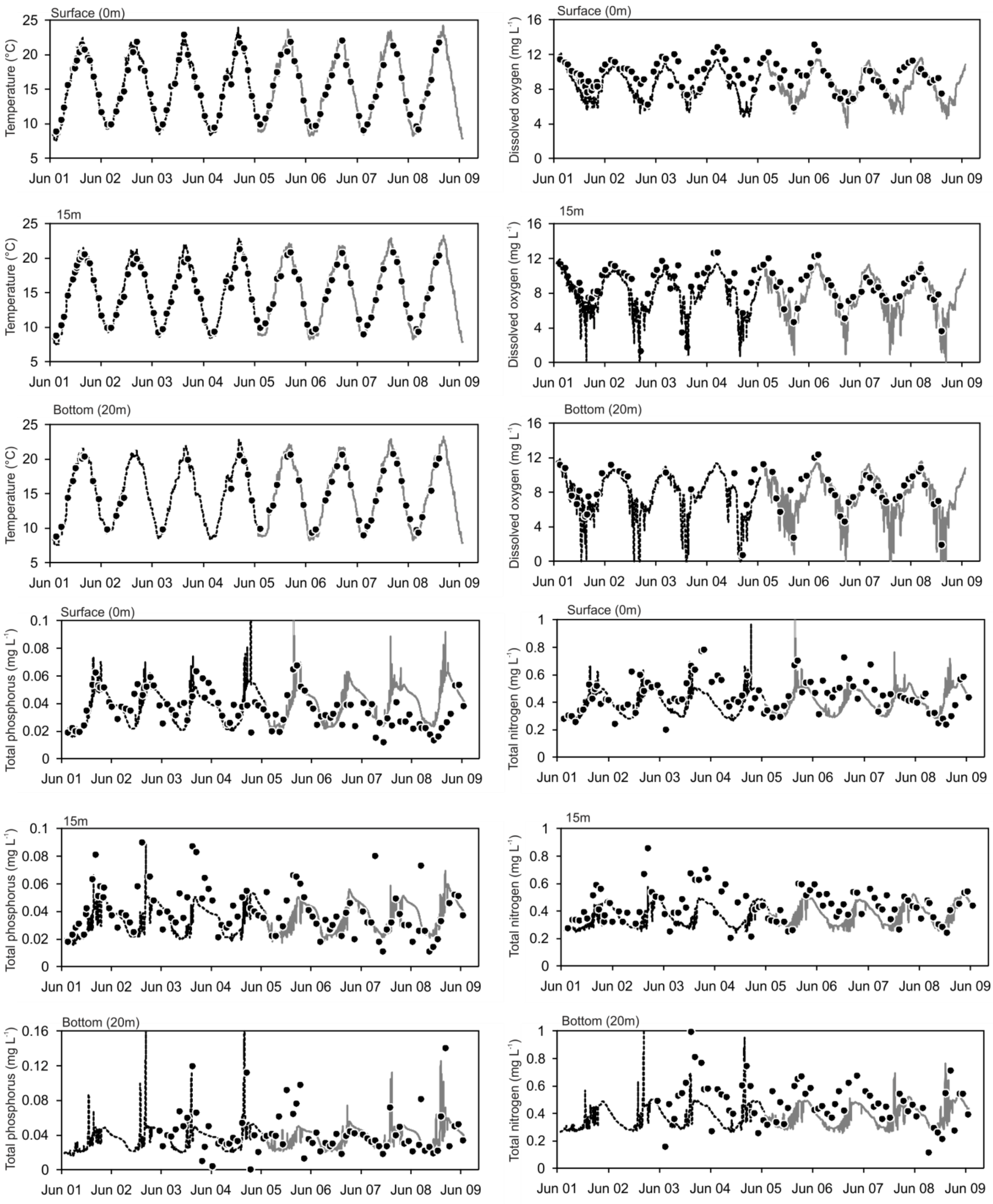


Figure 14. Comparison of model simulation results against field observations (black circles) in the surface (0 m), 15m depth and bottom (20 m) waters of Lake Rotorua during the calibration period (solid grey line) and validation period (dashed black lines) for temperature, dissolved oxygen, total phosphorus and total nitrogen.

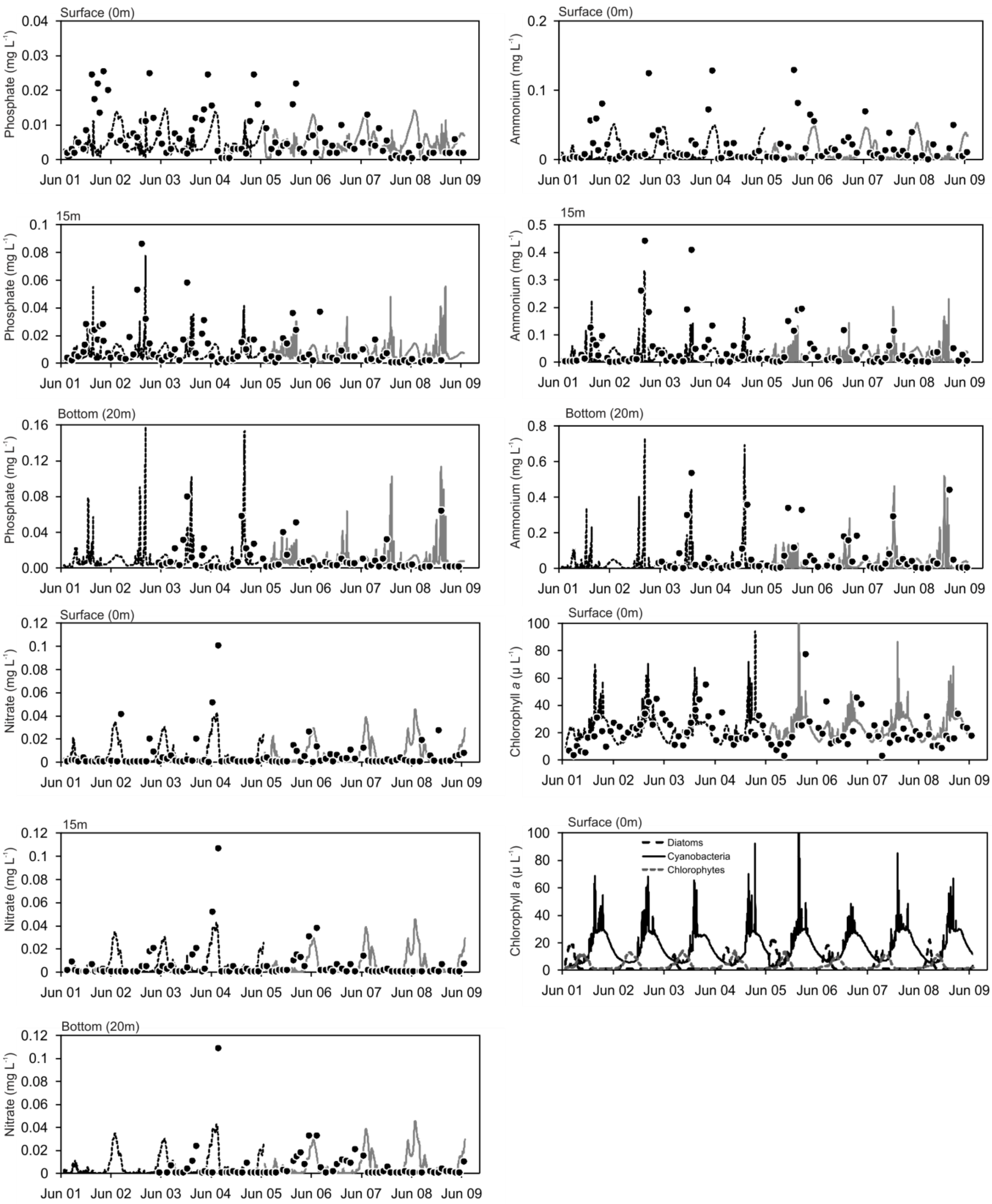


Figure 15. Comparison of model simulation results against field observations (black circles) in the surface (0 m), 15m depth and bottom (20 m) waters of Lake Rotorua during the calibration period (solid grey line) and validation period (dashed black lines) for phosphate, ammonium, nitrate and chlorophyll a.

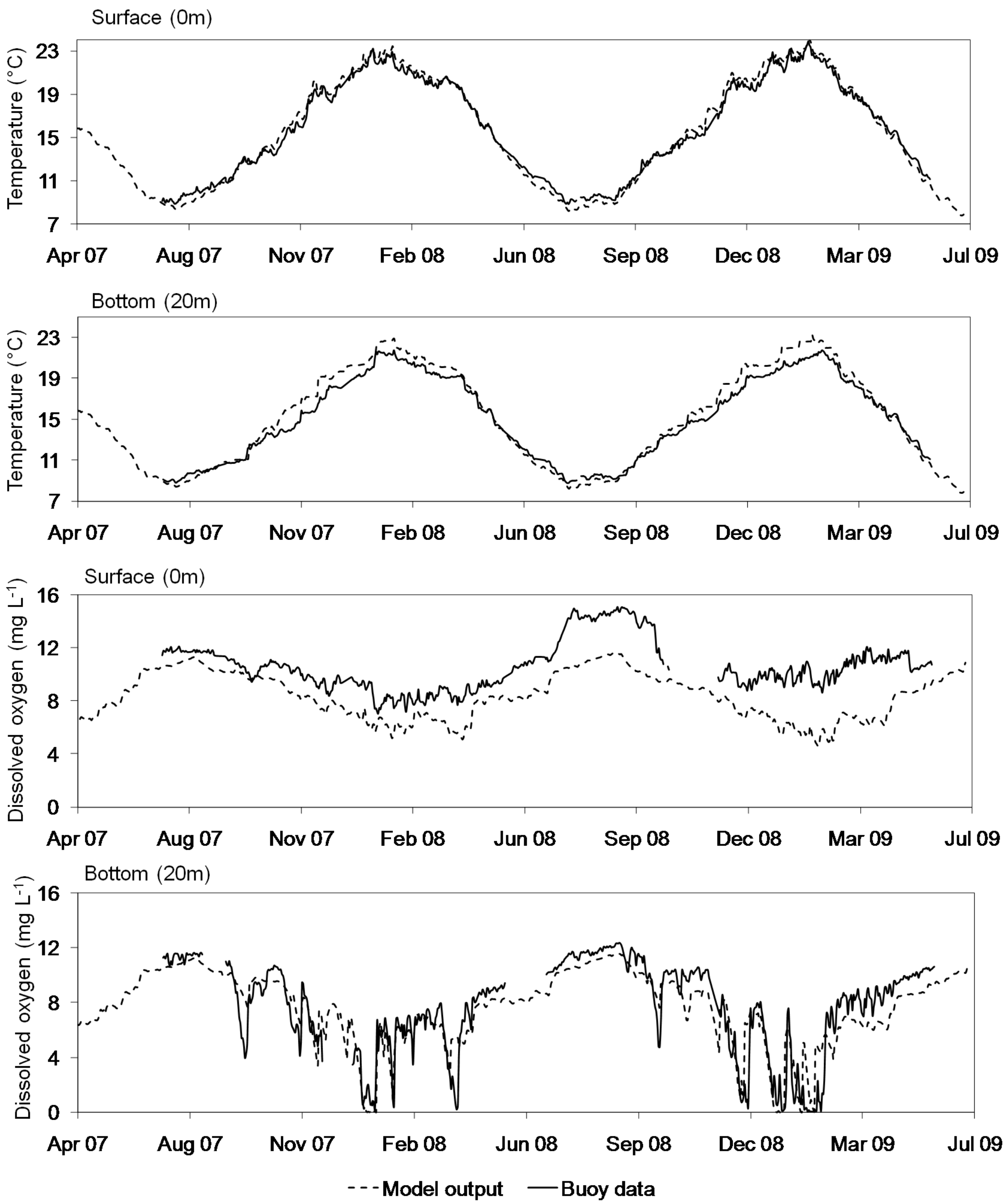


Figure 16. Comparison of model simulation (dashed line) results against high frequency observations (solid line) in the surface (0 m) and bottom (20 m) waters of Lake Rotorua for the period 2007 to 2009.

(A)

(B)

(C)

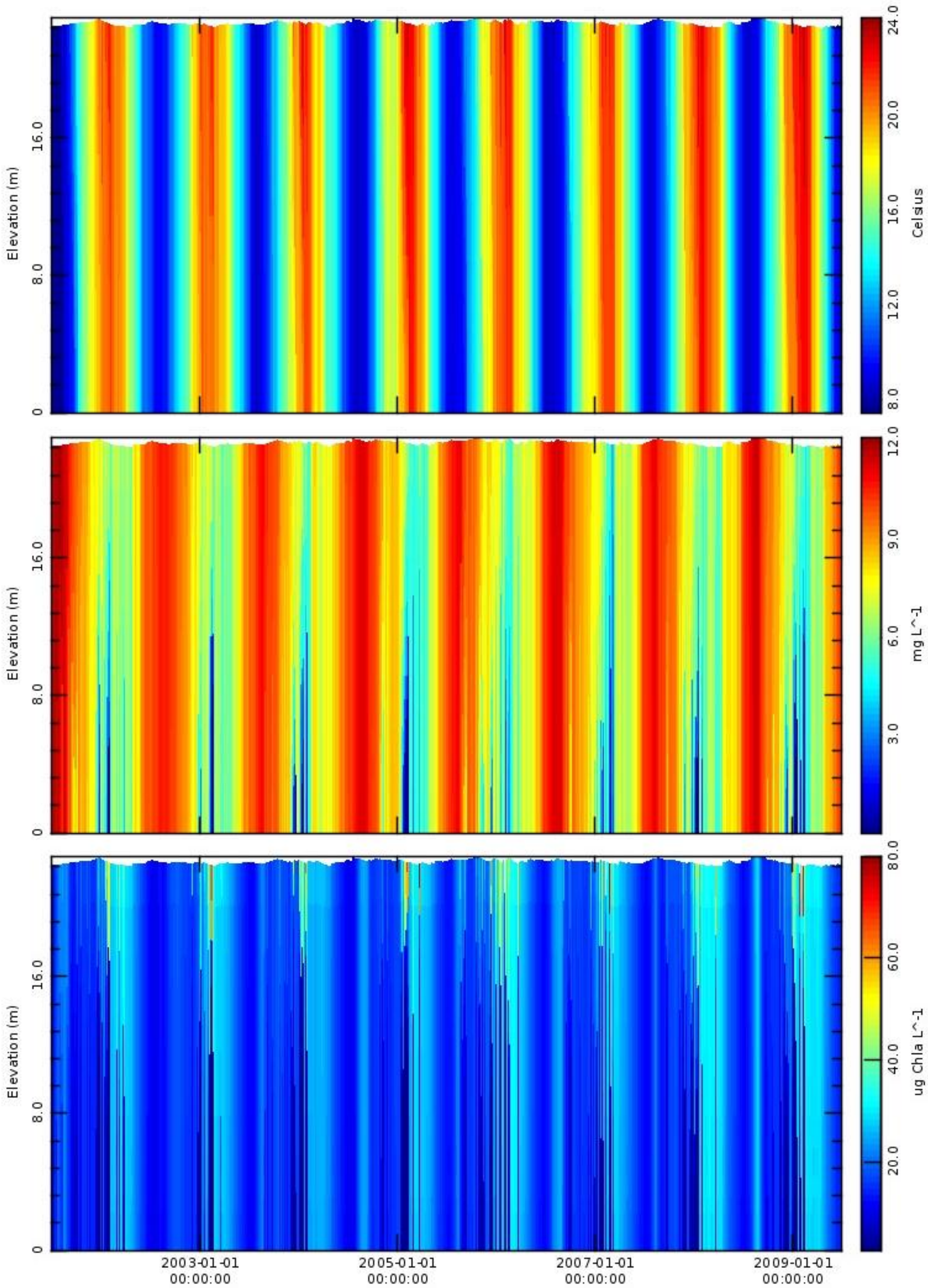


Figure 17. Time-elevation model output for (A) temperature, (B) dissolved oxygen concentrations and (C) total chlorophyll a concentrations for the calibration and validation period between 2001 and 2009. The vertical axes represent the water height from the lake bottom as a function of time on the horizontal axes. The plot colour represents the value of the model output as labelled to the right of the colour bar.

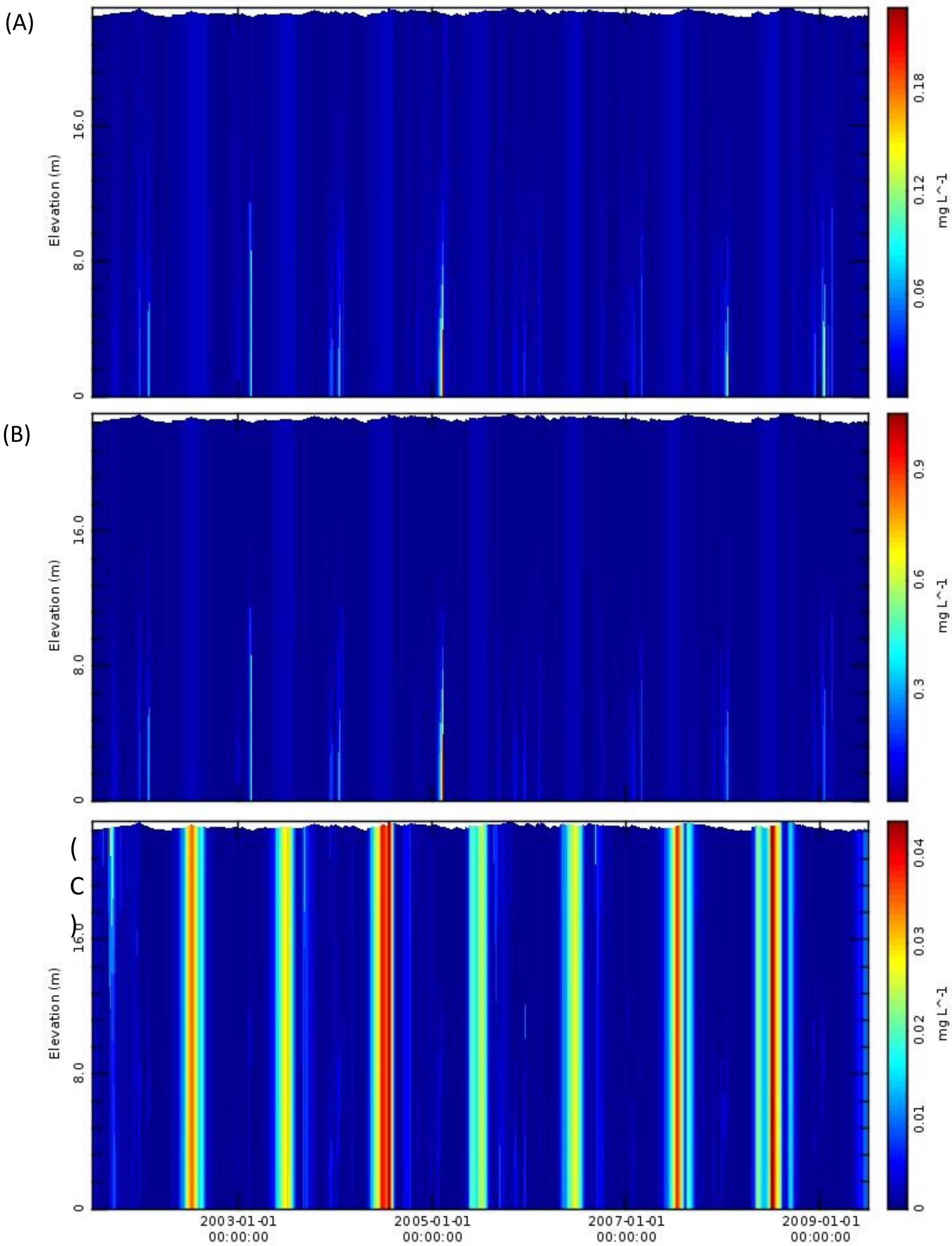


Figure 18. Time-elevation model output data for concentrations of (A) phosphate, (B) ammonium and (C) nitrate for the calibration and validation period between 2001 and 2009. The vertical axes represent the water height from the lake bottom as a function of time on the horizontal axes. The plot colour represents the value of the model output as labelled to the right of the colour bar.

Trophic Level Index (TLI) values derived from model output were aimed to be calibrated to within ±0.1 units of measured TLI values. Figure 17 shows that model output reproduced the measured TLI within this criterion, except for the years 2002 to 2004. This period was dominated by the invasive, highly buoyant and N-fixing cyanobacterium, *Anabaena planktonica* (Environment Bay of Plenty, unpublished data). The occurrence and potential impacts of *A. planktonica* are further discussed below.

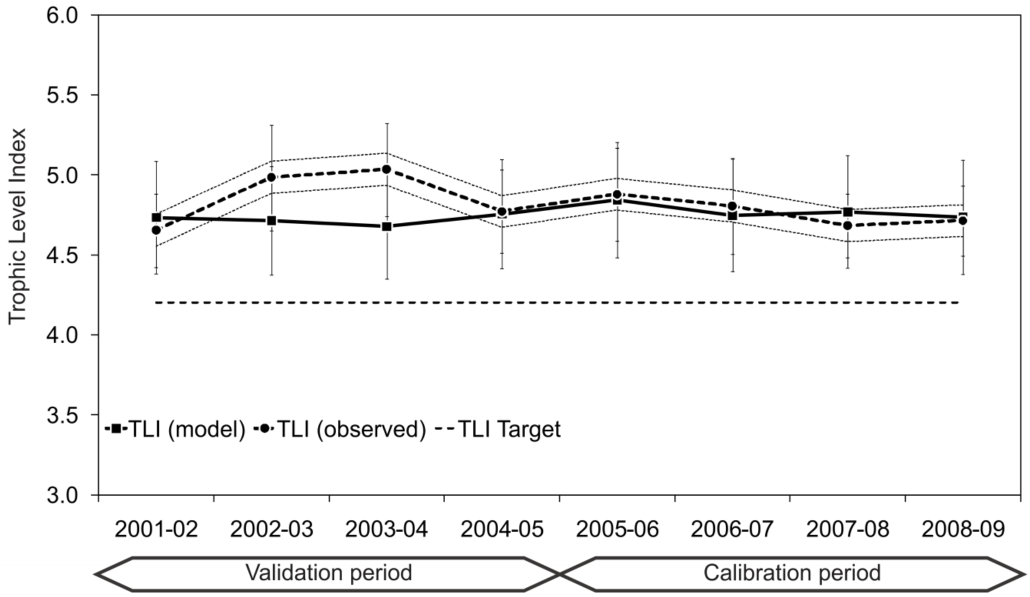


Figure 19. Measured and simulated Trophic Level Index values (with intra-annual variability represented by ± one standard deviation) compared with the base scenario (July 2002 - June 2009).

3.2 Base scenario

The base scenario (R0) spanned six separate 8-year periods of 1920-2099 (July-June). Simulations of historical periods agreed well with the (mostly anecdotal) information available on the historical TLI of just below the target value of 4.2 in the 1960s, which is considered to be the trophic state of Lake Rotorua preceding the effects of substantial intensification of agriculture in the catchment. The sediment parameterisation method as well as the external N loads based on ROTAN output appear to have contributed to the accuracy of historical simulations. TLI values were below 4.2 during the period 1921-1929 with a mean value of 4.04 (Figure 20). The simulations for the period 1971-1979 showed that the TLI increased to a mean value of 4.48 and then to a mean value of 4.74 for the calibration/validation period of 2001-2009. The TLI value in the base (R0) scenario was predicted to continue to increase to a mean value of 4.92 for the period 2091-2099 and occasionally exceeding 5.0, with a maximum value of 5.04 in 2095. The climate change scenario tested was shown to have a consistently negative effect on water quality from 2061 onwards, causing a mean increase of 0.13 TLI units for R0 by 2099 when compared to the same simulation with no climate change.

3.3 Scenarios for effect of reduced nutrient loading on TLI

The land use change scenarios R250, R300 and R350 showed that the TLI of Lake Rotorua would decrease but would remain above the 4.2 target TLI, with a mean of 4.28 for R350 during the period 2091-2099. The proportional reduction in TLI units appeared to be roughly proportional to that of the catchment loads for N and P. While the diversion of Hamurana springs, which would be equivalent to approximately 50 t N yr^{-1}, resulted in a further decrease of TLI for a given land use change scenario, TLI values remained above the TLI target with mean values by 2099 of 4.79 for R0, 4.35 for R250 (Hamurana diversion) and 4.29 for R300 (Hamurana diversion).

Only the diversion of Hamurana springs, in addition to R350 resulted in a mean TLI value of 4.19 for the period 2091-2099 (Figure 20c).

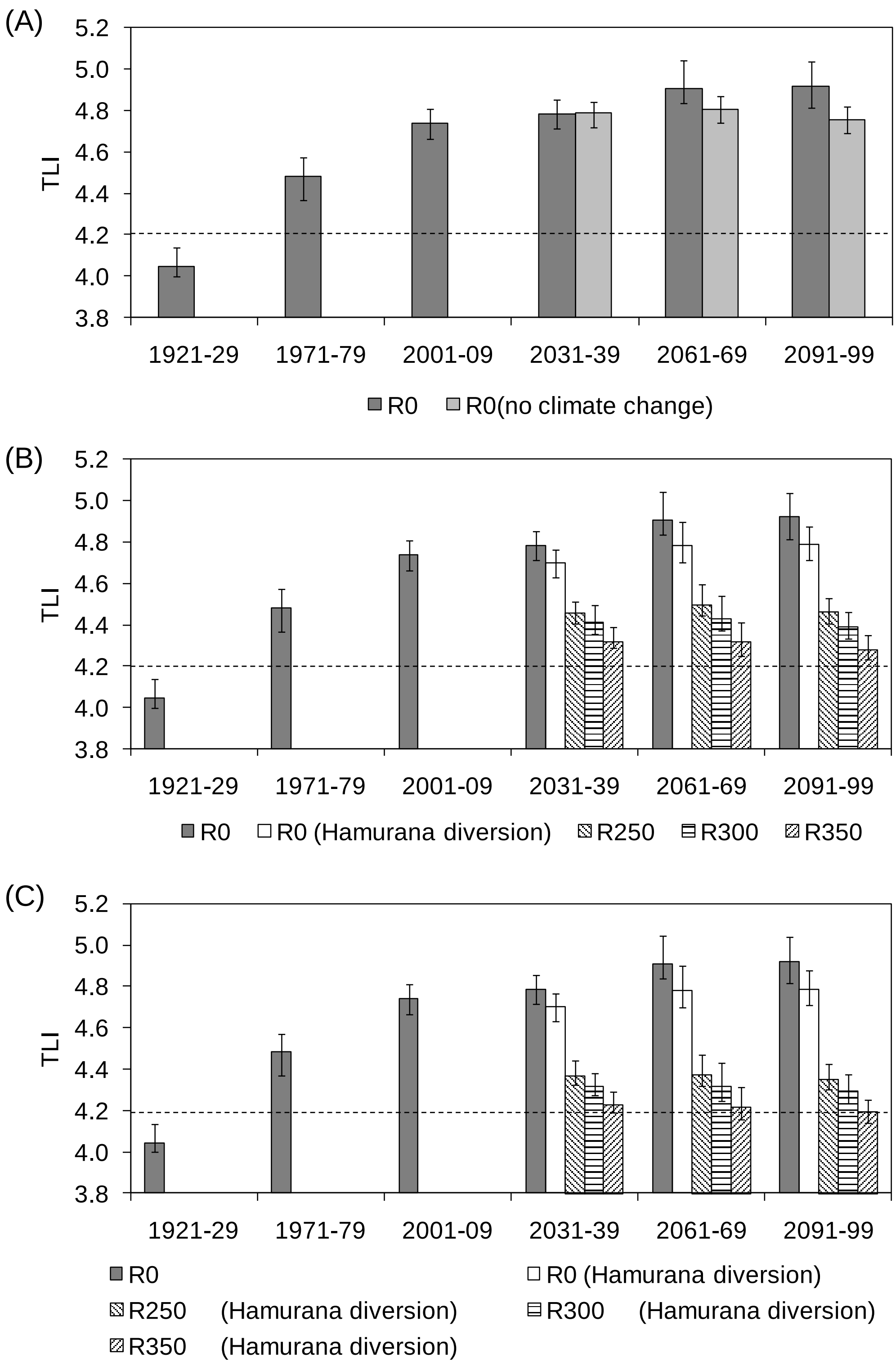


Figure 20. Mean TLI over six 8-year periods spanning 1920 - 2100 for (A) R0 (business as usual) with climate change, and R0 without climate change, (B) land use change scenarios including R0, R250, R300 and R350 (numbers in scenario names indicate the reduction in nitrogen load (t yr^{-1}) relative to R0) and (C) land use change scenarios including R0, R250, R300 and R350 with Hamurana springs diversion. The TLI target of 4.2 is shown as a dashed line. Bars represent inter-annual variation (range) of TLI for the respective 8-year periods.

3.4 Sediment management

The simulation of a sediment capping layer using a P binding agent resulted in a decrease of TLI just below the target TLI of 4.2 in the year of application (2031; Figure 21). The TLI for this scenario increased subsequently as a result of the decay function applied to the P uptake efficacy (equation 12), and values of TLI were identical to the R350 scenario 5 years after the sediment capping application.

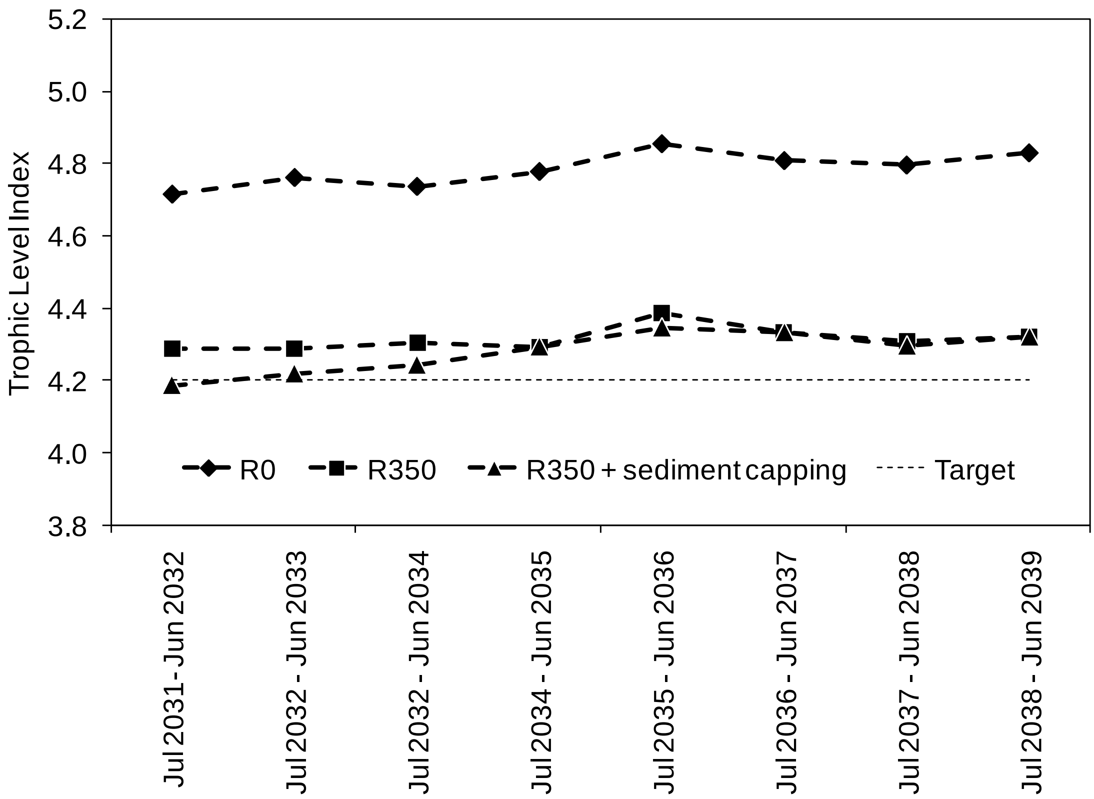


Figure 21. Simulated TLI values for R0, R350, R350 + sediment capping for the period July 2031 until Jun 2039. Target TLI of 4.2 is indicated as horizontal dashed line.

3.5 The effect of reduced nutrient loading on frequency of cyanobacteria blooms

Figure 22 shows the distribution of simulated cyanobacteria concentrations for each land use change scenario during the periods 2031-2039, 2061-2069 and 2091-2099. The results indicate that a reduction in nutrient loading would result in a reduced frequency of high cyanobacteria concentrations. While some high concentrations of around 50 mg m^{-3} still occurred even in the most drastic land use change scenario (R350), the frequency was greatly reduced. Most notably, the frequency of cyanobacteria concentrations around 30 mg m^{-3} was reduced substantially for all land use change scenarios, while the frequency of lower cyanobacteria concentrations (15-20 mg m^{-3}) was increased.

Table 12 shows a summary of the simulated cyanobacteria concentrations during each 8-year period for each scenario. The upper table summarises the number of days out of the total simulations days (2922) during which the cyanobacteria concentration was greater than 20 mg m^{-3}. The middle table shows the relative proportion of days during which the cyanobacteria concentration was greater than 20 mg m^{-3}. For example, during the period 2091-2099 in scenario R0, period 2091-2099, 40% of this entire period had cyanobacteria concentrations exceeding 20 mg m^{-3}. During the same period in scenario R350, only 16% of the entire period had cyanobacteria concentrations exceeding 20 mg m^{-3}. The lower table shows the relative change in number of days exceeding 20 mg m^{-3} compared to the base line (i.e. R0). For example, in scenario R350, period 2091-2099, the number of days exceeding the cyanobacteria reference concentration was reduced by 60%.

The climate change scenario increased the frequency of cyanobacteria blooms. Occurrences of concentrations around 20 to 30 mg m^{-3} increase with climate change, and more severe blooms (i.e. > 30 mg m^{-3}) increased only slightly. The increase in cyanobacteria blooms with climate change is likely to be stimulated by multiple stressors in the model. An increase in water temperature will favour the growth of cyanobacteria as can be seen by the parameters for phytoplankton temperature representation. Furthermore, internal nutrient loading is likely to increase with climate change as a result of increased water temperature (internal loading itself is temperature dependent in the model), and extended duration of stratification events. These additional nutrients in the water column and less frequent mixing events (see parameter for buoyancy) will promote cyanobacteria growth.

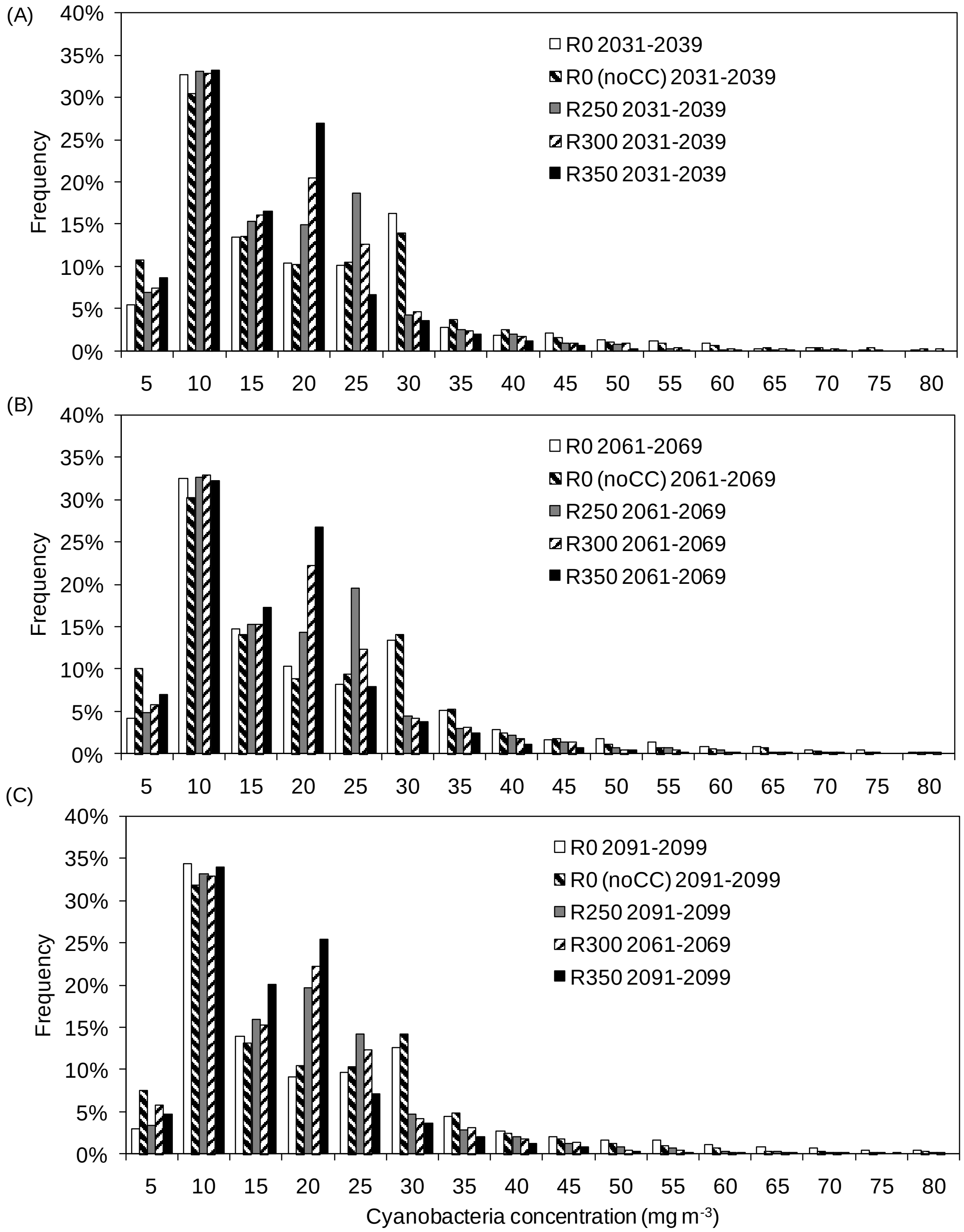


Figure 22. Probability distribution of cyanobacteria concentrations in Lake Rotorua for the base scenario (R0), R0 without climate change (R0 no CC) and three other external nutrient load scenarios for A) 2031-2039, B) 2061-2069 and C) 2091-2099. Extreme concentrations (>80 mg m-3) are infrequent, and thus omitted from the graphs.

Table 11. Summary of simulated cyanobacteria concentrations during each 8-year period for each scenario.

Number of days with cyanobacteria concentration > 20 mg m^{-3}					
Period	R0	R0 (noCC)	R250	R300	R350
1921-1929	148	148	148	148	148
1971-1979	1031	1031	1031	1031	1031
2001-2009	1060	1060	1060	1060	1060
2031-2039	1112	1029	871	682	429
2061-2069	1115	1076	963	700	490
2091-2099	1158	1091	812	614	461

Proportion of days with cyanobacteria concentration > 20 mg m^{-3}					
Period	R0	R0 (no CC)	R250	R300	R350
1921-1929	5%	5%	5%	5%	5%
1971-1979	35%	35%	35%	35%	35%
2001-2009	36%	36%	36%	36%	36%
2031-2039	38%	35%	30%	23%	15%
2061-2069	38%	37%	33%	24%	17%
2091-2099	40%	37%	28%	21%	16%

Relative change of days with cyanobacteria concentration > 20 mg m^{-3}					
Period	R0	R0 (no CC)	R250	R300	R350
1921-1929	0%	0%	0%	0%	0%
1971-1979	0%	0%	0%	0%	0%
2001-2009	0%	0%	0%	0%	0%
2031-2039	0%	-7%	-22%	-39%	-61%
2061-2069	0%	-3%	-14%	-37%	-56%
2091-2099	0%	-6%	-30%	-47%	-60%

3.6 The effect of reduced nutrient loading on bottom water oxygen concentrations

Figure 23 shows the distribution of simulated bottom water (20 m) dissolved oxygen concentrations for each land use change scenario during the periods 2031-2039, 2061-2069 and 2091-2099. The results showed that a reduction in nutrient loading would result in a decrease of low oxygen concentrations in bottom waters. The frequency of higher oxygen concentrations (> 8 mg L^{-}) increased substantially with reduction in nutrient loads. Table 13 shows a summary of simulated bottom water dissolved oxygen concentrations during each 8-year period for each scenario. The upper table summarises the number of days out of the total simulations days (2922) during which dissolved oxygen concentrations were lower than 2 mg L^{-1}. The middle table shows the relative proportion of days during which dissolved oxygen concentrations were lower than 2 mg L^{-1}. For example, during the period 2091-2099 in scenario R0, period 2091-2099, 13% of this entire period had dissolved oxygen concentrations below 2 mg L^{-1}. During the same period in scenario R350, dissolved oxygen concentrations exceeded 2 mg L^{-1} for the entire period. The lower table shows the relative change in number of days with dissolved oxygen concentrations < 2 mg L^{-1} compared to the baseline (R0). For example, in scenario R350 period 2091-2099, the number of days exceeding the dissolved oxygen reference concentration would be reduced by 100%.

The climate change scenario tested had a negative effect on bottom water dissolved oxygen concentrations. By 2099, the number of days with dissolved oxygen concentrations in bottom waters below 2 mg L^{-1} increased by 56%. The duration of hypolimnetic oxygen depletion during stratification periods was also increased. The effect of climate change on bottom water oxygen concentrations became more severe from 2061 onwards. The frequency of bottom water concentrations between 0 and 0.5 mg L^{-1} almost doubled with climate change. The increased frequency of anoxic events may have multiple causes. Any changes in external loading due to climate change as simulated by ROTAN will affect the sediment oxygen demand (see Table 8 for sediment parameters). Furthermore, an increase in water temperature due to climate change will increase sediment oxygen demand, since this process is temperature dependent. An increase of stratification events due to the increase in water temperature will also increase the duration of anoxic periods during these periods.

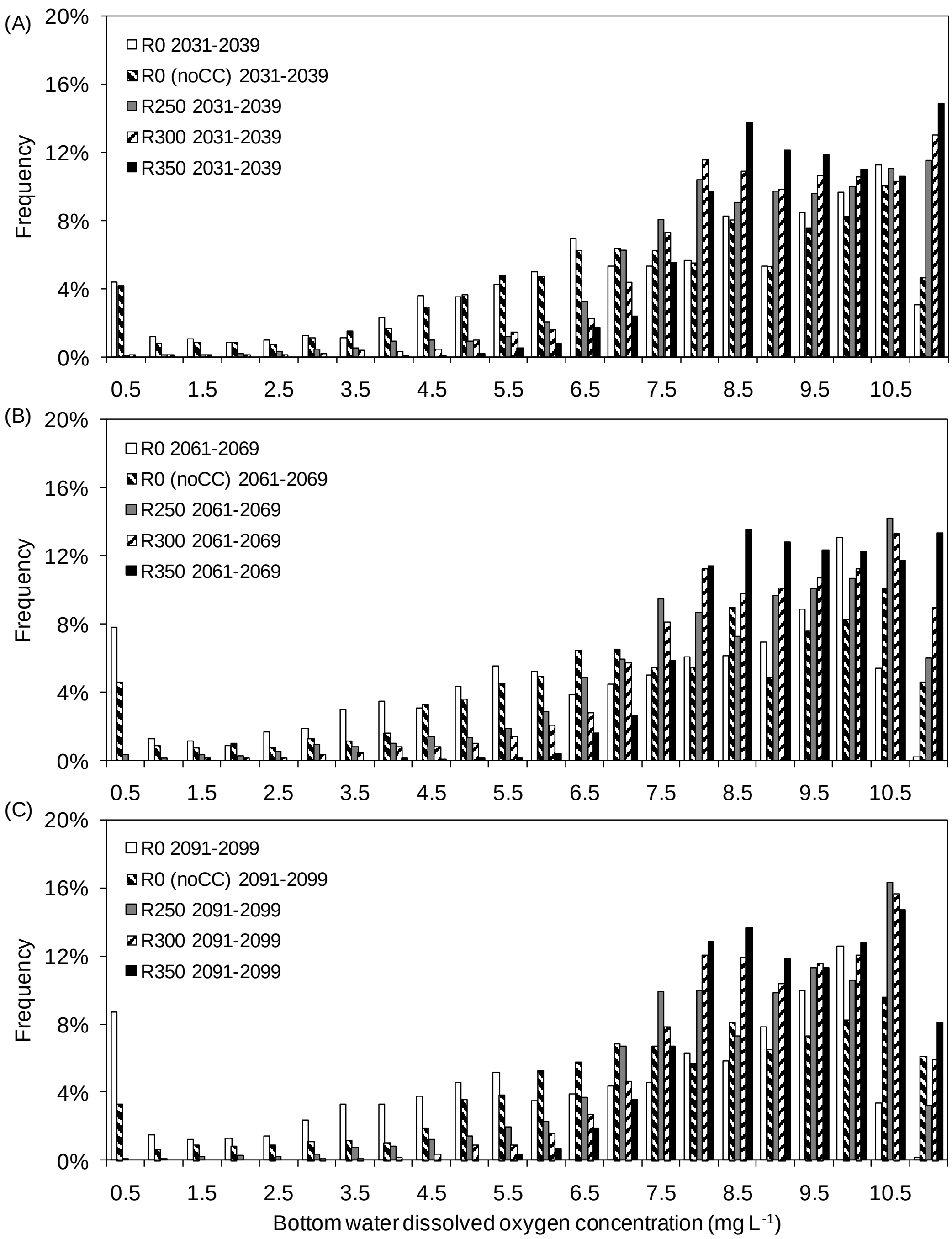


Figure 23. Probability distribution of bottom-water (20 m) dissolved oxygen concentrations in Lake Rotorua for the base scenario (R0), R0 without climate change (R0 no CC) and three other external nutrient load scenarios for A) 2031-2039, B) 2061-2069 and C) 2091-2099.

Table 12. Summary table of simulated bottom water dissolved oxygen concentrations during each 8-year period for each scenario.

Number of days with bottom water dissolved oxygen concentration < 2 mg L^{-1}					
Period	R0	R0 (noCC)	R250	R300	R350
1921-1929	0	0	0	0	0
1971-1979	2	2	2	2	2
2001-2009	123	123	123	123	123
2031-2039	219	190	15	7	0
2061-2069	324	204	30	3	0
2091-2099	373	163	18	0	0

Proportion of days with bottom water dissolved oxygen concentration < 2 mg L^{-1}					
Period	R0	R0 (noCC)	R250	R300	R350
1921-1929	0%	0%	0%	0%	0%
1971-1979	0%	0%	0%	0%	0%
2001-2009	4%	4%	4%	4%	4%
2031-2039	7%	7%	1%	0%	0%
2061-2069	11%	7%	1%	0%	0%
2091-2099	13%	6%	1%	0%	0%

Relative change of days with bottom water dissolved oxygen concentration < 2 mg L^{-1}					
Period	R0	R0 (noCC)	R250	R300	R350
1921-1929	0%	0%	0%	0%	0%
1971-1979	0%	0%	0%	0%	0%
2001-2009	0%	0%	0%	0%	0%
2031-2039	0%	-13%	-93%	-97%	-100%
2061-2069	0%	-37%	-91%	-99%	-100%
2091-2099	0%	-56%	-95%	-100%	-100%

4. Discussion

This report, which combines model simulations of climate, catchment discharge and nutrient loads, and lake water quality, has emphasised several important aspects of management that are required to improve water quality in Lake Rotorua. A satisfactory level of calibration was considered to have been achieved, which included the development of empirical models to dynamically represent changes in bottom-sediment nutrient fluxes. The calibrated model was used to provide insights into management strategies required to reduce the TLI to its target value of 4.2. The model fit was less satisfactory in 2003-4, mostly due to one exceptional even when *Anabaena planktonica* was strongly dominant. Similarly Özkundakci et al. (2011) found that chlorophyll *a* concentrations were under-predicted in Lake Okaro in 2004 when *A planktonica* was strongly dominant. This species was first detected in New Zealand in 2000, and has spread rapidly throughout the North Island of New Zealand (Wood et al., 2005; Environment Bay of Plenty, unpubl. data). It has been reported to dominate the phytoplankton community for short periods within 2-3 years of its first recognised presence in a water body (Ryan et al., 2003). Simulation of N-fixation by cyanophytes was outside the scope of the present study and thus was not included in the conceptual model. Adapting the present conceptual model to include N-fixation would ideally rely on direct measurements of N-fixation. Given the emphasis on nitrogen load reductions for many lake ecosystems in New Zealand, a field study to measure nitrogen fixation rates in a lake which had low N:P ratios and a high proportion of potentially N-fixing cyanobacteria, would be extremely valuable in quantifying the inputs of N to lakes from this source.

4.1 Nutrient load reduction

Large reductions in external nutrient loads to the lake were required in order to decrease the Trophic Level Index (TLI) towards the target value (4.2), which is aimed at improving water quality to levels that were generally considered to be 'acceptable' around the 1960s. Early intervention to attenuate catchment nutrient loads will bring about more rapid responses because large aquifers respond only slowly to land use change due to the relatively small annual increments of water to these aquifers relative to the volume of water stored in them. Quite substantial load reductions were simulated to occur in ROTAN within 1-2 decades because of a relatively rapid response of smaller aquifers compared with the less variable background concentrations contributed by the large aquifers. The lake response, on top of the catchment response, was more prolonged, and even in 2091-99 there was a gradual improvement in water quality (a decreasing TLI) compared with 30 years prior (2061-69). Thus catchment nutrient load reductions must be sustained and significant to reduce TLI in Lake Rotorua towards the target value. On the other hand, as nutrient loads were reduced in the R350 case (350 t N yr^{-1} load reduction) the TLI became progressively less responsive and the rate of decrease was reduced with higher N load reductions. Our results say nothing about the specific ways in which catchment nutrient loads will be reduced, i.e., whether through land use change, best catchment management practices or specific catchment engineering options (e.g., diverting or treating geothermal inflows) (see discussion in Abell and Hamilton 2012).

4.2 Climate change

If there is a warming in climate according to IPCC projections (i.e. c. 2.5 ^{o}C annual mean increase in air temperature in the Bay of Plenty) then it is likely that the TLI target will be more difficult to attain. Our report does not infer whether or not climate change will occur but, because the best available scientific information indicates that an increase is probable (IPCC 2007), we considered it prudent to build such an increase into our predictions of the future TLI for Lake Rotorua. The selected scenario of a 2.5 ^{o}C increase in mean annual air temperature conforms to the A1B mid-range case based on the different models and scenarios recognised by the IPCC. The mean annual global extremes of between 1.4 and 5.8 ^{o}C projected by models recognised by the IPCC are likely to downscale to be similar to the possible temperature increase in the Bay of Plenty. It may be useful to examine the models' responses to the IPCC extreme values in the future as well as to plot observed temperatures against model predictions at various intervals (e.g., decadal) to refine the range of projected temperature increases for lake modelling and provide updated information in future.

The mid-level increase in temperature due to climate change is likely to make it more difficult to attain the TLI target. The primary driver for this is a change in the duration of stratification and mixing in the lake. Lake Rotorua is a polymictic lake and is particularly sensitive to changes in air temperature compared with the deeper Rotorua lakes which are monomictic (e.g. Rotoiti, Tarawera, Okataina). The increased temperature will prolong the duration of intermittent stratification events and result in more prolonged oxygen depletion and anoxia in the bottom waters of the lake, leading to higher internal nutrient loads. Cyanobacteria species will also benefit from increases in stratification, particularly those species that exhibit strong buoyancy control or are naturally buoyant due to presence of gas vesicles. In general the climate change scenarios conform to recent hypotheses about how lakes may respond to climate change (Moss et al. 2011; Carey et al. 2012) and indicate that the cyanobacteria, more than any other phytoplankton phyla, may become more dominant, resulting in increased frequency, duration and magnitude (biomass) of blooms. It should also be emphasised that there is much that is not well understood about how other components of the lake ecosystem may respond, which could potentially have a direct and indirect impact on TLI components. Secondary producer effects are not accounted for in the model (similar to Trolle et al. (2011)) but have the potential to impact upon phytoplankton biomass and species' successions in Lake Rotorua and these interactions are influenced by climate change also (Greig et al. 2011). In general, however, the model supports recent statements about active management and intervention actions to manage nutrient inputs to lakes so that the effect of increasing water temperature, which would otherwise stimulate phytoplankton biomass, can be mitigated by a general decrease in lake trophic status (Carey et al. 2012).

Changes in rainfall with climate change have the potential to have a more profound influence on the TLI. The frequency of extreme rainfall events (e.g., a one in one-hundred year rainfall event) is expected to roughly double under the A1B scenario of 2091 to 2099. At present the catchment model ROTAN is relatively poorly set up to be able to deal with these types of events because it represents these only as dilution events in dealing volumetrically with the quantity of water and not with any changes in sediment and nutrient generation (e.g., from changes in erosion of sediments and nutrients, or from changes in leaching).

4.3 Sediment management

The eutrophication of Lake Rotorua has occurred due to the coalescence of several factors including:

- increasing catchment nutrient loads (and input of treated sewage until 1991);
- higher levels of nutrients and algal biomass;
- increasing deposition of organic material associated with higher rates of sedimentation of algal cells;
- deoxygenation (anoxia) of bottom waters when there are sustained stratification events (of more than a few days), and;
- high diffusive fluxes of ammonium and phosphorus from the bottom sediments to the water column, particularly in the presence of anoxia.

There is an active feedback between these factors resulting in a cycle that reinforces algal growth and results in poor water clarity. Breaking this positive feedback cycle requires sustained controls on catchment nutrient loads but model simulations indicate that significant improvements can occur over moderate time scales, as evidenced from the relationship between ROTAN-predicted catchment N loads and N concentrations in the bottom sediments. Sustained catchment nutrient load management will have far-reaching consequences in ultimately reducing the frequency and duration of dexoygenation events and occurrences of cyanobacterial blooms (i.e., reversing the eutrophication feedback cycle).

Actively reducing sediment nutrient loads by sediment capping (or some similar one-off technique) is a means of more rapidly attenuating the internal load of nutrients but the implications from the model simulations are that while there is persistence of reasonably high sediment and nutrient loads from the catchment, sediment capping effects on trophic status may be relatively short-lived. High current velocities in the internal part of the lake basin (M. Gibbs, NIWA; observations from an Acoustic Doppler Current Profiler) and wave action in the shallower littoral area (Stephens et al. 2004) are likely lead to large amounts of sediment re-working (Burger 2006) that could also jeopardise the efficacy of a sediment capping operation. The present strategy of alum dosing of specific inflows with high P levels (Puarenga and Utuhina) may in fact provide more effective means of removal of P from the water column and is something that will be simulated and explicitly included in further Lake Rotorua model simulation studies.

4.4 Hamurana diversion

Diverting the Hamurana inflow did not yield results as expected based on nutrient load reductions alone. This may be attributed to the loss of this cold, saturated inflow and the oxygenation of bottom sediments which it induces as it inserts as an underflow into the lake. The loss of this inflow induces greater anoxia-driven nutrient releases from the bottom sediments and the benefits of the reduction in external nutrient load are more or less compensated for by the increase in the internal nutrient load.

The modelling scenarios presented in this report are based on idealised assumptions and only limited consideration has been made of how the required nutrient reductions to meet the TLI target will be met. Substantial additional work will be required to understand the social, economic and political elements of possible changes in the Lake Rotorua catchment, particularly where substantial land use change is considered necessary to meet TLI targets for Lake Rotorua.

5. Conclusions

This modelling study has been based on a comprehensive set of input data to simulate the water quality dynamics of Lake Rotorua, including climate, hydrology and morphology, as well as in-lake data for comparative purposes. A model calibration undertaken with input data for 2005-2009 was used to fix model parameters that were subsequently used for a 2001-2004 validation period and for forecasting water quality for one decade at different intervals in the future (2031-39, 2061-69 and 2091-99). An empirical model was used to provide a dynamic representation of expected future changes in sediment composition, and a future climate was represented by modifying the 'present climate data' (2001-9) with a mid-range change (A1B scenario) derived from climate model outputs summarised by the IPCC. The simulations indicate that a catchment-based (external load) reduction of 350 tonnes of nitrogen per year, with a similarly large proportional reduction of phosphorus load, will be required to reach the target Trophic Level Index of 4.2, representing a boundary close to where the lake trophic state classification would transition from eutrophic to mesotrophic. In achieving this reduction the model simulations indicate that there would be substantially reduced incidence of deoxygenation events and severe cyanoabcterial blooms. A warming climate (c. 2.5 ^{o}C increase in air temperature by 2100) is predicted by make the TLI target more difficult to attain for a given nutrient load reduction. Artificially modifying inputs of nutrients arising from the bottom sediments (e.g., through capping of sediments with mineral media) could hasten transition to the TLI target but would likely require repeated capping applications at intervals of a few years, in the absence of sustained reduction in nutrient loads. A restoration option of diversion of the Hamurana Stream appears unlikely to achieve the desired level of TLI reduction because whilst its removal would reduce external nutrient loads to the lake it could adversely affect oxygen levels in bottom waters, leading to increased internal nutrient loads. Further work is now required to examine the sustainability of alum dosing, currently being used where the Utuhina and Puarenga Stream inflows enter the lake, as a tool to reduce both external and internal phosphorus loads.

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Appendices

1.0 Additional land use P mitigation investigation. Simon Park, 2012.

Reductions in P load to Lake Rotorua

Headway Ltd

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Cc	John McIntosh
From:	Simon Park, Headway Ltd
Date:	18 June 2012

Purpose of this paper

To provide a basis for reductions in P load from the Lake Rotorua catchment for use in Lake DC modelling based on:

- Combinations of on-farm P mitigation strategies designed to achieve a high but credible overall reduction in P loss from pastoral land across dairy, drystock and "lifestyle" land uses
- Separate and additional P load reductions from proposed sewage reticulation, septic tank upgrades and urban stormwater improvements
- Catchment P load estimates based on land use areas and P loss coefficients for status quo and mitigated scenarios consistent with the R-0, R-250, R-300 and R-350 ROTAN scenarios.

On farm P mitigation levels

The Rich McDowell 2010 report1 provides a range of P mitigation methods, efficacy and cost-effectiveness. These P mitigation "strategies" were compiled specifically for the Lake Rotorua catchment. The effectiveness levels are given for individual strategies or methods and are attached as an Appendix.

To maximise P mitigation on any individual farm, the methods would need to be tailored to the relevant farm system, topography, soil type and nutrient status etc. Some mitigations, notably optimum Olsen P levels using RPR, will apply to the whole farm. Most other mitigations will realistically apply to a proportion of the farm.

A combination of known or "standard" on-farm phosphorus management practices can reduce P losses per hectare by up to 50%, although this can be achieved more easily on dairy farms than on drystock farms. The main methods are:

- reducing soil Olsen P levels to the lower end of the productive optimum range

- using low solubility fertiliser i.e. RPR (reactive phosphate rock) instead of super-phosphate
- low-rate effluent irrigation, possibly in combination with greater effluent storage

These practices are in McDowell's Table 2 which is attached as an Appendix to this report. Similar P mitigation methods are found within the Overseer phosphorus mitigation guidance (2012) which is in turn based on the fertiliser industry's "Nutrient Management Code of Practice" (Fert Research, 2007).

The cost-effectiveness of P mitigation measures varies greatly, from profit-enhancing (reducing soil Olsen P levels to the lower end of optimum range) to relatively expensive wetlands and detention dam structures.

While P losses from "critical source areas" can be responsible for the majority of total farm P losses (McDowell, 2010), there is currently no tool available to model those losses, nor how such losses may be reduced by focusing P mitigation efforts on CSA. Due to the current high standard of stream fencing and planting in the Lake Rotorua catchment, it is likely that there is less scope to achieve very high CSA-based reductions locally. Despite this limitation, there is still likely to be combinations of standard P management (Olsen P etc) and CSA initiatives that make an overall farm P loss reduction of 50% achievable and credible.

Two approaches were taken to check what combination of P mitigation would achieve (or approach) a 50% reduction in P loss, based on McDowell's Table 2 (McDowell, 2010) and an Overseer analysis.

Combinations of P mitigation methods from McDowell's Table 2

Some P mitigation methods will apply to 100% of a farm and some will realistically apply to much smaller areas (based here on the author's judgement). The effectiveness rates were simply based on the mid-point of the range given by McDowell. It is conservatively assumed that there are no synergies between mitigation methods and, as percentages, they cannot be simply summed. Rather, each successive mitigation applies to the balance of the "yet to be mitigated P loss". The selected combination for a hypothetical dairy farm is shown in Table 1 below.

P Mitigation Strategy	Effective-ness %	Mid-point effectiveness	Relevant farm area	Effective-ness net	Cumulative reduction
Optimum soil test P	5-20	12.5%	100%	12.5%	12.5%
Low solubility P fertilizer	0-20	10%	100%	10.0%	21.3%
Stream fencing	10-30	20%	30%	6.0%	26.0%
Greater effluent pond storage	10-30	20%	15%	3.0%	28.2%
Low rate effluent application	10-30	20%	15%	3.0%	30.3%
Restricted grazing of cropland	30-50	40%	5%	2.0%	31.7%
Alum to pasture	5-30	0%	0%	0.0%	31.7%
Alum to grazed cropland	30	0%	0%	0.0%	31.7%
Grass buffer strips	0-20	10%	10%	1.0%	32.4%
Sorbents in and near streams	20	10%	0%	0.0%	32.4%
Retention dams / water recycling (2)	10-80	45%	25%	11.3%	40.0%
Constructed wetlands (3)	-426-77	0%	0%	0.0%	40.0%
Natural seepage wetlands (3)	<10%	5%	5%	0.3%	40.3%

Table 1: Combined P mitigation strategies, adapted from McDowell (2010).

The cumulative reduction in P loss is just over 40%, short of the 50% "target" reduction. Therefore a set of more ambitious assumptions was applied as follows:

- Optimum Olsen P efficacy increased from 12.5% to 20%
- Effluent pond storage and low rate irrigation efficacies increased from 20% to 25%, but still applied to 15% of farm area (typical effluent block proportion)
- Fodder crop grazing restriction efficacy increased from 40% to 50%
- Area treated by grass buffers raised from 10% to 30% of farm, and area treated by retention dams raised from 25% to 35%

The net effect of these changes was to increase the cumulative on-farm P mitigation to just over 50%. While some elements could be classified as Critical Source Areas (e.g. grazed crops), a more comprehensive approach to CSA mitigation could be expected to contribute to a 50% overall reduction with less ambitious assumptions than used here. This CSA rationale would arguably apply to drystock and dairy farms alike.

Overseer estimate of on-farm P mitigation

A hypothetical and simplified dairy farm was created in Overseer using the current version[1] (5.4.10). The main farm parameters reflect the author's knowledge of Rotorua dairy farm systems, and include: 200ha effective area, rolling topography, Oturoa sandy loam soil, Olsen P = 60, 2.7 cows/ha, maintenance super-phosphate, 180 kgN/ha/yr as urea, 30ha (15%) effluent block with nil fertiliser and 1800 mm rainfall. The Overseer prediction of N and P loss was 10,007 kgN/yr and 595 kgP/yr, corresponding to rates of 50kgN/ha/yr and 3.0 kgP/ha/yr respectively. Note that while ROTAN has indicated there is little or no attenuation of N once it leaves the root zone, there will be significant P load attenuation. However, the object is to illustrate what proportional reduction can be achieved by applying credible mitigation strategies within Overseer. The three changes made were:

- **Reduce Olsen P from 60 to 40** (mgP/kg soil). Note that the optimum range for pumice soils at average production levels is 35-45. While some Rotorua dairy farms are high producing, most are below national and regional averages due to climate limitations. Further, an Olsen P of 60 is fairly typical (see Redding et al, 2006).
- Use a low rate effluent irrigation system
- Use RPR instead of super phosphate

The Overseer file was run for each mitigation as the sole change from the status quo, then again with all three together. The results are summarised in Table 2.

Mitigation	Farm P loss, kg/y	kgP/ha/y	% change
Olsen P = 40	461	2.305	-22.5%
Low rate effluent	554	2.77	-6.9%
Use RPR	532	2.66	-10.6%
All 3 above	366	1.83	-38.5%

Table 2: Combined P mitigation strategies using a hypothetical Rotorua dairy farm in Overseer.

Overseer allows the user to apply grass filter strips in addition to the more typical mitigations used in Table 2. An ambitious set of filter strip parameters can reduce P loss by a further 10%. However, it is more realistic to expect a range of CSA-focused mitigations would be necessary to lift mitigation from about 38% to about 50% overall.

[1] Overseer Version 6 was not available at the time of writing (early June 2012). It is expected that Version 6 will have an improved P sub-model.

Recent P loss estimates for the Lake Rotorua catchment

Any P mitigation scenario will necessarily be relative to the status quo P load. For modelling purposes, the current or status quo load is equivalent to the R-0 scenario whereby land use, nutrient loss rates and all other inputs are held constant at 2010 levels. The range of current P load estimates is:

- 39.8 tP/yr from Table 7 from the 2007 Proposed Rotorua-Rotoiti Action Plan, based on 2005 land use areas, P loss coefficients and other nutrient sources
- 39.1 tP/yr from the Action Plan's Table 6 which was based on actual stream and other data/assumptions.
- 51.2 tP/yr via CLUES modelling carried out by UoW (Wei Ye & Deniz Ozkundakci in 2010/2011), based on the land use layers used in the associated ROTAN modelling by NIWA (actually the R-0 scenario which is based on 2010 land use)
- 32.0 tP/yr used in the draft Lake DC report, from Table 2 (sum of 24.94 PO_4 and 7.02 organic P), based on stream data (plus RF) from the 2001-2009 calibration period used for Lake DC. Subsequent discussion has noted that the calibration data set may have under sampled storm events and associated nutrient loads, especially particulate N & P.
- 28-33 tP/yr in Rutherford & Timpany's 2008 NIWA report "Storm nutrient loads in Rotorua streams", using 1992-2005 stream data. As the title indicates, this sum does not include P from rainfall, ephemeral sub-catchments and near-lake springs. A separate BoPRC GIS assessment that ephemeral sub-catchments comprise about 20% of the total catchment.
- 39-40 tP/yr from Hoare's 1980 paper.

Considering the range in estimates, it appears prudent to assess P mitigation potential against two status quo P loads, rather than settle on a "correct" P load prematurely.

Land use areas and P loss coefficients

ROTAN used 12 land use categories although some of these were closely related (Forest and Forest Puarenga) and some were small areas with high loadings (e.g. Tikitere) to reflect a consistent GIS input structure. These are given in Table 2 of the final ROTAN report (Rutherford et al, 2011). For this desk top assessment of P loads, it is sufficient to use a simpler set of five land use categories, adjusted for the smaller surface catchment area (about 6000ha less than the groundwater catchment) and the GIS data provided by Michele Hosking in 2010. The relative proportionality of land uses in the various ROTAN scenarios has been maintained but with a downward adjustment in forestry because most of the probable "additional" groundwater catchment near Mamaku is dominated by forest. The resultant land uses and areas are shown in Table 3 below:

	ROTAN scenario, area (ha)			
Land use	R-0	R-250	R-300	R-350
Dairy	4499	2250	0	0
Drystock	14861	8890	12491	8910
Lifestyle	1053	2577	2577	2577
Forest	16713	31487	30135	33717
Urban	3353	3353	3353	3353
Lake	8077	8077	8077	8077
Totals	48556	48556	48556	48556

Table 3: Simplified Lake Rotorua catchment land use areas.

The P loss coefficients applied to different land uses forms the basis of catchment load predictions in Table 7 of the Proposed Rotorua-Rotoiti Action Plan (EBOP, 2007), including: pasture at 0.9 kgP/ha/yr and forestry at 0.10-0.12 kgP/ha/yr. More recently, McIntosh has applied higher coefficients in the draft Lake Rerewhakaaitu and Tarawera Action Plans, using pasture at 1.1-1.2 kgP/ha/yr and forestry at 0.4 kg/ha/yr. Both sets of coefficients were used in the simple spreadsheets reported later in this report.

P loads and potential reductions

The land use areas (adjusted from ROTAN scenarios) have been combined with status quo and mitigated P loss coefficients to give a range of P mitigation reductions for the catchment overall. Each overall load is a combination of on-farm mitigation and land use change. All the ROTAN-based mitigation scenarios (R-0, R-250, R-300 and R-350) are subject to P mitigation adjustments and are compared to "R-0 with no P mitigation" i.e. the status quo. To avoid confusion, the R-0 mitigated scenario is called R-P. In addition to the 50% reduction in pastoral P loss and all relevant land use changes (mainly to forestry, as per ROTAN), the following changes are applied:

- Urban P loss is reduced by 0.5 tP/yr, applied as a 20% reduction from 0.70 kgP/ha/yr to 0.56 kgP/ha/yr, which corresponds to Action Plan assumptions
- The combined septic tank reticulation and WWTP upgrades reduce P load by 1.0 tP/yr, in line with Action Plan assumptions

No changes are applied to springs and rainfall and the P-locking plants are ignored i.e. it is assumed that the range of land-based P mitigations are potentially a substitute for stream alum dosing, at least in this desktop study.

	Export coef't. kgP/ha/y			**P load, tP/yr**				
Land use	R-0	mitigated	% change	**R-0**	**R-P**	**R-250**	**R-300**	**R-350**
Dairy	0.9	0.45	50%	4.0	2.0	1.0	0.0	0.0
Drystock	0.9	0.45	50%	13.4	6.7	4.0	5.6	4.0
Lifestyle	0.8	0.4	50%	0.8	0.4	1.0	1.0	1.0
Forest	0.11	0.11	0%	1.8	1.8	3.5	3.3	3.7
Urban	0.70	0.56	20%	2.3	1.9	1.9	1.9	1.9
rainfall	0.15	0.15	0%	1.2	1.2	1.2	1.2	1.2
Septic tanks			50%	1.2	0.6	0.6	0.6	0.6
WWTP			25%	1.6	1.2	1.2	1.2	1.2
springs			0%	13.0	13.0	13.0	13.0	13.0
			Total P load	**39.5**	**28.9**	**27.4**	**27.9**	**26.6**
			Change Vs R-0	**-**	**10.6**	**12.1**	**11.6**	**12.8**

Table 4a: Overall P load estimates featuring 50% pasture P mitigation.

The reductions in P load relative to R-0 range from 10.6 to 12.8 tP/yr. The R-P reduction of 10.6 tP/y does not include any land use change component. These large load reductions largely reflect the ambitious 50% reduction in pastoral P losses. As discussed above, this may be difficult achieve on drystock farms and probably lifestyle blocks as well, where a more achievable level may be a 30% reduction. The corresponding results are shown in abbreviated form as Table 4b below:

	P load, tP/yr				
	R-0	R-P	R-250	R-300	R-350
Total P load	39.6	31.8	29.6	30.7	28.8
Change Vs R-0	-	7.8	10.0	8.9	10.7

Table 4b: Overall P load estimates featuring 50% dairy and 30% drystock/lifestyle P mitigation levels.

The impact of assuming different P loss coefficients (as per Rerewhakaaitu and Tarawera draft action plans) is illustrated in Table 5a below:

	Export coef't. kgP/ha/y			**P load, tP/yr**				
Land use	R-0	mitigated	% change	**R-0**	**R-P**	**R-250**	**R-300**	**R-350**
Dairy	1.1	0.45	50%	4.9	2.5	1.2	0.0	0.0
Drystock	1.2	0.45	50%	17.8	8.9	5.3	7.5	5.3
Lifestyle	0.8	0.4	50%	0.8	0.4	1.0	1.0	1.0
Forest	0.4	0.4	0%	6.7	6.7	12.6	12.1	13.5
Urban	0.80	0.64	20%	2.7	2.7	2.1	2.1	2.1
rainfall	0.15	0.15	0%	1.2	1.2	1.2	1.2	1.2
Septic tanks			50%	1.2	0.6	0.6	0.6	0.6
WWTP			25%	1.6	1.2	1.2	1.2	1.2
springs			0%	13.0	13.0	13.0	13.0	13.0
			Total P load	**50.0**	**37.2**	**38.4**	**38.7**	**38.0**
			Change Vs R-0	**-**	**12.8**	**11.6**	**11.3**	**12.0**

Table 5a: Overall P load estimates with different status quo export coefficients and 50% pastoral P mitigation levels.

The status quo export coefficients used in Table 5a are combined with the lower 30% mitigation levels applied to drystock and lifestyle land uses, and shown in abbreviated form as Table 5b.

	P load, tP/yr				
	R-0	R-P	R-250	R-300	R-350
Total P load	50.0	40.9	40.9	42.1	40.6
Change Vs R-0	-	9.1	9.1	7.9	9.4

Table 4b: Overall P load estimates with different status quo export coefficients, 50% dairy and 30% drystock/lifestyle P mitigation levels.

Discussion

This desktop assessment encompasses multiple assumptions, including admittedly ambitious levels of on-farm P mitigation. However, a review of recent literature (notably McDowell 2010 which referenced numerous P studies) indicates that mitigation levels up to 50% are credible. This was largely confirmed by the combined spreadsheet analysis (based on Table 2 in McDowell, 2010) and the hypothetical Overseer assessment.

For the R-P scenario with no land use change occurring, the net reduction in P load relative to R-0 ranged from 7.8 to 10.6 tP/y. Across all other scenarios (R-250, R-300 & R-350), reductions relative to R-0 ranged from 8.9 to 12.8 tP/yr.

The level of P load reduction will depend partly on the starting point. This desktop assessment has mainly used the values from Table 7 in the 2007 Proposed Rotorua-Rotoiti Action Plan, where total load was close to 40 tP/yr, as reflected in Tables 4a and 4b. By contrast, assuming higher export coefficients in Tables 5a and 5b (as per Rerewhakaaitu and Tarawera draft action plans) gave a status quo load of 50 tP/yr. Perhaps by coincidence, this is close to the initial CLUES prediction of 51.2 tP/y and may reflect storm load P that is under-represented in the Table 7 estimate. Despite the different status quo load (~50 tP/y), most mitigation scenarios were still around 10 tP/yr, with a range from 7.9 to 12.8 tP/yr. It is also possible that the relatively high forestry P export coefficient of 0.4 kgP/ha/yr could be subject to mitigation by forestry harvest best practice, at least on plantation forests. This would increase potential mitigation, or allow a lower level of pastoral P mitigation.

Overall, there appears to be a valid basis for simplifying the wide range of mitigation scenarios and use a somewhat arbitrary but still credible reduction of 10 tP/yr. This could be applied to a range of status quo loads in the reasonable knowledge that there will be a number of credible means of achieving that level of reduction.

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Appendix 1: Table 2 from McDowell (2010)

Table 2. Summary of efficacy and cost of P mitigation strategies

Strategy		Effectiveness (%)	Cost (NZD $/kg P conserved)
Optimum soil test P	management	5-20[1]	highly cost-effective[1]
Low solubility P fertilizer		0-20	0-30
Stream fencing		10-30	5-65
Greater effluent pond storage		10-30	30
Low rate effluent application to land		10-30	45
Tile drain amendments	amendment	50	25-100
Restricted grazing of cropland		30-50	150-250
Alum to pasture		5-30	150->500
Alum to grazed cropland		30	160-260
Grass buffer strips	edge of field	0-20	>250
Sorbents in and near streams		20	350
Retention dams / water recycling[2]		10-80	>500
Constructed wetlands[3]		-426-77	>500
Natural seepage wetlands[3]		<10%	>500

[1] depends on existing soil test P concentration, but no cost if already in excess of optimum.

[2] upper bound only applicable to retention dams combined with water recycling

[3] potential for wetlands to act as a source of P renders upper estimates for cost infinite.