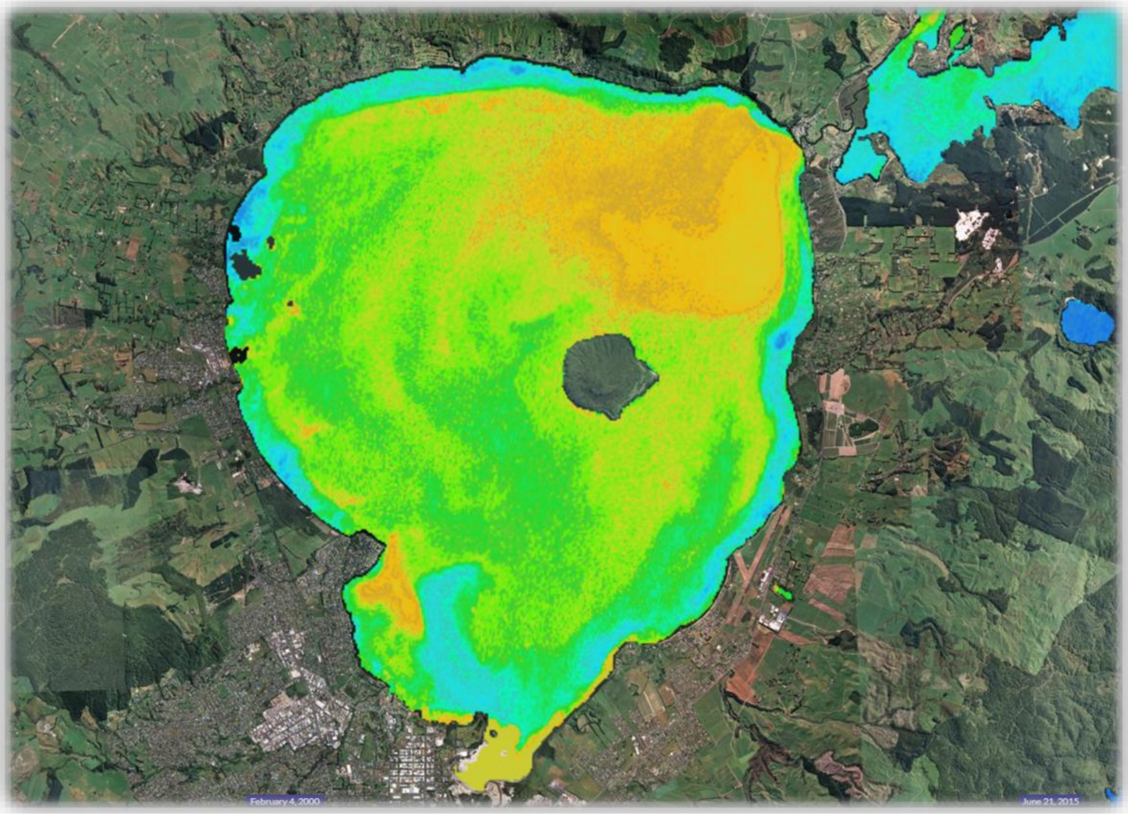
# Assessing the effects of nutrient load reductions to Lake Rotorua:

## Model simulations for 2001-2015



## February 2019

### ERI report: 124

Prepared for Bay of Plenty Regional Council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## Availability



## Data providers

Bay of Plenty Regional Council

Ministry of Works

NIWA

NZ MetService

## Cover image

False colour image of remotely sensed surface chlorophyll *a* variation in Lake Rotorua (source: Takiwa NZ lakes).

## Acknowledgements

We acknowledge the assistance of Bay of Plenty Regional Council for provision of data, and funding for this study. We especially acknowledge Andy Bruere, Paul Scholes and Glenn Ellery for their assistance with numerous requests made by the authors to support the information requirements of the authors. We thank Deniz Özkundakci and David Burger for their work with earlier versions of the model that set a basis for the continued improvements in calibration and validation of the model as part of the present study. NIWA and the New Zealand Meteorological Service provided climate data for Rotorua Airport that was used as input to the modelling. The DYRESM-CAEDYM model used in this study was developed at the Centre for Water Research, The University of Western Australia. Thanks to Troy Baisden for reviewing a draft version of this report. Finally, we thank those who undertook formal reviews of this report; Prof. Warwick Vincent (Plan Change 10 Science Review), Grant Tempero (internal reviewer) and John Tyrrell (Environmental Research Institute).

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## Executive Summary

Lake Rotorua has a history of catchment urbanisation and pastoral development, processes which have been linked to water quality decline observed in inflows and in the lake between the 1960s and 2000s. We have identified multiple studies through the 1980s that demonstrate accelerating eutrophication of Lake Rotorua and have examined the relationship between land use and trophic state. In the late 1980s, 'sustainable catchment nutrient loads' were established in peer-reviewed scientific literature, and a water quality target was set using a Trophic Level Index (TLI). For Lake Rotorua this value was set to 4.2 as part of the Bay of Plenty Regional Council (BoPRC) Water and Land Plan. This target is intended to reflect a desired water quality similar to that in the early 1960s – a state to which the community aspired to restore the lake water quality. Since the 1990s multiple strategies have been developed to reduce external nutrient loads in order to improve the water quality of Lake Rotorua.

Treated wastewater (sewage) was discharged directly to Lake Rotorua from 1973 until 1991, when a new treatment plant was established which discharged to 300 ha of plantation forest (Whakarewarewa Forest LTS – Land Treatment System) in the Waipa catchment of Lake Rotorua. Nutrient loads to the lake were greatly reduced by the LTS and lake water quality measurements initially indicated increased water clarity. In the early 2000s, following severe and widely publicised blooms of cyanobacteria, nutrient budgets were revisited. Because Lake Rotorua consistently failed to meet its water quality target, BoPRC, along with Te Arawa Lakes Trust and the Rotorua District Council, developed the Lake Rotorua 'Action Plan' (BoPRC, 2009). The plan specified that land use and management practices should be suitably aligned with the target TLI, to reduce catchment nutrient loads. The required nitrogen load reduction from levels in 2009 would need to be 112 t $yr^{-1}$, and a reduction of 224 t $yr^{-1}$ would be required from the projected load for 2055. The latter value is higher due to increasing concentrations resulting from lag times as groundwater transits through aquifers to the lake. Internal (lake-sediment) nutrient loads, which have been found to be comparable to external nutrient loads, also mean that there is a lag in recovery of Lake Rotorua from eutrophication. To address these lags and inability to meet the TLI target, BoPRC commenced alum dosing in the Utuhina Stream in 2006, and in the Puarenga Stream in 2010. Several studies indicate that alum dosing has had a positive impact on water quality (e.g., dissolved oxygen and cyanobacterial concentrations), exemplified by the TLI target being met for several years following commencement of dosing the two inflows concurrently.

The present application of the DYRESM-CAEDYM lake model builds upon a series of lake modelling studies aimed at understanding processes in Lake Rotorua and supporting management decisions in the lake and catchment. Specifically, we aimed to improve the boundary conditions for the model by providing detailed estimates of inputs of water, nitrogen and phosphorus to the lake at a daily scale. We sought to increase the number of different nutrient loading scenarios to encompass a 'matrix' involving different combinations

of nitrogen and phosphorus loads. This matrix allows insights into water quality outcomes for the lake arising from a variety of management strategies that may alter the relative loads of nitrogen and phosphorus to the lake. We use the modelled scenarios to assess the effects of catchment load reductions on internal loading to the lake, and we use modifications of the calibrated model to estimate the degree to which post-2006 alum dosing has altered internal loading to the system, thus reducing the 'total load' (external plus internal) to the lake. By examining these relationships, we aim to inform on the extent that presently good water quality (in a relative sense) might have been driven by geoengineering (i.e., alum dosing of the two inflows) as opposed to changes in catchment loading and/or climate. Several of the model scenarios are designed to assess what level of catchment N and P load reductions might be required to meet the TLI target of 4.2 without the need for ongoing, long-term geoengineering.

Model performance was comparable, and in many cases superior to, previously published applications of DYRESM-CAEDYM to Rotorua and other lakes. Simulations of catchment nutrient load change scenarios suggest considerable potential for water quality improvement through catchment load management. Simulations projected that through catchment N and P reduction, and without the need for alum, TLI could be reduced from approximately 4.8 (as observed 2001-2007) to approximately 4.3, or near the target of 4.2 for specified under Plan Change 10. While the TLI reduction from 4.8 to 4.2 appears modest, the absolute reductions in TP, TN and chl *a* are relatively large due to the logarithmic nature of the TLI component equations. For example TP and TN mean concentrations were reduced by c. 75% between the maximum and minimum load scenarios. Furthermore, although TLI under the lowest loading scenarios were reduced to only c. 4.3 (i.e., the TLI did not meet the target), historical loading and TLI estimates showed that catchment N load prior to 1970 was likely to be less than the target load of 435 t N $y^{-1}$, and that the lake TLI may possibly have been >4.2 by the late-1960s. Scenarios of increased loading suggested that substantial further degradation could occur if present trends of increases in the catchment nitrogen load are left unchecked. Only synergistic effects of reductions in both nitrogen and phosphorus loads resulted in simulated TLI near the target levels. This finding is consistent with previous research, including other DY-CD applications, and with the weight of scientific evidence that dual nutrient management and reduction is needed for restoration to be effective for Lake Rotorua.

There is growing evidence that in Lake Rotorua, varying nutrient loading patterns and the application of alum has produced periods of co-limitation by N and P. Part of this evidence relates to reduced water column P post-alum application. For example, after 2010 there has been a reduction of dissolved reactive phosphorus (DRP) in monthly water quality sampling data, and surface DIN concentrations have sometimes been strongly elevated compared with historical levels. We calibrated and validated the DY-CD model for the period 2001-2007 in order to avoid the potential for a confounding influence of alum dosing from 2006 onwards, and particularly from 2010 when dosing rates were strongly elevated (Figure A). When the calibrated model was run from 2001-2017 without any consideration given to alum dosing, this 'baseline' model greatly overestimated nutrient and chlorophyll concentrations for the alum dosing period post 2006. In this sense, the 'baseline' model simulation over the period 2007-2017 provides an estimate of what water quality in Lake Rotorua may have been if alum dosing had not occurred (Figure A). Because the calibrated model accounts for changes to external loading and climate over the pre- and post-alum periods, these simulations provide weight to the proposition that alum dosing of Puarenga and Utuhina Streams has positively impacted lake water quality over and above any benefit caused by direct adsorption of DRP by alum in the two streams. Although both dosing rates and water quality varied widely from 2007-2017, our simulations showed on balance that alum dosing had a similar effect to reducing modelled nutrient release rates (internal load) by c. 50%. It should be noted that internal loads are dominated by dissolved (largely bioavailable) nutrients, whereas catchment loads contain relatively more particulate and organic species, only some of which may become bioavailable at longer time-scales.

The present study highlights opportunities to progress the scope and robustness of catchment and lake modelling for Lake Rotorua. Priorities for development over the next 5-year cycle under LR M2 would include:

- Modelling of climate change effects on load delivery and in-lake processes.
- Model ensemble, including multiple models with different process representations of internal load (nutrient recycling). This approach would likely generate a wider, but potentially more realistic, range of potential outcomes resulting from management strategies.
- Modelling higher trophic levels and food web processes
- More detailed uncertainty estimation, including Monte Carlo (MC) or Bayesian Markov Chain Monte Carlo techniques, to further improve confidence in outputs by generating an envelope of expected outcomes.
- Coupled catchment-lake modelling, including a catchment model such as the Soil and Water Assessment Tool (SWAT), capable of modelling the role of contaminant mobilisation and delivery as well as associated mitigation options.

In summary, this project represents an evolution of process-based lake model application to Rotorua, refined over the past 12 years and multiple studies. Detailed modelling of Lake Rotorua catchment inputs and in-lake processes have shown that substantial reductions in nutrient loading will be required to achieve water quality at or near the TLI target (Figure B) whilst also reducing or removing the need for actions such as alum dosing. Modelling results also support the assertion that alum dosing has altered in-lake processes leading to a reduction of both nitrogen and phosphorus, as well as phytoplankton, over the past 10 years (Figure A). These simulations reinforce the urgent need to work towards established nutrient targets (435 t TN $y^{-1}$ and 37 t TP $y^{-1}$), and the importance of combined reductions in both nitrogen and phosphorus loads to achieve the TLI target of 4.2.

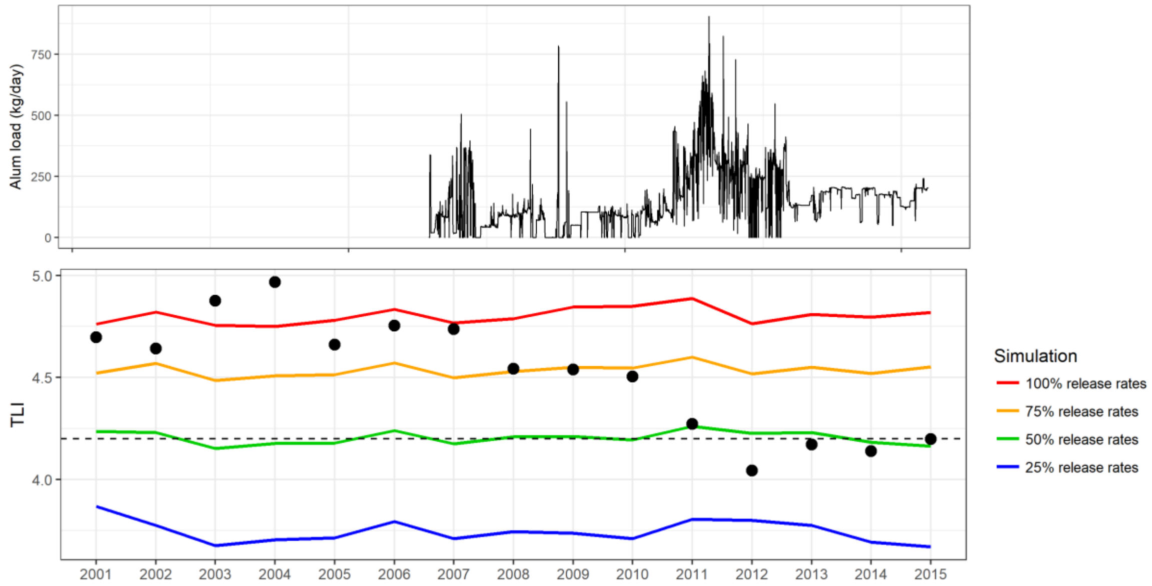


Figure A. Alum dose rate (top, expressed as kg Al per day), and bottom, observed TLI (black dots) along with TLI simulated under four different scenarios of sediment nutrient flux rates. Release rates of 100% represent the 'baseline' calibrated DY-CD lake model, and other scenarios represent reductions in modelled sediment nutrient flux rates. Simulations show that stream alum dosing may have had an effect similar to a reduction of c. 50% of internal nutrient load, and that this effect varies with dose rate.

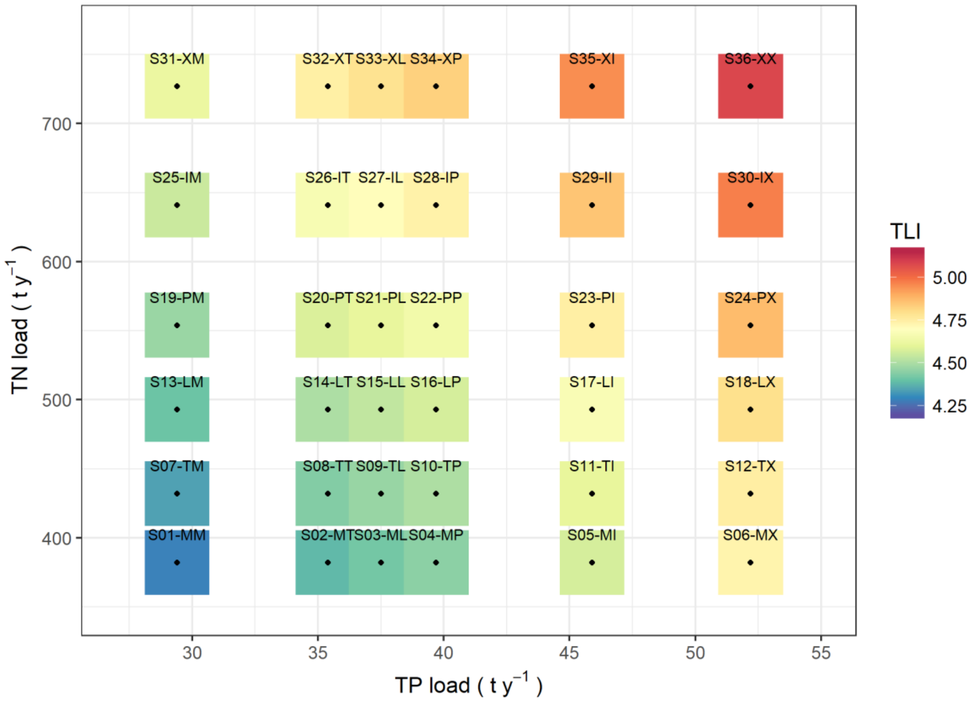


Figure B. Modelled TLI under 36 scenarios covering various combinations of catchment TN and TP loading. Results presented demonstrate the influence of TP and TN loading (t $y^{-1}$) on TLI over the scenario period from July 2000 to June 2007. The lowest simulated TLI was 4.29 (S01-MM) and the highest simulated TLI was 5.08 (S36-XX). Each point and coloured tile on the plot represents one nutrient loading scenario corresponding to its position on the X and Y axes. The last two letters of each scenario name represent the N and P load, respectively, where: M = minimum load, and T=Target load, L = lower load, P = Present load (2001–2007), I = Increased load, and X = Maximum load.

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## Introduction

Lake Rotorua has a history of catchment urbanisation and pastoral development, processes which have been linked to water quality decline observed in inflows and in the lake between the 1960s and 2010s (Hamilton et al., 2015; Hoare, 1980; Rutherford et al., 1989; Rutherford et al., 2011). The scientific and public concern generated by lake eutrophication has spurred an extensive history of water quality study within the Rotorua Lakes region, with 400 journal articles, theses and reports published between 1922 and 2002 (Miller, 2003). In addition, active community and iwi engagement alongside extensive water quality management by Bay of Plenty Regional Council (BoPRC) has resulted in a long history of catchment management and eutrophication mitigation strategies.

Early scientific research within the Rotorua lakes focused on geochemistry and geothermal influences (Phillips & Grigg, 1922), and trout ecology (Phillips, 1924), however interest turned to aquatic macrophytes after the establishment of invasive *Lagarosiphon major* and *Egeria densa* in the 1950s (Anderson, 1964; Fish, 1963). Concern over lake eutrophication and algal blooms developed in the 1960s and led to early limnological studies of Lake Rotorua (Anderson, 1965), and comparison to other New Zealand lakes (Jolly, 1968), alongside biological studies of microscopic algae (phytoplankton) (Cassie, 1969) and zooplankton (Chapman, 1969). Links were made between eutrophication of Lake Rotorua and catchment development and land use intensification (McColl, 1972). These factors led to the formation of the community-led Lake Weed Society, now known as the Lakes Water Quality Society (https://lakeswaterquality.co.nz/) whose vision is to restore the 12 lakes of the Rotorua District to the state they were prior to the 1960s.

The expansion of invasive macrophytes led The Department of Lands and Survey to develop a weed spraying program which continues today. A number of scientific studies were carried out which analysed ecological effects of weed spraying, focusing on changes within phytoplankton communities (Cassie, 1967). Subsequently, attention again turned to eutrophication, with focus on the study of external nutrient loading from the lake catchment (Fish, 1970a, 1970b) and the first nutrient budgets being generated (Fish, 1971; Fish & Andrew, 1971). Through the 1970s research continued into lake eutrophication (Burnet & Wallace, 1975; McColl, 1974; E. White, 1974), and identified the importance of storm events in contributing to nutrient loads (Fish, 1975; Hoare et al., 1976; Mitchell, 1976). In the late 1970s sources and sinks of nutrients were first attributed to lake sediments with identification of internal nutrient loading (Fish, 1979; Matthews, 1979; White et al., 1978).Multiple studies throughout the 1980s described accelerating eutrophication of Lake Rotorua and again focused on the relationship between land use and lake trophic state (Hoare, 1981, 1982, 1983; Rutherford, 1985; Timmins & Savage, 1981; Vincent et al., 1984; White, 1983b). Nutrient limitation of algal growth in Lake Rotorua and other New Zealand lakes was studied in the early 1980s, and nitrogen was found to be a limiting nutrient in some lakes, including Rotorua (Vincent, 1981a, 1981b; White, 1983a; White et al., 1985). This contrasted with global lake phytoplankton limitation assessments tending strongly towards phosphorus limitation, particularly in the northern hemisphere (Abell et al., 2010).

In the late 1980s, 'sustainable catchment nutrient loads' were established in peer-reviewed scientific literature (Rutherford et al. 1989), and a water quality target was set to a Trophic Level Index (TLI, Burns et al. 1999) value of 4.2, on the basis that this would reflect water quality similar to that in the 1960s, a state to which the community aspired to restore the lake. Since the 1990s multiple strategies have been developed to reduce external nutrient loads in order to improve the water quality of Lake Rotorua (Gibbs & Lusby, 1996; Macaskill et al., 1997; McIntosh, 1993; Rodda et al., 1997; Rutherford et al., 1996; Williamson et al., 1996). Treated wastewater (sewage) was discharged directly to Lake Rotorua from 1973 until 1991, when a new treatment plant was established which discharged to 300 ha of plantation forest – the Whakarewarewa Forest Land Treatment System (LTS). Phosphorus is bound mostly to the highly adsorptive sediments in the forest and nitrogen that is not taken up by trees or denitrified is otherwise leached as nitrate to groundwater, which discharges to Lake Rotorua through the Waipa and then the Puarenga Stream (Me et al., 2018). Nutrient loads to the lake were greatly reduced by the LTS and lake water quality measurements initially indicated increased water clarity (Rutherford et al., 1996). The current resource consent for this discharge ends in 2021.

In the early 2000s, nutrient budgets were revisited (Hamilton et al., 2004; Rutherford, 2008). Because Lake Rotorua consistently failed to meet its water quality target, BoPRC, along with Te Arawa Lakes Trust and the Rotorua District Council, developed the Lake Rotorua 'Action Plan' (BoPRC, 2009). The plan specifies that land use and management practices should be suitably aligned with reducing catchment nutrient loads. Catchment nutrient loads include all inputs of nutrients from outside the lake boundary (surface and groundwater), hereafter referred to as 'external loading'. Additional nutrient loads may be supplied to the water column fropm lake sediments, wither by wind-driven resuspension or by chemical release (desorption) from sediments, hereafter referred to as 'internal loading'. Artificial nutrient controls to stream inflows and/or lake bottom sediments could also be considered, ultimately aimed at reducing total (external + internal) nutrient loading to levels similar to 1960 (Rutherford, 1984). The Action Plan posited that external nutrient loading of TN as at 2009 would need to be reduced by 112 t $yr^{-1}$, and a reduction of 224 t $yr^{-1}$ would be required from the projected load at 2055 (which is higher due to increasing concentrations resulting from groundwater time lags). A required reduction of 2 t $y^{-1}$ from the 2009 TP load was also estimated.

The coupled hydrodynamic-ecological model DYRESM-CAEDYM (DY-CD) was first applied to Lake Rotorua by Burger et al. (2008) for the period 2001 to 2004. This study showed that at this time internal loading from nutrient releases from the lake sediments was substantially greater than the external load derived from the catchment (see Burger et al. 2008; BoPRC 2009). Further applications of the DY-CD model assessed the effects of different nutrient loading regimes and climate change on Lake Rotorua trophic state (Hamilton et al., 2012). Climate change was shown to have a negative effect on water quality, whereby model simulations using projected climate for the period 2091-2099 resulted in TLI increase from 4.79 to 4.92 and increased frequency of cyanobacteria blooms.

Alum dosing commenced in the Utuhina Stream in 2006, and in the Puarenga Stream in 2010, and water quality after 2010 was generally at or better than the TLI target of 4.2. Hamilton et al. (2015) updated the lake model and applied it to assess the likely effects of inflow alum dosing. The DY-CD model simulations were only able to approach the TLI target by assuming direct effects of alum locking in stream inflows as well as including internal load suppression and increased particulate matter deposition. Thus, the study showed that alum dosing not only reduced phosphate concentrations within treated inflows, but likely had an in-lake effect whereby phosphate within the lake water column was “locked up” as well as a reduction in sediment phosphorus releases. The study concluded that water quality improvements within the lake were derived from a regime shift where nutrient limitation moved from primarily N-limitation to more frequent co-limitation and possibly even P-limitation during very high alum dose rates.

Abell et al. (2015) compared the influence of treated wastewater discharge on Lake Rotorua water quality under different treatment systems and discharge regimes (LTS or directly to the lake), again using DY-CD. Model simulations showed that the effects associated with different treatment options (on modelled TLI) would be neutral to minor, with very small changes in TLI values of <0.01 to 0.02 units. However, 3-D simulations demonstrated that wind driven circulation patterns had potential to lead to locally elevated concentrations of treated wastewater, although concentrations of treated wastewater remained below 1% of all water even within the near shore locations where concentrations were most elevated.

### Objectives

The purpose of the present study was to build on previous applications of DY-CD to Lake Rotorua, specifically by improving the estimation of daily inputs of water, nitrogen and phosphorus to the lake, and by expanding the number of future nutrient loading scenarios to encompass a matrix of combinations of nitrogen and phosphorus loads. We simulated a variety of management strategies (N vs P catchment loading reductions) and their effects on lake water quality. Further, we used the modelled scenarios to assess likely effects of catchment load reductions on internal loading to the lake, and used modifications of the calibrated model to estimate the degree to which alum dosing post-2006 has altered internal loading, thus lowering the total load (external plus internal) to the lake. By considering these aspects, we aimed to evaluate whether the recent improvements in water quality have been driven by lake-geoengineering (alum dosing) or by potential changes in catchment loading and climate, and to identify the level of catchment N and P load reductions that would be required to achieve similar water quality without the need for ongoing, long term geoengineering.

# Methods

## Study site – Lake Rotorua

Lake Rotorua is a polymictic, eutrophic lake with surface area of 80.8 km$^2$, a catchment of 500.5 km$^2$ (Figure 1), mean depth of 10.3 m, and maximum depth of 52.9 m. However, the area of depth greater than 26 m comprises 0.14 km$^2$ or just 0.2 % of total lake area. This deep area contains a steep sided crater at the south end of the lake, near Sulphur Bay (Figure 2). Water quality of Lake Rotorua has varied substantially over past decades (Figure 3), with periods of very poor water quality (sometimes supertrophic) in the 1980s and 2000s, and widely publicised severe cyanobacteria blooms in the early 2000s. Two surface inflows, the Puarenga and Utuhina Streams, have been dosed with aluminium sulphate (alum) since 2006 and 2010, respectively. Water quality presently generally meets the target of TLI ≤ 4.2 (Figure 3).

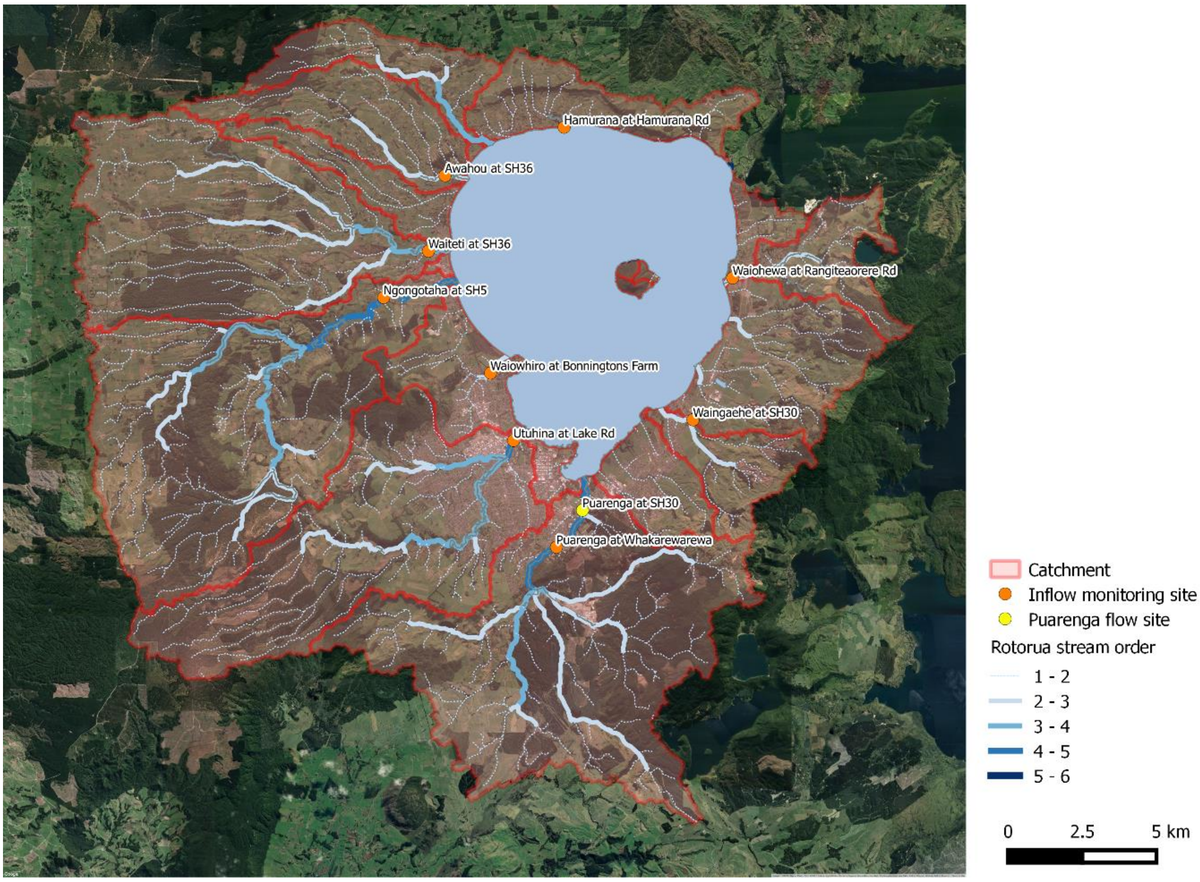


Figure 1: The catchment of Lake Rotorua, showing sub-catchment boundaries (red line), inflow water quality/discharge monitoring sites (orange dots), and the location of the Puarenga discharge monitoring site (yellow dot). Stream order is represented by line thickness (REC; NIWA). Detail in map key.

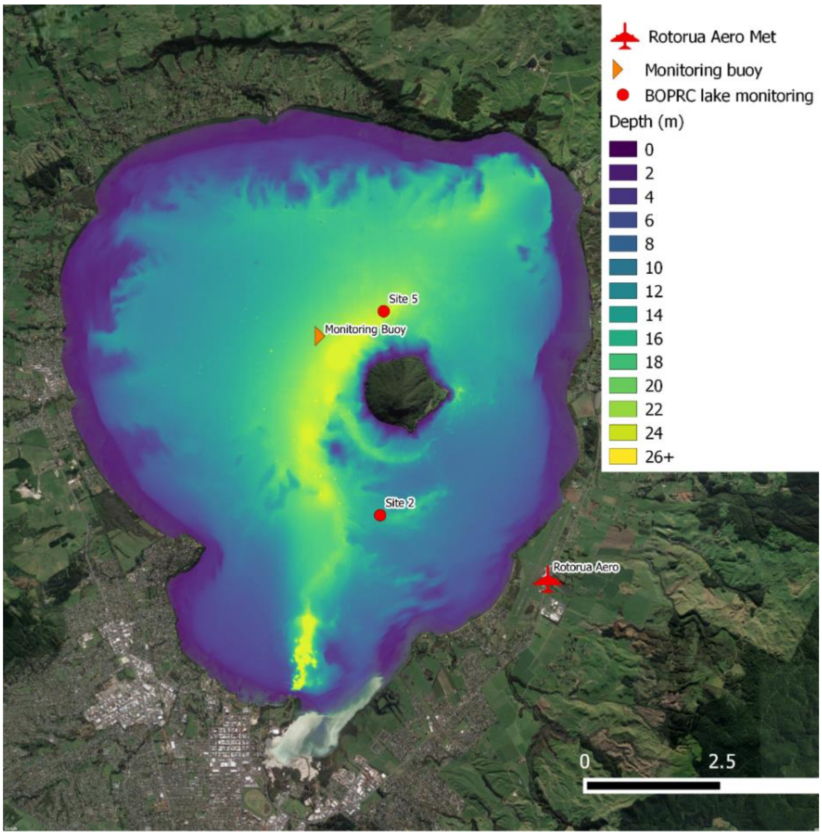


Figure 2. Lake Rotorua bathymetric map (colour scale). Also shown are the meteorological station (Rotorua Aero; red plane symbol), BoPRC lake water quality monitoring sites 2 and 5 (red dots), and the automated monitoring buoy (orange triangle). Bathymetric data are from side-scan sonar survey with a 120˚-wide fan perpendicular to the vessel (Coastal Marine Group, University of Waikato, 2006).

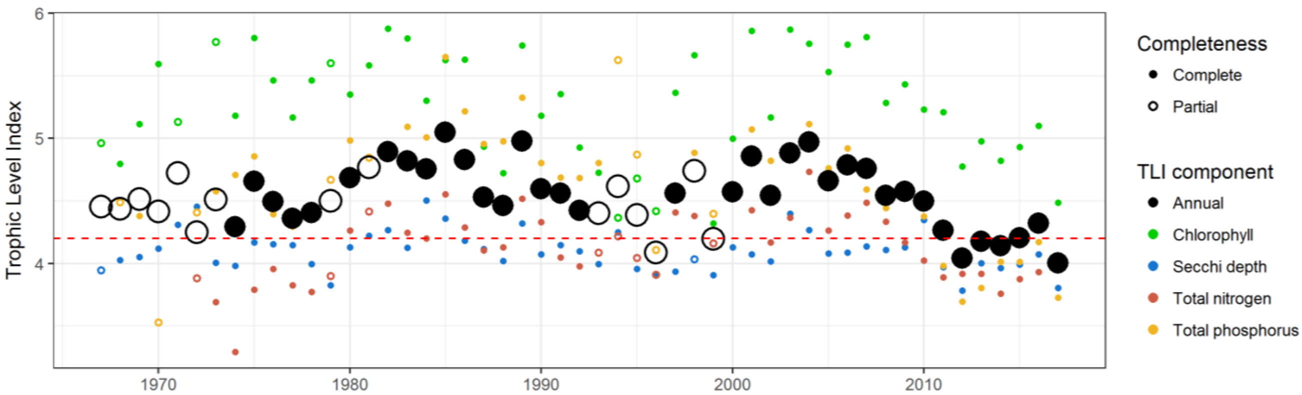


Figure 3. Annual Trophic Level Index (TLI) at mid-lake sites by hydrological year (previous July to present June). Each coloured circle is the mean of seasonal means, with the TLI equation (Burns et al. 1999; eqns 25 p. 23) applied. Large black circles are the annual TLI (average of four components). Solid circles denote years for which at least one measurement was available for all four seasons (solid black circles denote that all four component variables of the TLI were sampled each season). Open circles denote that measurements were missing for at least one season. The lake TLI target (4.2) is shown by the dashed red line. Values may differ from those routinely reported by BoPRC due to the aggregation method used in these calculations.

## Lake model description

The one-dimensional (1D) hydrodynamic model DYRESM (version 3.0.0-69) was coupled with the aquatic ecological model CAEDYM (version 2.3.1-01), in order to simulate water quality in Lake Rotorua. The model was developed at the Centre for Water Research, The University of Western Australia. DYRESM resolves the vertical distribution of temperature, salinity, density, and the vertical mixing processes in lakes and reservoirs. The version of DYRESM used was specifically modified to allow additional thermal flux representative of geothermal heat input to the bottom of the lake, with this model version created and maintained by Dr Bob Spigel (National Institute of Water and Atmospheric Research, New Zealand). 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 suspended particulate matter. Many applications have been made of DYRESM-CAEDYM to different lakes (e.g., Burger, 2006; Gal et al., 2009; Özkundakci et al., 2011; Trolle et al., 2011) and these publications contain detailed descriptions of the model equations.

### Bathymetry

Model bathymetry was derived from a multibeam echo sounding survey completed in 2006 by the Coastal Marine Group at the University of Waikato (120˚-wide fan perpendicular to the vessel). Summary area and volume statistics were calculated from a 5 m resolution raster depth coverage (Figure 4). Note that the deepest part of the lake is a small crater near Sulphur Bay with a depth of c. 40 m, however, because this area is very small the lake bathymetry was truncated at a depth of 25 m for the model simulations.

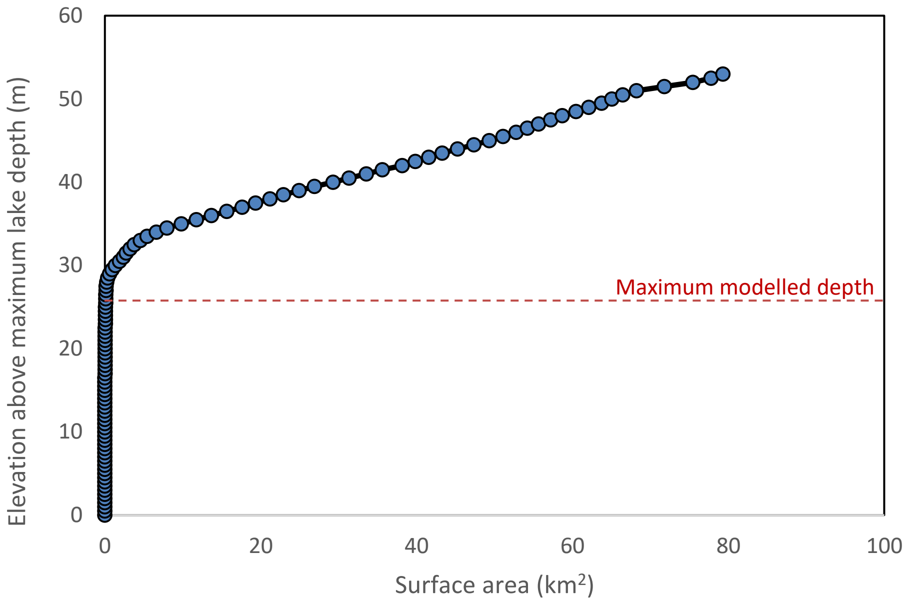


Figure 4. Hypsography of Lake Rotorua, derived from multi-beam echo sounding (see Figure 2). Note that the deepest part of the lake is a small crater near Sulphur Bay and the surface area below 25 m elevation is too small to show at the scale of the x axis.

### Meteorology

Meteorological data required for simulations (1997–2017) were obtained from the National Climate Data Base (1989 to 2012) and the New Zealand Metservice (2013 to 2017), for the Rotorua Airport climate station c. 50 m from the Lake Rotorua shoreline (location shown in Figure 2). Variables 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 5). Data are collected at Rotorua airport at various time intervals from one hour to whole-day, and for the purposes of model input were standardised to daily average values except for rainfall, which was a daily total value.

Airport air temperature ($T_a$) for the modelling period was adjusted using regression with air temperature measurements of 2007 – 2012 from a high-frequency monitoring buoy at the central lake in order to account for air mass modification by thermal inertia effects of the lake. The model for modified air temperature ($T_r$) was ($r^2$=0.97, p<0.01):

$$T_r = 0.0124\, T_a^2 + 0.5177\, T_a + 4.7714 \qquad \text{Equation 1}$$

### Inflows and outflow

Surface inflows to the lake were derived from gauging and water quality monitoring locations administered by BoPRC (locations shown in Figure 1), with a combination of linear interpolation between water quality and flow measurements (for missing data and notcontinuous data), and storm flow modelling adapted from methods previously used by Abell et al. (2015). The method considered nine major inflows, ungauged quick flow (representing ungauged overland flow), slow flow (representing groundwater direct to the lake), and geothermal sources (Figure 6).

Methods used to estimate daily inflow volumes (Figures 7 to 9) and whole-lake water balance (Figure 6) are described in detail in McBride et al. (2018).

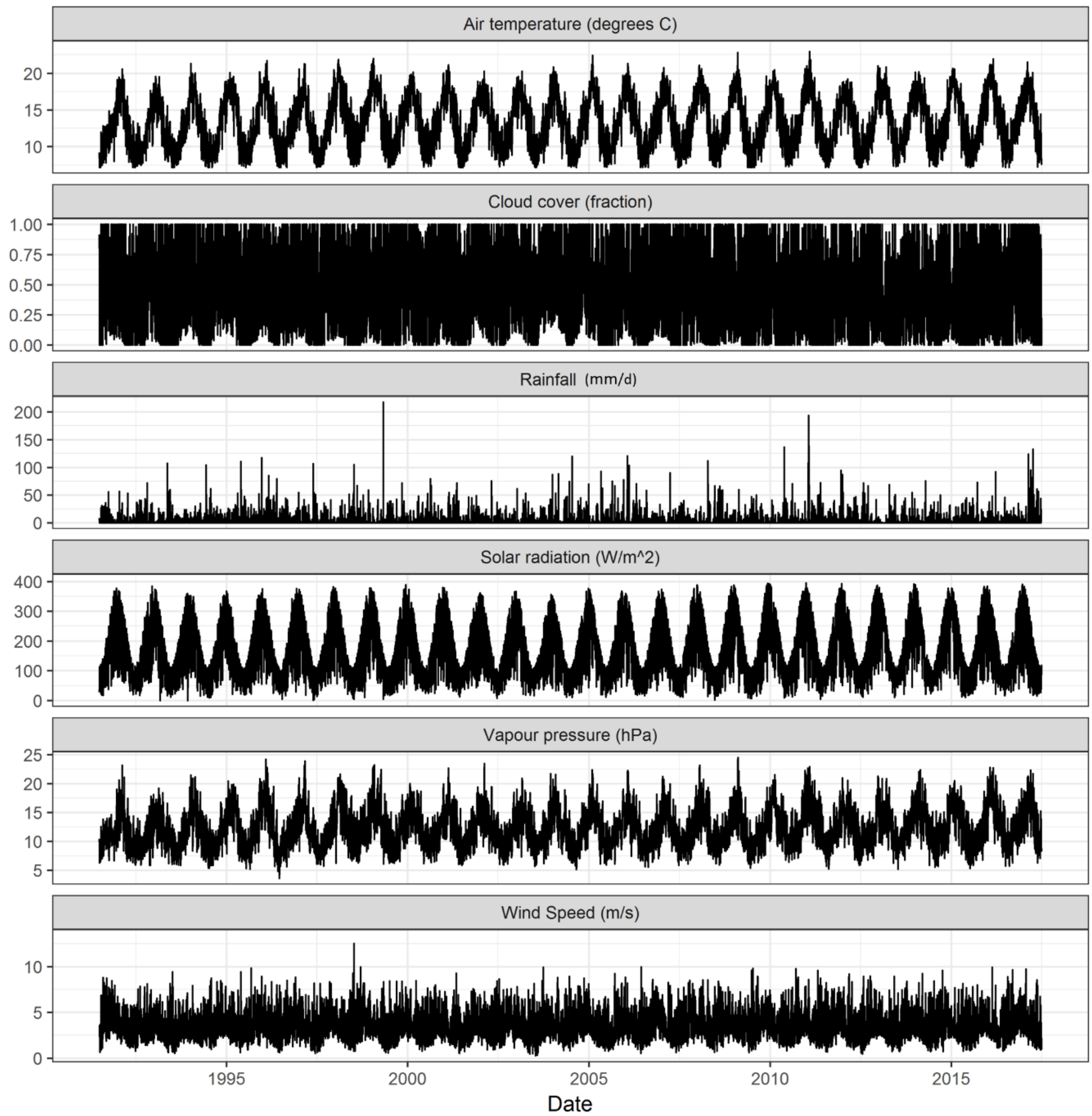


Figure 5. Meteorological forcing variables of air temperature (°C), cloud cover (fraction of whole sky), rainfall (m), solar radiation (W $m^{-2}$), vapour pressure (hPa) and wind speed (m $s^{-1}$), collected from Rotorua Airport climate station (location shown in Figure 2).

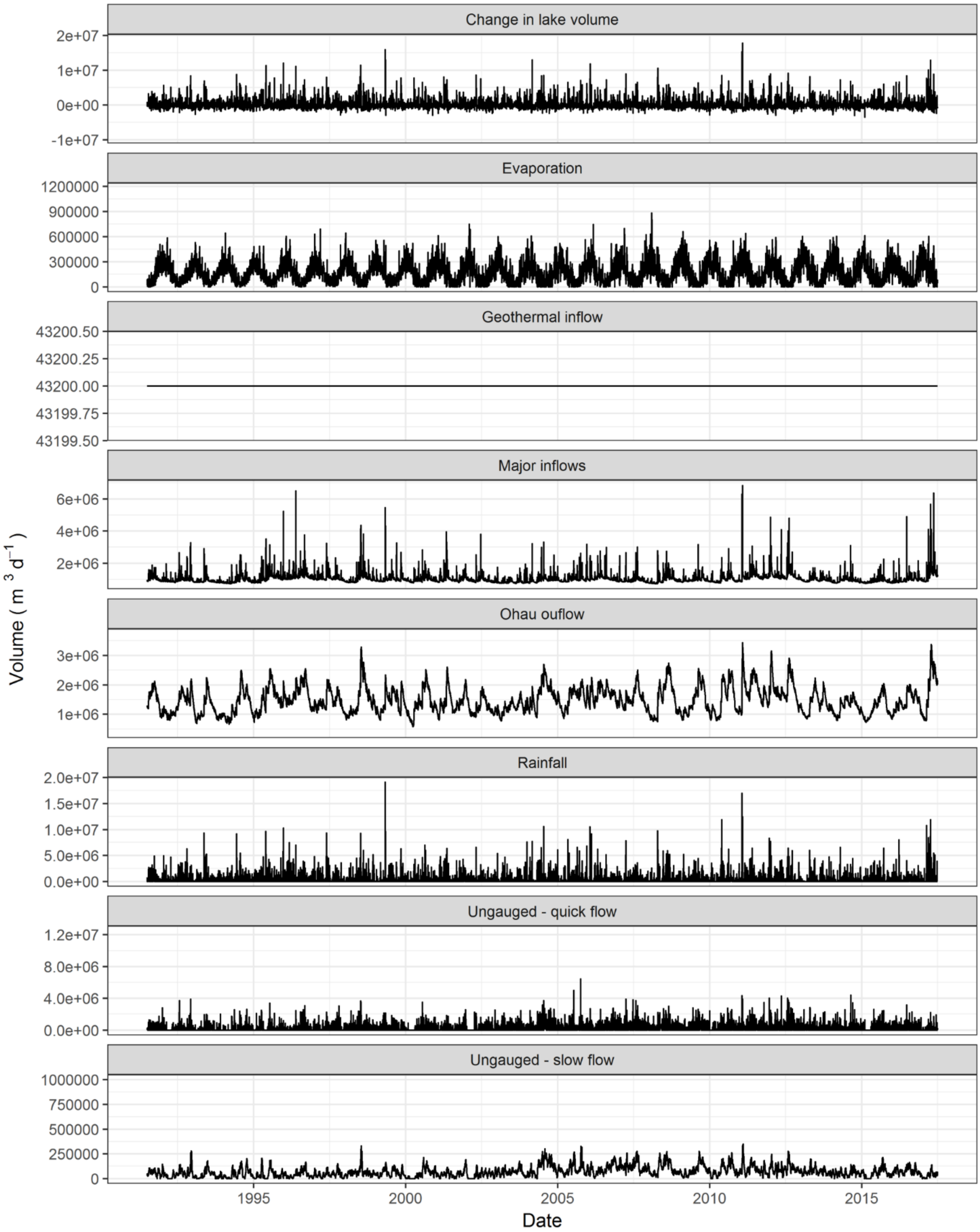


Figure 6. Lake Rotorua interpolated and modelled inflow summary and water balance components (McBride et al. 2018).

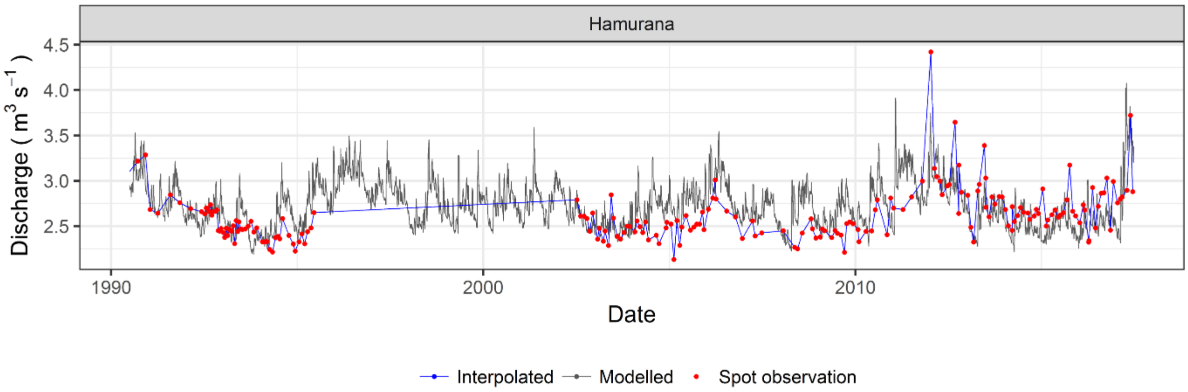


Figure 7. Comparison of modelled (grey line) and gauged (red dots with blue connecting line) discharge for Hamurana Stream.

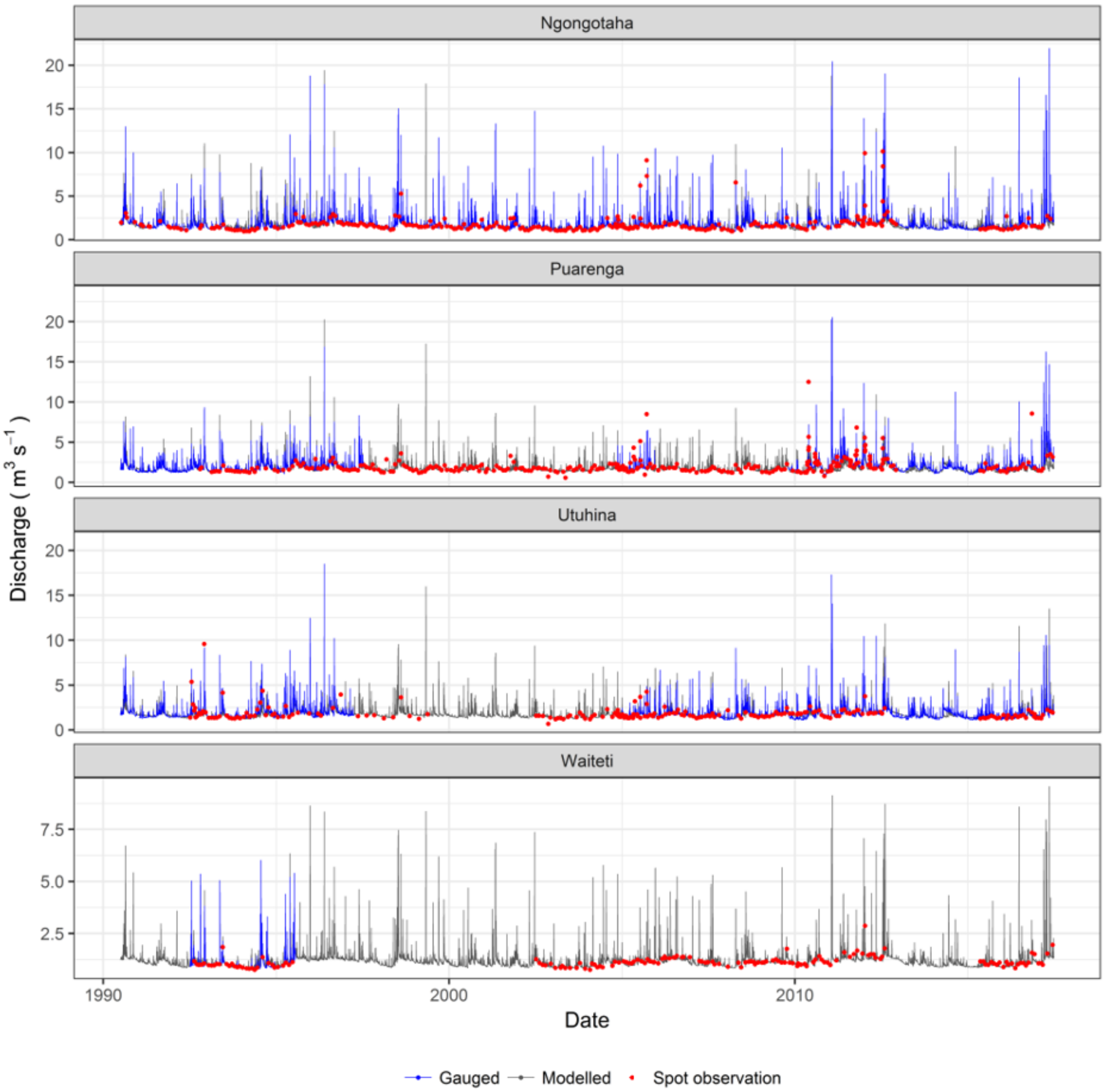


Figure 8. Comparison of modelled (grey line) and gauged (blue line) discharge for four major inflows, with spot observations (red dots).

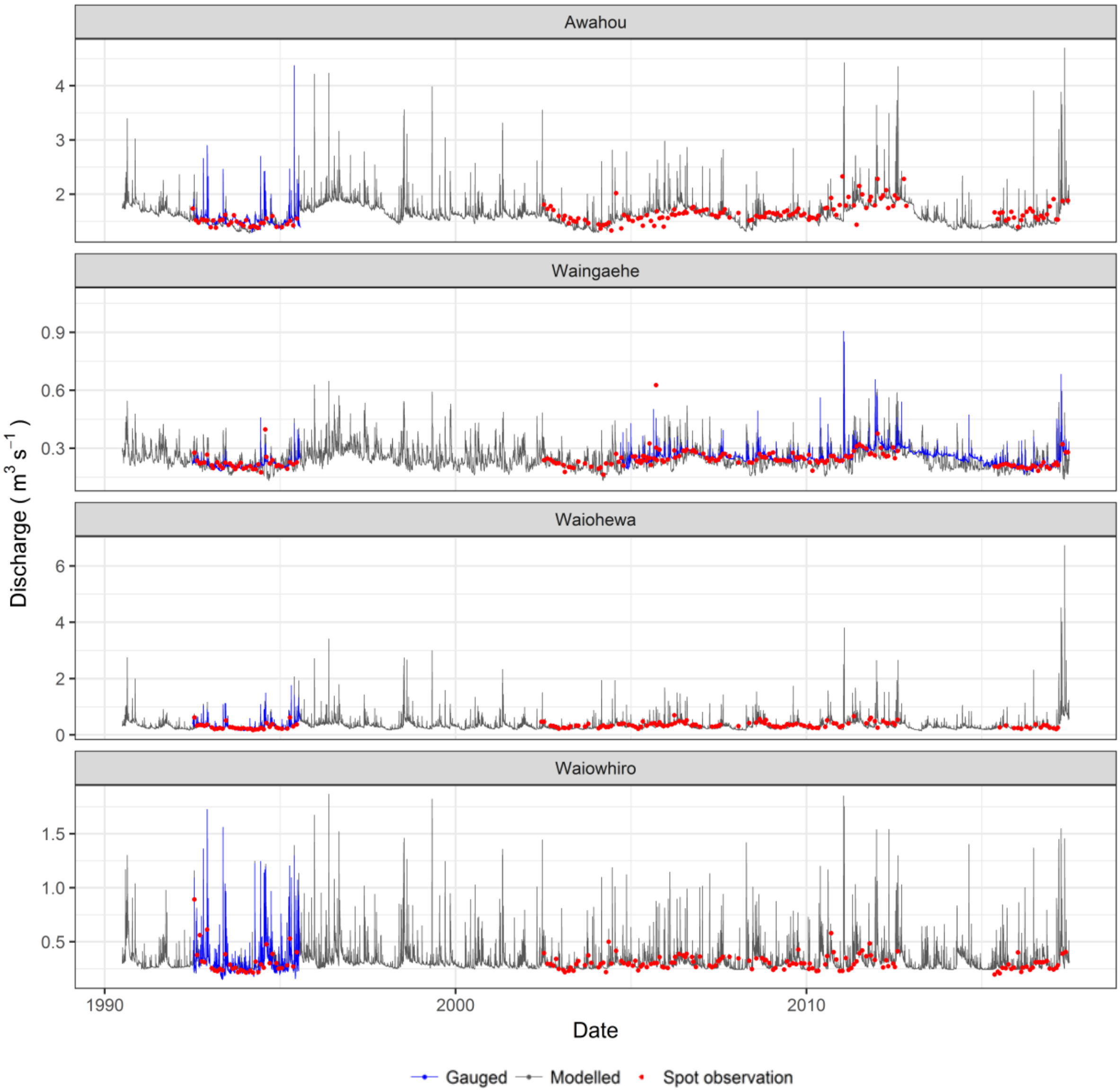


Figure 9. Comparison of modelled (grey line) and gauged (blue line) discharge for four major inflows, with spot observations (red dots).

## Nutrient concentrations

Daily nutrient and sediment concentrations for all water inputs to the DY-CD model were collated and/or modelled. Approximately monthly measurements for most major stream inflows were available (source: BoPRC), and daily concentrations were derived by a combination of linear interpolation and/or concentration-discharge relationships observed during stormflow events. Methods used and the results of these approaches are described in detail in McBride et al. (2018).

Lake Rotorua inflow loads used in the model calibration were dominated by major streams, however geothermal sources and rainfall still made large contributions to the nutrient budget (Figure 10). Refer to McBride et al (2018) for more discussion of source contributions to overall loads.

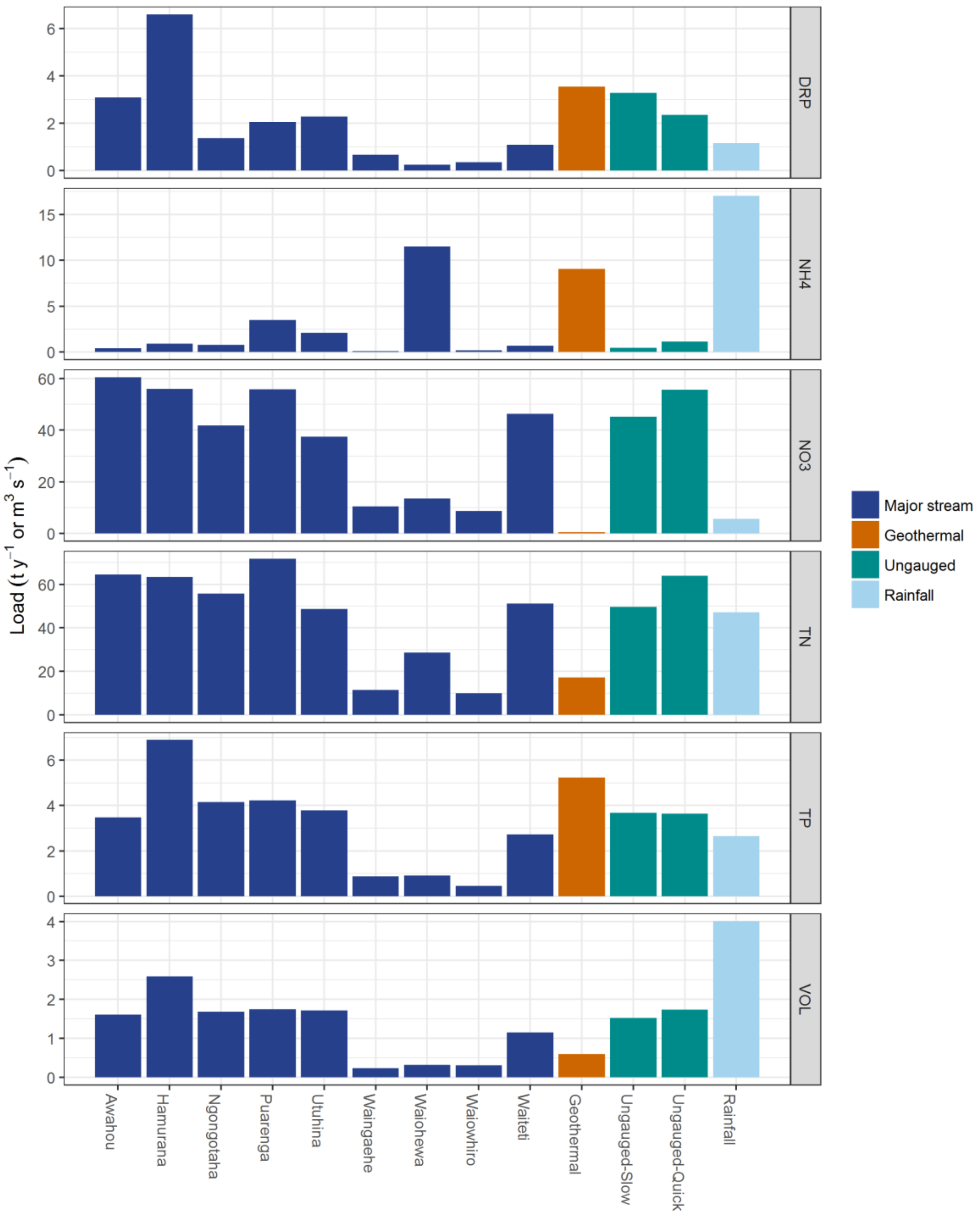


Figure 10. Nutrient (t y$^{-1}$) and hydraulic (m$^3$ s$^{-1}$) load averages for the hydrological years 2001 – 2007, derived from inflow modelling (McBride et al. 2018) and used in model calibration.

## Phytoplankton

Three functional groups of phytoplankton were simulated in the lake model: cyanobacteria, diatoms and a third category broadly representative of chlorophytes and other taxa. The Hamurana and Awahou inflows, and rainfall contributions, were given a concentration of zero for each of the three phytoplankton groups simulated. Those inflows not dominated by groundwater springs were prescribed a 'seeding' concentration of 0.1 µg chl *a* L$^{-1}$ for each phytoplankton group.

## Model simulation periods

In this study, a DYRESM-CAEDYM (DY-CD) model of Lake Rotorua was configured with daily boundary inputs of meteorology, catchment discharge and inflow water quality, for the period of July 2000 to June 2017. The model was run with an internal time-step of one-hour, with daily output obtained at midday (instantaneous output). For each simulation, modelled state variables were initialised using in-lake observations (source: BoPRC) from the nearest date to the starting date of the simulation. For model calibration, the period July 2001 to June 2004 was used, and the independent time period for model validation was July 2005 to June 2007. These periods were selected in order that the model could be calibrated and validated without the potentially confounding influence of alum application to the Utuhina (2006 to present) and Puarenga (2010 to present) inflows (Hamilton et al. 2015). We assumed that any in-lake impacts of alum dosing prior to June 2007 would be minor due tot eh short duration of dosing and relatively low dose rates at the time.

36 catchment N and P load change scenarios were simulated using meteorological and catchment inputs for the period 1 July 2001 to 30 June 2007 (i.e. the combined calibration/validation period. Four additional scenarios were simulated for the period 2001 to 2017, with modified internal (sediment) nutrient release rates to assess the possible effects of alum dosing on internal loading over the past 10 years.

## Calibration and validation

Monthly samples collected and analysed by BoPRC were used to assess model performance. DYRESM-CAEDYM was calibrated against field data from Lake Rotorua Site 2, for a four-year period between July 2000 and June 2004 for variables of temperature, DO, $PO_4$-P (dissolved reactive phosphorus), TP, $NH_4$-N, $NO_3$-N and TN. Near-surface samples were collected using a tube sampler from 0 – 6 m in the water column, and a Schindler-Patalas trap from between 18 and 20 m. These field samples were compared with simulation output from the mid-points of these depth ranges (3 m and 19 m), respectively, because obtaining average simulation values across specified depth ranges was impractical. The three simulated phytoplankton groups collectively contributed to a total simulated chlorophyll *a* (chl *a*) concentration, but with cyanobacteria 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). The sum of the chl *a* concentrations for all three groups was calibrated against surface chl *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 maintained within literature ranges. (e.g., Schladow and Hamilton, 1997; Trolle et al., 2011), with calibrated values given in Appendix 1. 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 values from modelling studies in the literature, to assess an acceptable model error for prediction purposes.
The TLI value was calculated for each year of the simulation period. The relevant equations for determination of the TLI are:

TLc = 2.22 + 2.54 *log*( chl *a* ) Equation 2

TLs = 5.10 + 2.27 *log*( 1/SD - 1/40 ) Equation 3

TLp = 0.218 + 2.92 *log*(TP) Equation 4

TLn = -3.61 + 3.01 *log*(TN) Equation 5

This enables the average trophic level index to be calculated:

TLI = ( TLc + TLs + TLp + TLn)/4

where:

$TL_c$, $TL_s$, $TL_p$ and $TL_n$ represent the individual level trophic level indices for the individual variables of chl *a* (mg $m^{-3}$), Secchi disk depth (m), total phosphorus (mg $m^{-3}$) and total nitrogen (mg $m^{-3}$).

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.

### Secchi depth estimation from model output

As Secchi depth is not explicitly included as a state variable in DY-CD, this variable was derived from the simulated daily attenuation coefficient for photosynthetically active radiation (PAR) as (Holmes, 1970):

$$Z_{SD} = 1.5K_d \qquad \text{Equation 6}$$

where:

$z_{SD}$ is the Secchi depth (m)
1.5 is a constant obtained by analysing field measurements of light profiles and corresponding Secchi depths.
$K_d$ is the diffuse attenuation coefficient for PAR ($m^{-1}$), calculated after (Gallegos, 2001):

$K_d = K_w + K_c * \text{chl}a + K_y * \text{DOC} + K_s * \text{TSS}$

where:

$K_w$ is the background extinction coefficient, with a value of 0.2 ($m^{-1}$).
$K_c$ is the specific attenuation coefficient (SAC) for chl *a* with a value of 0.012 ($m^2$ (mg chl *a*)$^{-1}$) (Gallegos, 2001).
$K_y$ is the SAC for dissolved organic carbon with a value of 0.0507 $m^2$ (g DOC)$^{-1}$.
$K_s$ is the SAC for total suspended sediment with a value of 0.05 ($m^2$ (g TSS)$^{-1}$).

NOTE: DOC was not used in this calculation, but was loosely accounted for in the background attenuation term ($K_W$) of the DY-CD calibration.

## Global model sensitivity analysis

A global sensitivity analysis was used to determine the sensitivity of model output for key water quality variables to change in model parameters. This analysis therefore informed the calibration process, and provides a guide for future uncertainty analysis. A total of 79 parameters were analysed, and values were varied randomly within bounds of ±25% of the calibrated value. For this preliminary sensitivity analysis, the model was run for one hundred iterations whereby all 79 parameters were adjusted randomly for each iteration, and the coefficients of correlation and determination was calculated. Parameter sensitivity was inferred by the relationship betweenparameter values and model performance for TP, TN, chl *a* and Secchi depth.

## Model scenarios

### *Catchment nutrient loads*

Simulations were undertaken to assess different catchment nutrient loading scenarios. A '6 x 6' matrix of nitrogen and phosphorus loading scenarios (n = 36) was generated (Figure 10, Table 1) by running each load combination from July 2000 to June 2007. Results for each TLI variable were retained for the last six years of each simulation averaged to obtain projected TLI outcomes for each load combination. The nutrient load change chosen for the scenarios was informed by recent estimations of historical nutrient loads (McBride et al. 2018), and simulations of catchment nitrogen load using the catchment model ROTAN-Annual (Rutherford et al. 2016). The highest TN loads account for groundwater lag times and little mitigation (727 t N $yr^{-1}$ inclusive of atmospheric deposition), similar to estimates presented in Rutherford (2016) of 700 t N $y^{-1}$ (exclusive of atmospheric deposition). The TN load target (435 t N $y^{-1}$) was also simulated, as well as a scenario lower than the target (TN 382 t $yr^{-1}$). Within this lowest loads scenario, the TP loads were adjusted using similar proportional change (S36-XX). Present nutrient loading was also simulated (S22-PP).

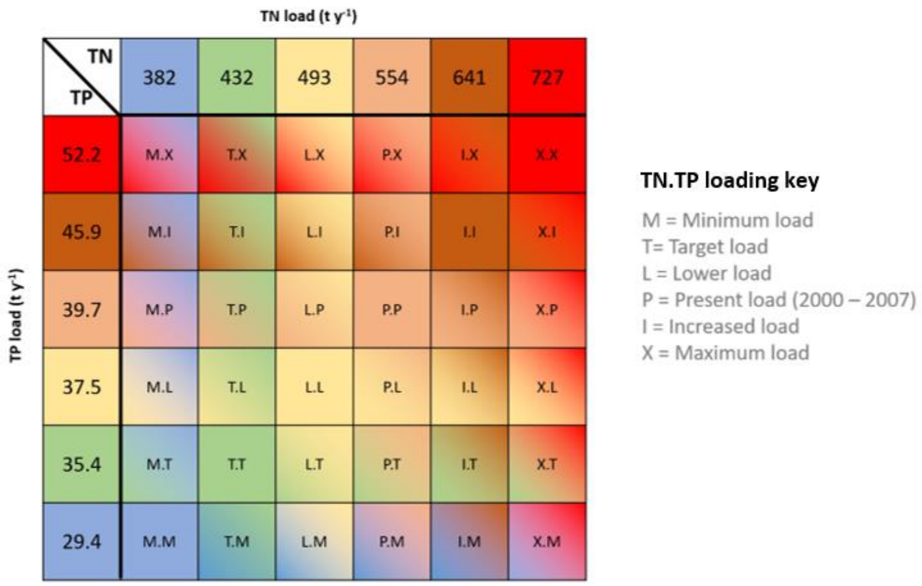


Figure 11. Conceptual diagram showing the matrix of 36 nutrient loading scenarios to Lake Rotorua. Catchment TP load (exclusive of stormflow particulates) is shown on the y-axis and catchment TN load (exclusive of stormflow particulates) is shown on the x-axis. Letter codes inside the matrix correspond to the scenario names in Table 2.

Changes in external nutrient loading are likely to lead to changes in sediment composition, oxygen demand and nutrient release rates, thus leading to a new equilibrium between external and internal loading (Søndergaard et al., 2003). Maximum potential nutrient release rates from the sediment were therefore modified proportionally to changes in external loads (Table 1). In addition, sediment oxygen demand (SOD) was modified based on previously published empirical relationships of hypolimnetic volumetric oxygen demand (HVOD) and nutrient loads (Özkundakci et al., 2012):

For TN < 520 t $yr^{-1}$:

$$HVODs = 0.0046 * \exp(0.0091 * TN\_load) \qquad \text{Equation 7}$$

where HVODs is the HVOD estimated for each scenario.

For TN ≥ 520 t $yr^{-1}$:

$$HVODs = 0.0071 * TN\_load - 3.161 \qquad \text{Equation 8}$$

For each scenario, relative change in HVODs was used to scale the baseline SOD:

$$SODs = SOD - ( (1 - (HVODs/ HVOD) ) * SOD) \qquad \text{Equation 9}$$

where SODs is scenario SOD and HVOD is the HVOD for the baseline (present) scenario,

Scenario SOD was also varied in response to scenario TP load. Because the relationship shown in Equation 7 was unavailable for TP load, we scaled change in TP load to an equivalent change in N load and used Equation 7 to estimate SOD for each TP load scenario. The final SOD for each scenario was then determined as the average of estimated SOD predicted by TN and TP loading under each scenario.

Ammonium and nitrate fluxes were set in direct proportion to change in external TN loading under each scenario, and phosphate flux was similarly scaled by TP load. Estimated nutrient and oxygen fluxes used in each scenario are summarised in Table 1.

### *Simulating the effects of varying internal nutrient loads*

In order to investigate the effect of changes in internal loading of nutrients to the lake water column derived from lake sediments, three additional scenarios were run based on present day external loads (Scenario 22-pp), for the period 2001 to 2017. These scenarios reduced maximum potential release rates of dissolved nutrients to 75%, 50% and 25% of release rates that were calibrated for the period 2001 to 2007 (Table 1). The objective of these scenarios was to provide an indication to what extent internal loading might have changed over the years following the model calibration and validation periods (i.e., post-2007), due to establishment of new equilibrium between modified water column production as a response to external loading, and the associated response of internal recycling (loading) from sediments.

Table 1. Internal loading reduction scenarios, showing catchment nutrient loads and modified sediment nutrient release rates intended to approximate the effects of reducing internal loading while maintaining similar external loading over the period 2001 – 2015.

| | Sediment fluxes | | | | | Catchment nutrient loads | | | |
|---|---|---|---|---|---|---|---|---|---|
| | PO4-P | NH4-N | DOP-P | DON-N | SOD | TN load | TP load | TN load (NSP) | TP load (NSP) |
| | (mg $m^{-2}$ $d^{-1}$) | | | | (g $m^{-2}$ $d^{-1}$) | (t $y^{-1}$) | | (t $y^{-1}$) | |
| S22-PP | 0.070 | 0.400 | 0.00005 | 0.020 | 2.985 | 583 | 44.2 | 554 | 39.7 |
| S0.75 | 0.053 | 0.300 | 0.00004 | 0.015 | 2.250 | 583 | 44.2 | 554 | 39.7 |
| S0.5 | 0.035 | 0.200 | 0.00003 | 0.010 | 1.500 | 583 | 44.2 | 554 | 39.7 |
| S0.25 | 0.018 | 0.100 | 0.00001 | 0.005 | 0.750 | 583 | 44.2 | 554 | 39.7 |

Table 2. Summary of model scenario loads and corresponding modelled sediment nutrient release rates. Scenario Rotorua_scen_22-PP.nc represents current loading for both TN and TP for the scenario period from 2000 to 2007. Loads are reported as both total load, and 'no storm particulate' (NSP) load. NSP load is most analogous to loads used in catchment management and the sustainable load targets (Rutherford et al. 1989). NSP load does not include atmospherically deposited particulate nutrients or suspended sediment inputs from storm flood events. SOD is areal sediment oxygen demand.

| | Sediment fluxes | | | | | Catchment nutrient loads | | | |
|---|---|---|---|---|---|---|---|---|---|
| | PO4-P | NH4-N | DOP | DON | SOD | TN load | TP load | TN load (NSP) | TP load (NSP) |
| | | (mg m$^{-2}$ d$^{-1}$) | | | (g m$^{-3}$ d$^{-1}$) | t y$^{-1}$ | | t y$^{-1}$ | |
| Rotorua_scen_01-MM.nc | 0.053 | 0.282 | 0.00004 | 0.014 | 0.6 | 411 | 33.2 | 382 | 29.4 |
| Rotorua_scen_02-MT.nc | 0.063 | 0.282 | 0.00004 | 0.014 | 1.3 | 411 | 39.6 | 382 | 35.4 |
| Rotorua_scen_03-ML.nc | 0.066 | 0.282 | 0.00005 | 0.014 | 1.6 | 411 | 41.9 | 382 | 37.5 |
| Rotorua_scen_04-MP.nc | 0.070 | 0.282 | 0.00005 | 0.014 | 1.9 | 411 | 44.2 | 382 | 39.7 |
| Rotorua_scen_05-MI.nc | 0.081 | 0.282 | 0.00006 | 0.014 | 2.7 | 411 | 51.0 | 382 | 45.9 |
| Rotorua_scen_06-MX.nc | 0.091 | 0.282 | 0.00007 | 0.014 | 3.5 | 411 | 57.7 | 382 | 52.2 |
| Rotorua_scen_07-TM.nc | 0.053 | 0.316 | 0.00004 | 0.016 | 0.7 | 461 | 33.2 | 432 | 29.4 |
| Rotorua_scen_08-TT.nc | 0.063 | 0.316 | 0.00004 | 0.016 | 1.5 | 461 | 39.6 | 432 | 35.4 |
| Rotorua_scen_09-TL.nc | 0.066 | 0.316 | 0.00005 | 0.016 | 1.8 | 461 | 41.9 | 432 | 37.5 |
| Rotorua_scen_10-TP.nc | 0.070 | 0.316 | 0.00005 | 0.016 | 2.1 | 461 | 44.2 | 432 | 39.7 |
| Rotorua_scen_11-TI.nc | 0.081 | 0.316 | 0.00006 | 0.016 | 2.9 | 461 | 51.0 | 432 | 45.9 |
| Rotorua_scen_12-TX.nc | 0.091 | 0.316 | 0.00007 | 0.016 | 3.7 | 461 | 57.7 | 432 | 52.2 |
| Rotorua_scen_13-LM.nc | 0.053 | 0.358 | 0.00004 | 0.018 | 1.0 | 522 | 33.2 | 493 | 29.4 |
| Rotorua_scen_14-LT.nc | 0.063 | 0.358 | 0.00004 | 0.018 | 1.8 | 522 | 39.6 | 493 | 35.4 |
| Rotorua_scen_15-LL.nc | 0.066 | 0.358 | 0.00005 | 0.018 | 2.1 | 522 | 41.9 | 493 | 37.5 |
| Rotorua_scen_16-LP.nc | 0.070 | 0.358 | 0.00005 | 0.018 | 2.4 | 522 | 44.2 | 493 | 39.7 |
| Rotorua_scen_17-LI.nc | 0.081 | 0.358 | 0.00006 | 0.018 | 3.2 | 522 | 51.0 | 493 | 45.9 |
| Rotorua_scen_18-LX.nc | 0.091 | 0.358 | 0.00007 | 0.018 | 4.0 | 522 | 57.7 | 493 | 52.2 |
| Rotorua_scen_19-PM.nc | 0.053 | 0.400 | 0.00004 | 0.020 | 1.6 | 583 | 33.2 | 554 | 29.4 |
| Rotorua_scen_20-PT.nc | 0.063 | 0.400 | 0.00004 | 0.020 | 2.4 | 583 | 39.6 | 554 | 35.4 |
| Rotorua_scen_21-PL.nc | 0.066 | 0.400 | 0.00005 | 0.020 | 2.7 | 583 | 41.9 | 554 | 37.5 |
| Rotorua_scen_22-PP.nc | 0.070 | 0.400 | 0.00005 | 0.020 | 3.0 | 583 | 44.2 | 554 | 39.7 |
| Rotorua_scen_23-PI.nc | 0.081 | 0.400 | 0.00006 | 0.020 | 3.8 | 583 | 51.0 | 554 | 45.9 |
| Rotorua_scen_24-PX.nc | 0.091 | 0.400 | 0.00007 | 0.020 | 4.6 | 583 | 57.7 | 554 | 52.2 |
| Rotorua_scen_25-IM.nc | 0.053 | 0.459 | 0.00004 | 0.023 | 2.5 | 670 | 33.2 | 641 | 29.4 |
| Rotorua_scen_26-IT.nc | 0.063 | 0.459 | 0.00004 | 0.023 | 3.3 | 670 | 39.6 | 641 | 35.4 |
| Rotorua_scen_27-IL.nc | 0.066 | 0.459 | 0.00005 | 0.023 | 3.6 | 670 | 41.9 | 641 | 37.5 |
| Rotorua_scen_28-IP.nc | 0.070 | 0.459 | 0.00005 | 0.023 | 3.8 | 670 | 44.2 | 641 | 39.7 |
| Rotorua_scen_29-II.nc | 0.081 | 0.459 | 0.00006 | 0.023 | 4.7 | 670 | 51.0 | 641 | 45.9 |
| Rotorua_scen_30-IX.nc | 0.091 | 0.459 | 0.00007 | 0.023 | 5.5 | 670 | 57.7 | 641 | 52.2 |
| Rotorua_scen_31-XM.nc | 0.053 | 0.519 | 0.00004 | 0.026 | 3.3 | 756 | 33.2 | 727 | 29.4 |
| Rotorua_scen_32-XT.nc | 0.063 | 0.519 | 0.00004 | 0.026 | 4.1 | 756 | 39.6 | 727 | 35.4 |
| Rotorua_scen_33-XL.nc | 0.066 | 0.519 | 0.00005 | 0.026 | 4.4 | 756 | 41.9 | 727 | 37.5 |
| Rotorua_scen_34-XP.nc | 0.070 | 0.519 | 0.00005 | 0.026 | 4.7 | 756 | 44.2 | 727 | 39.7 |
| Rotorua_scen_35-XI.nc | 0.081 | 0.519 | 0.00006 | 0.026 | 5.5 | 756 | 51.0 | 727 | 45.9 |
| Rotorua_scen_36-XX.nc | 0.091 | 0.519 | 0.00007 | 0.026 | 6.3 | 756 | 57.7 | 727 | 52.2 |

# Results

## Historical water quality

Figure 11 shows the relationship between annual TN and TP loads from 1974 to 2006 (i.e. pre-dating commencement of alum dosing). Points on the figure have been coloured to denote lake TLI values in that year, and only years with a complete record of all four TLI variables are shown (i.e., years corresponding to solid black dots in Figure 3). The points are widely separated, indicating a relatively weak relationship between TLI and nutrient loading on a year-by-year basis. However, although high TLI values (poor water quality, orange–red dots) were observed across the full range of TP loading, low TLI (green–blue dots) tended to be associated with lower TN loading, perhaps indicative of N-limitation described by nutrient limitation studies during the 1970s and 80s. Prior to alum treatment, the TLI seldom reached the target value of 4.2 (Figure 12). Of note is a reasonably consistent increase in the TN load through the early 2000s while TP load remained steady, and some strongly elevated TP loads in the 1980s and in 1990 and 1991 when treated wastewater was discharged directly to the lake. The complexity of relationship among TLI, loading and time may be due in part to large interannual variation in nutrient loading (largely driven by hydrological variability) and a time-scale of response in lake water quality that is likely longer than the one year periods for which loads are presented here.

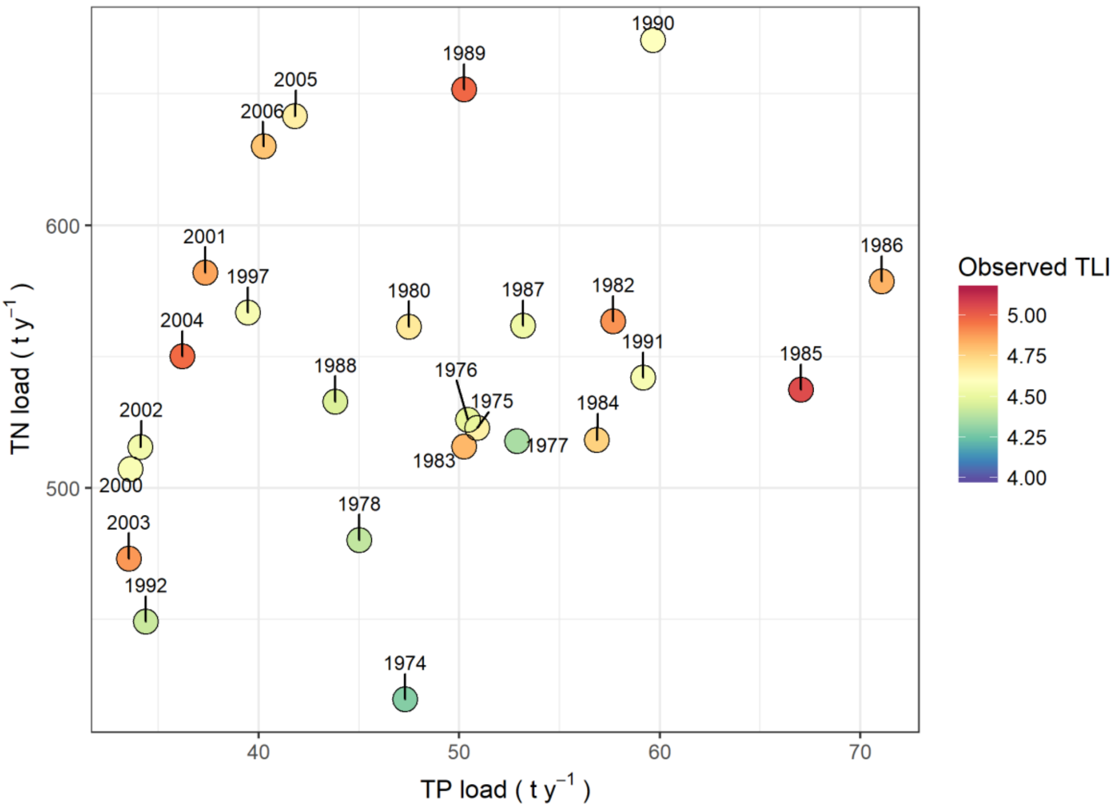


Figure 12. Annual TLI (represented by the colour of each dot) for each year of ‘complete’ monitoring of TLI variables in Lake Rotorua (see McBride et al. 2018 for details), plotted on a grid of observed N and P load for the corresponding year. Note that loads do not include internal loading.

The early 2000s was characterised by relatively high concentrations of DRP including some large peaks in bottom waters (Figure 13). Chlorophyll *a* concentrations were also elevated during this time and the summer-autumn peaks prior to 2007 likely correspond to major blooms of cyanobacteria (blue-green algae). A major transition appears to have occurred around 2007. Concentrations of DRP became close to detection limits in both surface and bottom waters whilst TP and TN concentrations generally declined and became less variable. There appear to be fewer 'spikes' in all nutrient species' concentrations and in chl *a*, and there appears to be less separation of surface and bottom nutrient species' concentrations. Of note, from 2010 onwards, there is a large seasonal variation in nitrate concentration which was less apparent (if present at all) in most other years. This variation was denoted by a progressive increase in nitrate concentrations through autumn, reaching a peak in early winter and then a rapid decline to very low concentrations in mid-summer.

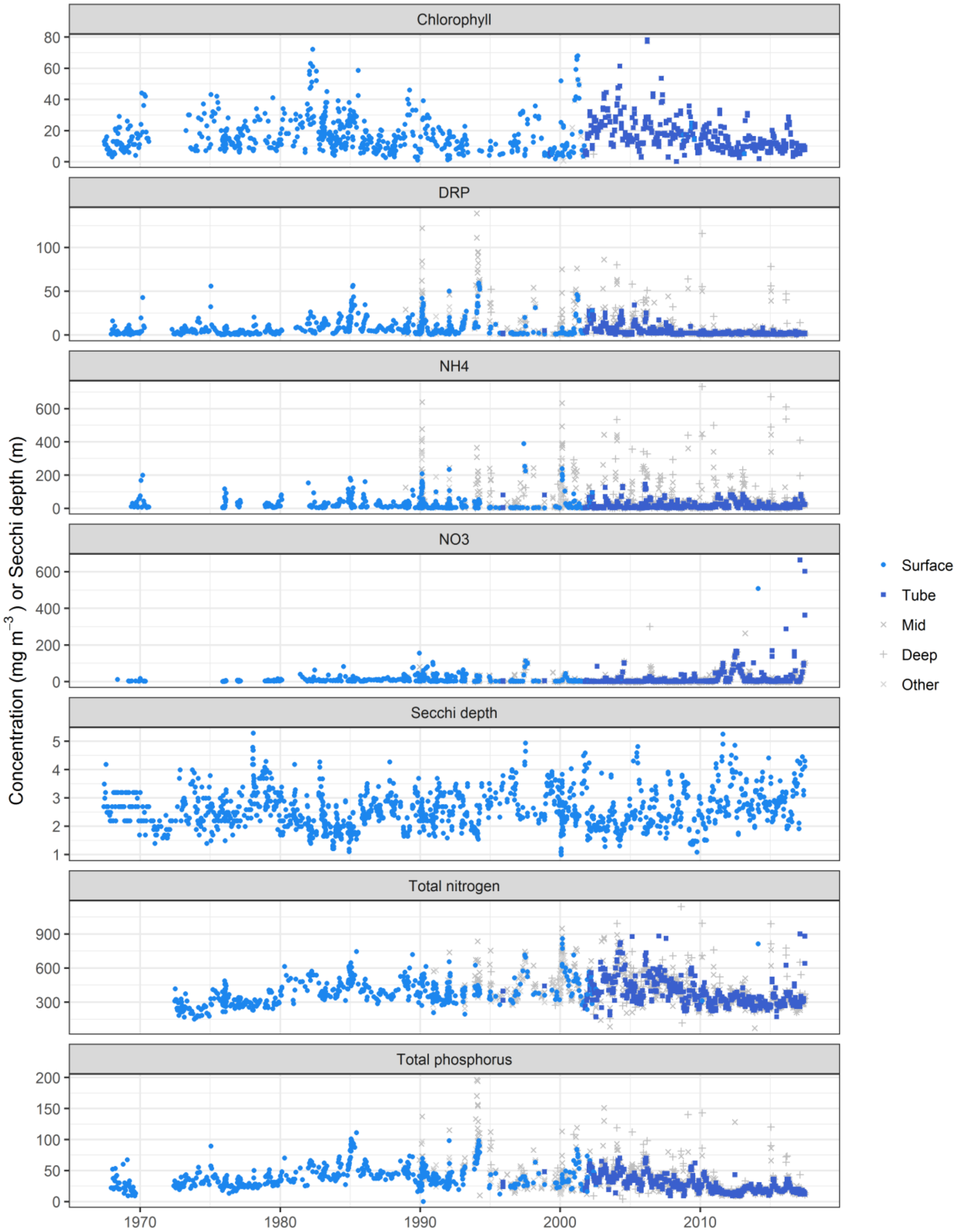


Figure 13. All available in-lake measurements for and chlorophyll *a*, dissolved reactive phosphorus, ammonium, nitrate, Secchi disk depth, total nitrogen, and total phosphorus. Measurements are categorised by sample method and depth of collection. Surface = 0 m, Tube = 0 – 6 or 0 – 10 m, Mid = 14 – 16 m, deep = > 18 m, Other = any other depth.

### Model performance - calibration and validation

Model calibration was undertaken from July 2000 to June 2004 and validation from June 2004 to June 2007, as shown for comparisons of measured and simulated data for plots of surface water quality variables in Figure 14 -16. The calibration of DYRESM-CAEDYM for Lake Rotorua has been progressively refined since the first published study of the coupled model application to Lake Rotorua (Burger et al. 2008). The current statistical performance of the model (Table 3) may be considered satisfactory compared with the earlier applications and to similar studies undertaken elsewhere (e.g., Arhonditsis et al., 2006). The target for the calibration was to obtain values within 0.1 of observed TLI. We obtained a satisfactory outcome with only small disparities between observed TLI and simulated values (Figure 18). Of note are the high TLI values of 2003 and 2004. These appear as outliers in observations and also correspond to a period when there were major blooms of the cyanobacterium *Dolichospermum planktonicum.*

Lake water temperature and dissolved oxygen were simulated with the highest precision among all variables (Figure 14). The model reproduced well the polymictic nature of thermal stratification, whereby during summer the lake could stratify for short periods (up to and sometimes more than one week). During summer the model reproduced the de-oxygenation of hypolimnion (bottom waters), and simulated a regular pattern of increased dissolved and total nutrient concentrations as shown by observations (Figure 15 & Figure 16), related to enhanced phosphate release (likely due to reductive dissolution from iron and manganese complexes) and ammonium release (likely due to ammonium buildup caused by the absence of nitrification under anoxic conditions, and decreased ammonium assimilation by aerobic microorganisms).

Related to this stratification and nutrient release, enhanced phytoplankton growth occurred during summer periods, which was generally reproduced by the model. However, the large magnitude cyanobacteria algal blooms in 2001 and 2006 were not simulated to the same magnitude as field data. We note that phytoplankton blooms are highly spatially heterogeneous, and some of this error may be in part due to the differing spatial scales of the model (horizontally averaged) and field samples (depth integrated point locations).

Model simulations indicated that cyanobacteria were dominant during summer periods, freshwater diatoms dominant during winter periods, and chloropytes co-dominant with cyanobacteria during spring (Figure 17). Model simulations reproduced TLI levels with acceptable accuracy, but did not reproduce the the same magnitude of yearly fluctuations. This is in part due to underestimation of large bloom events. Model-simulated TLI (Figure 18) in 2003/2004 is lower than the observed value, partially due to the fact that chl *a* concentration is underestimated.

Table 3. Statistics for model performance against field observations in three depth zones of Lake Rotorua for the calibration (2000-2004) and validation (2004-2007) periods. R: Pearson correlation coefficient (R). RMSE: root mean square error.

| | Calibration (2000-2004) | | | | | | Validation (2004-2007) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *Surface (0-6 m)* | | *15 m* | | *Bottom (18+ m)* | | *Surface (0-6 m)* | | *15 m* | | *Bottom (18+ m)* | |
| | **R** | **RMSE** | **R** | **RMSE** | **R** | **RMSE** | **R** | **RMSE** | **R** | **RMSE** | **R** | **RMSE** |
| **Temperature** | 0.991 | 0.612 | 0.994 | 0.556 | 0.993 | 0.585 | 0.997 | 0.451 | 0.997 | 0.471 | 0.997 | 0.470 |
| **Dissolved oxygen** | 0.811 | 1.596 | 0.700 | 2.026 | 0.731 | 1.947 | 0.706 | 1.915 | 0.827 | 2.098 | 0.787 | 2.088 |
| **Phosphate** | -0.151 | 0.013 | 0.008 | 0.019 | 0.292 | 0.018 | -0.013 | 0.009 | 0.180 | 0.011 | 0.265 | 0.016 |
| **Ammonium** | 0.010 | 0.036 | -0.037 | 0.109 | 0.291 | 0.132 | 0.272 | 0.037 | 0.167 | 0.217 | 0.303 | 0.120 |
| **Nitrate** | 0.218 | 0.008 | 0.160 | 0.010 | 0.642 | 0.016 | 0.246 | 0.019 | 0.007 | 0.021 | 0.011 | 0.021 |
| **Total phosphorus** | 0.604 | 0.014 | 0.531 | 0.023 | 0.195 | 0.019 | 0.456 | 0.015 | 0.458 | 0.012 | 0.276 | 0.017 |
| **Total nitrogen** | 0.373 | 0.128 | 0.464 | 0.123 | 0.504 | 0.174 | 0.160 | 0.133 | 0.152 | 0.124 | 0.048 | 0.136 |
| **Chlorophyll *a*** | 0.085 | 15.498 | | | | | 0.267 | 14.989 | | | | |

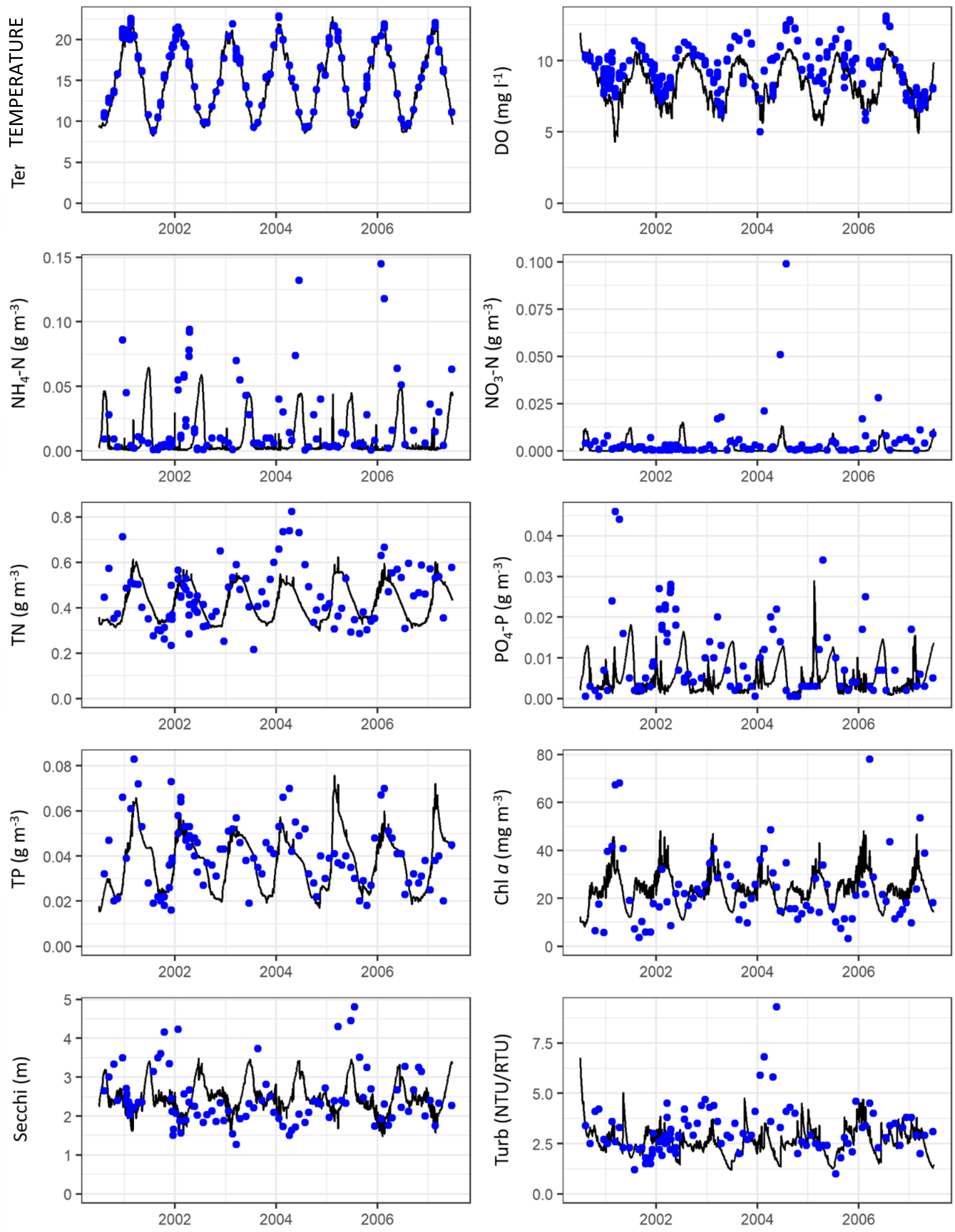


Figure 14. Comparison of observed values (blue dots) and model simulations from the calibrated baseline model (scenario 22-PP; black line) for (clockwise from top left) temperature (°C), dissolved oxygen (mg $L^{-1}$), nitrate-N (g $m^{-3}$), phosphate (PO4-P, in g $m^{-3}$), total chlorophyll *a* (mg $m^{-3}$), turbidity (NTU for observations and relative turbidity units (RTU) derived from suspended solids concentration for simulations), Secchi disk depth (m), total phosphorus and total nitrogen (g $m^{-3}$) in surface waters (average of 0 to 6 m depth).

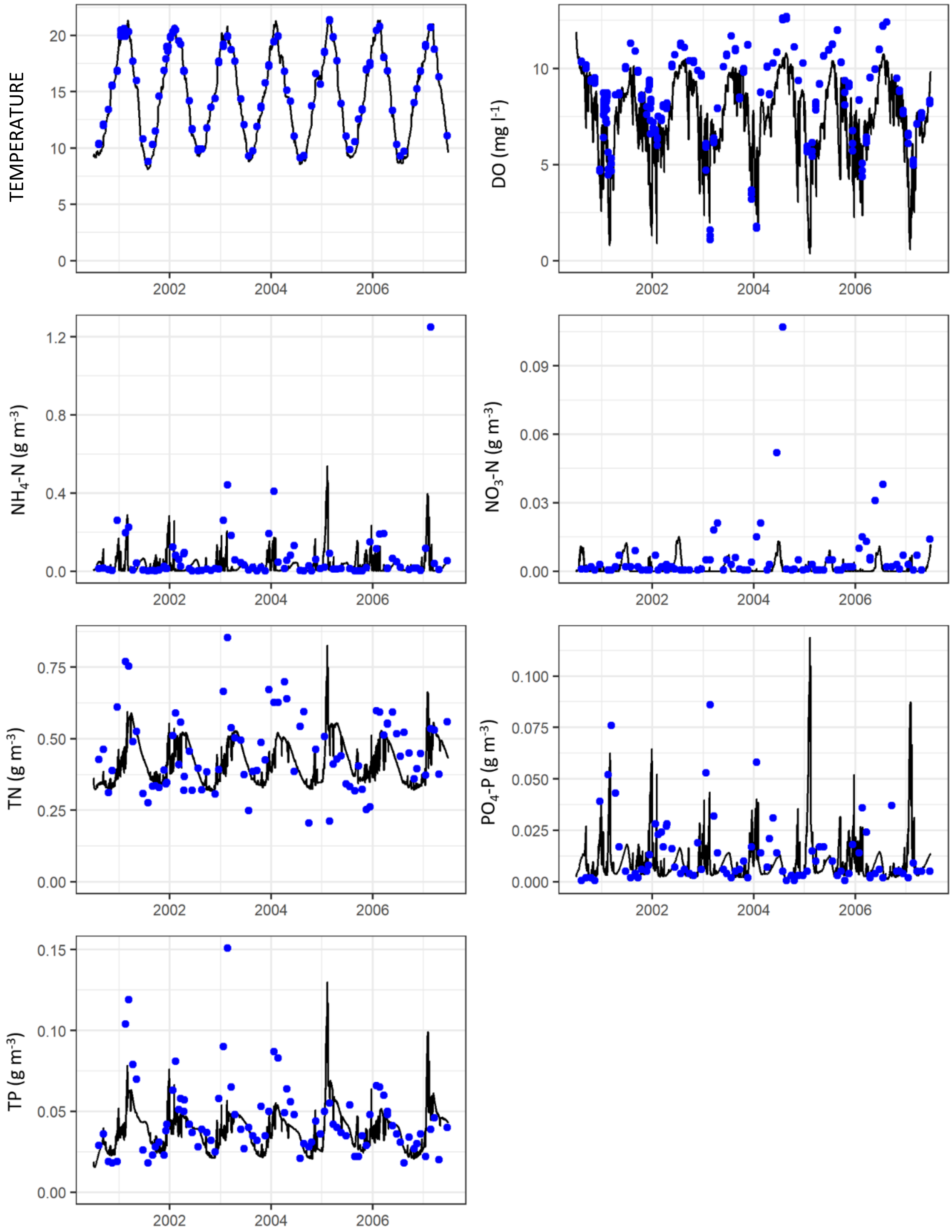


Figure 15. Comparison of observed values (blue dots) and model simulations from the calibrated baseline model (scenario 22-PP; black line) for (clockwise from top left) temperature (ºC), dissolved oxygen (mg $L^{-1}$), nitrate-N (g $m^{-3}$), phosphate (PO4-P, in g $m^{-3}$), Secchi disk depth (m), total phosphorus and total nitrogen (g $m^{-3}$) in mid-water column (of 15 m depth).

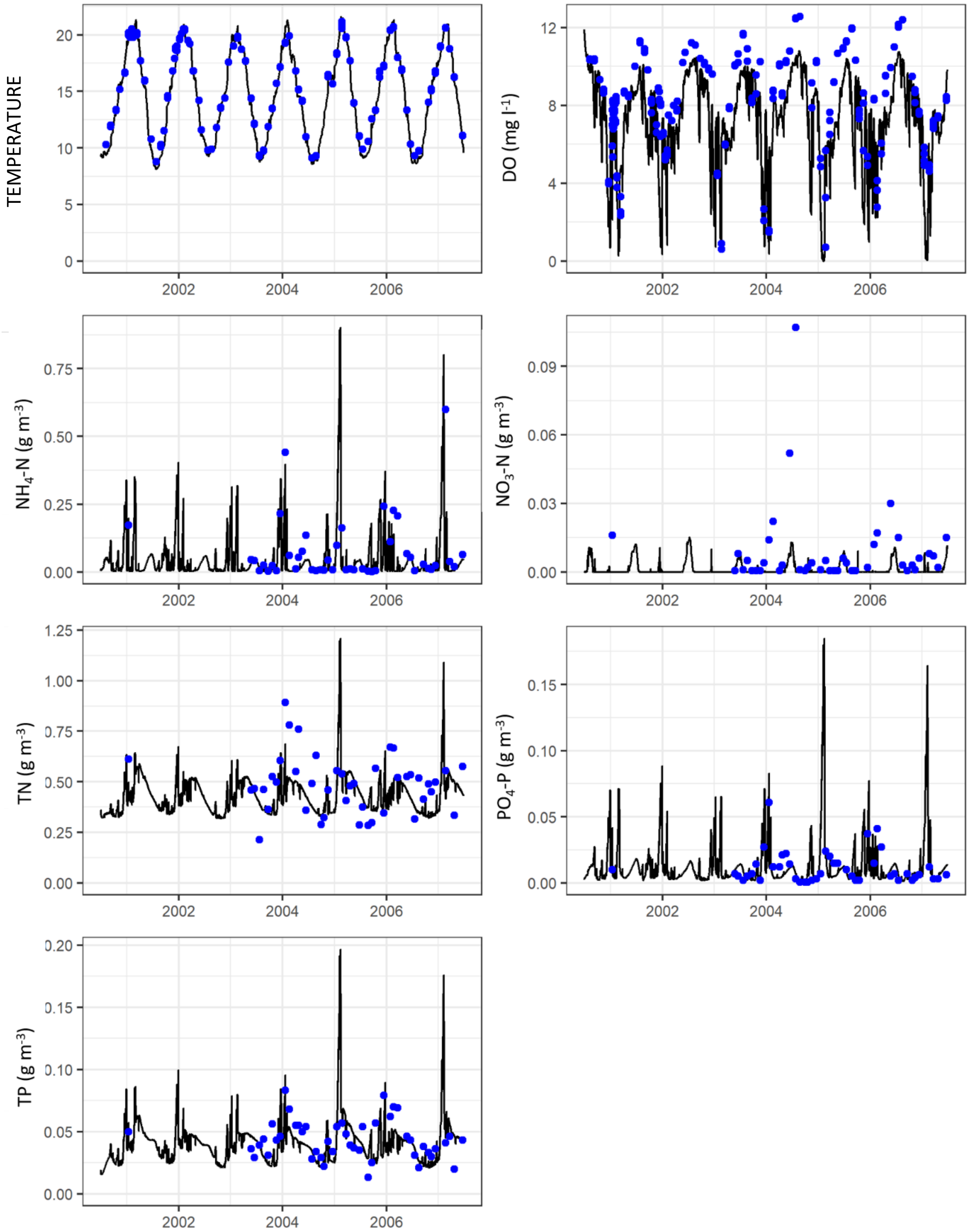


Figure 16. Comparison of observed values (blue dots) and model simulations from the calibrated baseline model (scenario 22-PP; black line) for (clockwise from top left) temperature (°C), dissolved oxygen (mg $L^{-1}$), nitrate-N (g $m^{-3}$), phosphate (PO4-P, in g $m^{-3}$), Secchi disk depth (m), total phosphorus and total nitrogen (g $m^{-3}$) in deep waters (greater than 18 m depth).

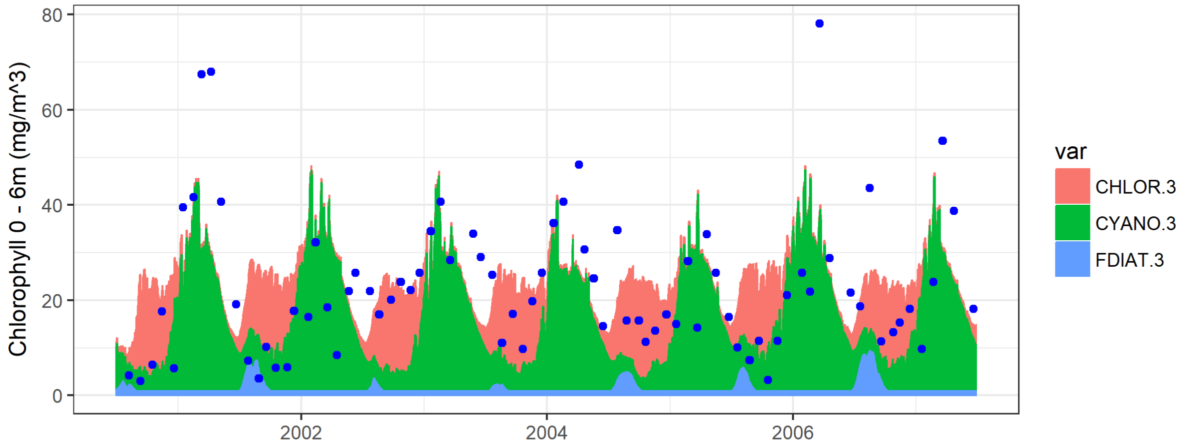


Figure 17. Observed total chlorophyll a concentration (blue dots), overlaid on a stacked area plot of simulated chlorophyll a concentrations for three functional phytoplankton groups: chlorophytes (pink), cyanobacteria (green) and diatoms (blue). All values are means for the upper water column (0-6 m), in mg $m^{-3}$.

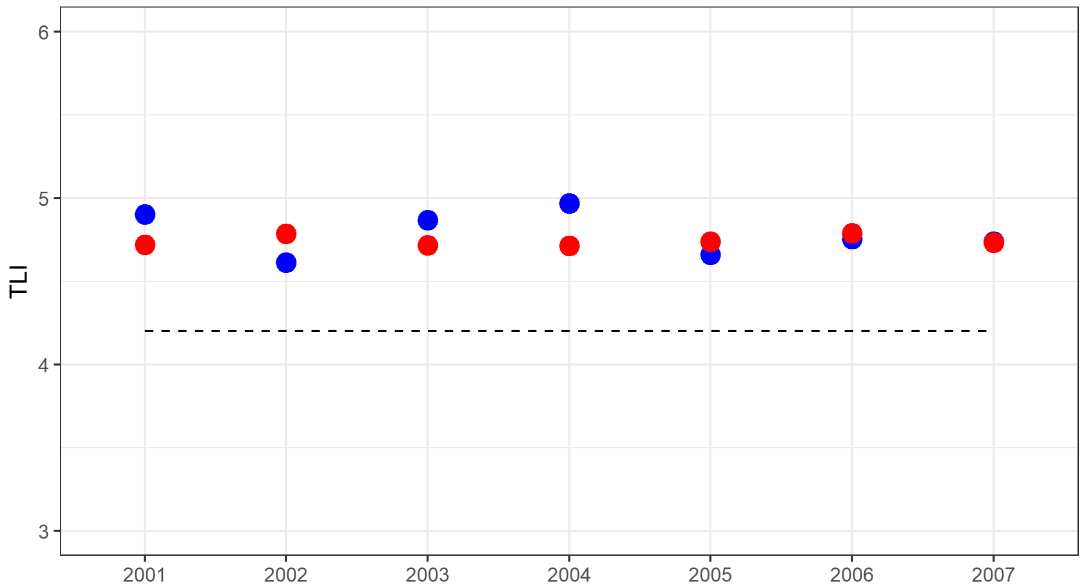


Figure 18. Comparison of simulated lake TLI (red) and observed TLI (blue) for Lake Rotorua, for the calibration (July 2000 to June 2004) and validation (July 2004 to June 2007) periods. The dashed black line is the TLI target for Lake Rotorua (TLI = 4.2).

## Global sensitivity analysis

The sensitivity analysis presented in Table 4 represents a simple, global approach to parameter sensitivity whereby multiple model parameters are modified for each model simulation (n simulations = 100) using common bounds of ±25% of the calibrated value. Parameters relating to growth and respiration for cyanobacteria were the most sensitive in determining chl *a,* Secchi depth, and overall.

Parameters determining the maximum potential, and half saturation of release rates for phosphate and ammonium were strongly sensitive in relation to TP and TN concentrations, respectively. All variables assessed were strongly sensitive to sediment oxygen demand (SOD), likely due to its influence on hypolimnion deoxygenation and subsequent internal loading. This reinforces the importance of accurately characterizing stratification and internal loading dynamics in polymictic and productive lakes.

Parameters relating to the density and diameter of suspended material (POM and suspended sediment) were also among the most sensitive, probably due to their influence on light attenuation and nutrient recycling. Those parameters exhibiting sensitivity with Secchi depth were generally similar to those determining chl *a*. This is due to the influence of phytoplankton concentration on light attenuation.

Table 4. Output of preliminary global parameter sensitivity analysis for DYRESM-CAEDYM model of Lake Rotorua 2000 – 2007 (100 iterations). Average overall score is the mean of $R^2$ for chl *a*, Secchi depth, Total P and Total N. A colour scale has been applied to the $R^2$ values with red representing the highest R2 and green repressing the lowest $R^2$.

| Parameter | | Average | chl a | | Secchi depth | | Total P | | Total N | |
|---|---|---|---|---|---|---|---|---|---|---|
| Abbreviation | Full name | Overall score | r | $R^2$ | r | $R^2$ | r | $R^2$ | r | $R^2$ |
| kr(CYANO) | Respiration rate coefficient | 0.091 | -0.385 | 0.148 | 0.350 | 0.122 | -0.088 | 0.008 | -0.292 | 0.085 |
| Pmax(CYANO) | Maximum growth rate | 0.073 | 0.359 | 0.129 | -0.313 | 0.098 | -0.087 | 0.008 | 0.237 | 0.056 |
| SmpPO4 | Release rate of PO4 | 0.061 | -0.008 | 0.000 | -0.035 | 0.001 | 0.489 | 0.239 | 0.069 | 0.005 |
| POMDensity1 | Density of POM particles (labile) | 0.059 | -0.022 | 0.000 | 0.006 | 0.000 | -0.259 | 0.067 | -0.409 | 0.167 |
| kr(CHLOR) | Respiration rate coefficient | 0.044 | -0.219 | 0.048 | 0.183 | 0.033 | 0.291 | 0.085 | -0.108 | 0.012 |
| KOxS-PO4 | Half sat constant for PO4 sediment flux | 0.042 | 0.018 | 0.000 | 0.009 | 0.000 | 0.400 | 0.160 | 0.082 | 0.007 |
| Smp-NH4 | Release rate of NH4 | 0.036 | 0.018 | 0.000 | -0.022 | 0.000 | -0.027 | 0.001 | 0.379 | 0.143 |
| rSOs | Static sediment exchange rate | 0.036 | 0.177 | 0.031 | -0.245 | 0.060 | 0.085 | 0.007 | 0.211 | 0.044 |
| PON1max | Max transfer of PONL-DONL | 0.036 | 0.175 | 0.031 | -0.242 | 0.059 | -0.121 | 0.015 | 0.198 | 0.039 |
| deSS(SSOL1) | Density of suspended solid particles | 0.033 | 0.229 | 0.052 | -0.233 | 0.054 | -0.011 | 0.000 | 0.164 | 0.027 |
| Pmax(CHLOR) | Maximum growth rate | 0.033 | 0.048 | 0.002 | -0.137 | 0.019 | -0.321 | 0.103 | -0.098 | 0.010 |
| KN(FDIAT) | Half saturation constant for nitrogen | 0.026 | 0.170 | 0.029 | -0.122 | 0.015 | -0.020 | 0.000 | 0.248 | 0.062 |
| DOD1max | Max mineralisaton of DOPL-PO4 | 0.026 | -0.145 | 0.021 | 0.235 | 0.055 | 0.046 | 0.002 | -0.164 | 0.027 |
| ws(CYANO) | Constant settling velocity | 0.026 | 0.170 | 0.029 | -0.132 | 0.018 | -0.219 | 0.048 | 0.091 | 0.008 |
| UPmax(CYANO) | Maximum rate of phytoplankton phosphorus uptake | 0.025 | -0.107 | 0.011 | 0.103 | 0.011 | -0.251 | 0.063 | -0.128 | 0.016 |
| KN(CYANO) | Half saturation constant for nitrogen | 0.024 | 0.135 | 0.018 | -0.179 | 0.032 | -0.070 | 0.005 | 0.207 | 0.043 |
| Tmax(CYANO) | Maximum temperature | 0.023 | 0.176 | 0.031 | -0.155 | 0.024 | -0.153 | 0.024 | 0.123 | 0.015 |
| fres(FDIAT) | Fraction of respiration relative to total metabolic loss rate | 0.023 | 0.174 | 0.030 | -0.213 | 0.045 | -0.060 | 0.004 | 0.120 | 0.014 |
| KeSS(SSOL1) | Specific attenuation coefficient of suspended solids | 0.023 | 0.168 | 0.028 | -0.180 | 0.033 | -0.162 | 0.026 | 0.057 | 0.003 |
| Kep(CHLOR) | Specific attenuation coefficient of phytoplankton | 0.021 | -0.121 | 0.015 | 0.185 | 0.034 | 0.056 | 0.003 | -0.183 | 0.034 |
| Topt(CYANO) | Optimum temperature | 0.019 | 0.030 | 0.001 | -0.031 | 0.001 | 0.183 | 0.033 | 0.203 | 0.041 |
| fres(CHLOR) | Fraction of respiration relative to total metabolic loss rate | 0.018 | 0.118 | 0.014 | -0.224 | 0.050 | -0.074 | 0.005 | 0.053 | 0.003 |
| UPmax(CHLOR) | Maximum rate of phytoplankton phosphorus uptake | 0.016 | 0.123 | 0.015 | -0.185 | 0.034 | -0.103 | 0.011 | 0.071 | 0.005 |
| POC1max | Max transfer of POCL-DOCL | 0.016 | -0.111 | 0.012 | 0.138 | 0.019 | 0.142 | 0.020 | -0.104 | 0.011 |
| UNmax(CHLOR) | Maximum rate of phytoplankton nitrogen uptake | 0.015 | 0.112 | 0.013 | -0.152 | 0.023 | -0.049 | 0.002 | 0.152 | 0.023 |
| vR(FDIAT) | Temperature multiplier for phytoplankton respiration | 0.015 | -0.083 | 0.007 | 0.015 | 0.000 | -0.217 | 0.047 | -0.081 | 0.007 |
| tcSS(SSOL2) | Critical shear stress of suspended solids | 0.014 | -0.109 | 0.012 | 0.178 | 0.032 | 0.088 | 0.008 | -0.082 | 0.007 |
| Tsta(CHLOR) | Standard temperature | 0.013 | -0.154 | 0.024 | 0.079 | 0.006 | -0.116 | 0.014 | -0.090 | 0.008 |
| resusRate | Composite resuspension rate | 0.011 | -0.074 | 0.005 | 0.060 | 0.004 | 0.183 | 0.033 | -0.040 | 0.002 |
| sedOrganicFrac | Fraction of sediment that is organics | 0.011 | 0.085 | 0.007 | -0.130 | 0.017 | -0.102 | 0.010 | 0.091 | 0.008 |
| Tsta(CYANO) | Standard temperature | 0.009 | -0.068 | 0.005 | 0.046 | 0.002 | -0.154 | 0.024 | -0.086 | 0.007 |
| KDOS(NH4) | Half sat constant for NH4 sediment flux | 0.009 | 0.008 | 0.000 | -0.003 | 0.000 | -0.128 | 0.016 | 0.145 | 0.021 |
| Topt(CHLOR) | Optimum temperature | 0.009 | 0.038 | 0.001 | -0.001 | 0.000 | 0.066 | 0.004 | 0.175 | 0.031 |
| DON1max | Max mineralisation of DONL-NH4 | 0.009 | 0.049 | 0.002 | -0.007 | 0.000 | -0.028 | 0.001 | -0.181 | 0.033 |
| KDOS(NO3) | Half sat constant for NO3 sediment flux | 0.009 | -0.131 | 0.017 | 0.097 | 0.009 | 0.054 | 0.003 | 0.074 | 0.005 |
| deSS(SSOL2) | Density of suspended solid particles | 0.009 | 0.034 | 0.001 | 0.126 | 0.016 | -0.123 | 0.015 | 0.046 | 0.002 |
| KoN2 | Denitrification rate coefficient | 0.008 | 0.054 | 0.003 | -0.126 | 0.016 | -0.098 | 0.010 | 0.074 | 0.006 |
| KN(CHLOR) | Half saturation constant for nitrogen | 0.008 | 0.040 | 0.002 | -0.071 | 0.005 | -0.148 | 0.022 | -0.071 | 0.005 |
| Tmax(CHLOR) | Maximum temperature | 0.008 | 0.084 | 0.007 | -0.093 | 0.009 | -0.131 | 0.017 | -0.021 | 0.000 |
| kr(FDIAT) | Respiration rate coefficient | 0.008 | 0.118 | 0.014 | -0.085 | 0.007 | -0.049 | 0.002 | 0.093 | 0.009 |
| KP(FDIAT) | Half saturation constant for phosphorus | 0.008 | -0.087 | 0.008 | 0.102 | 0.010 | -0.101 | 0.010 | -0.053 | 0.003 |
| ws(FDIAT) | Constant settling velocity | 0.008 | -0.014 | 0.000 | -0.067 | 0.005 | -0.082 | 0.007 | -0.139 | 0.019 |
| diaSS(SSOL2) | Diameter of suspended solid particles | 0.008 | 0.099 | 0.010 | 0.108 | 0.012 | -0.091 | 0.008 | 0.019 | 0.000 |
| KP(CYANO) | Half saturation constant for phosphorus | 0.007 | -0.096 | 0.009 | 0.129 | 0.017 | 0.020 | 0.000 | -0.056 | 0.003 |
| Tmax(FDIAT) | Maximum temperature | 0.007 | -0.028 | 0.001 | -0.016 | 0.000 | -0.166 | 0.028 | 0.001 | 0.000 |
| Smp-NO3 | Release rate of NO3 | 0.007 | -0.043 | 0.002 | 0.048 | 0.002 | -0.132 | 0.018 | -0.075 | 0.006 |
| Kep(FDIAT) | Specific attenuation coefficient of phytoplankton | 0.006 | 0.058 | 0.003 | 0.007 | 0.000 | 0.146 | 0.021 | 0.020 | 0.000 |
| IK(FDIAT) | Parameter for initial slope of P/I curve | 0.006 | 0.108 | 0.012 | -0.079 | 0.006 | 0.051 | 0.003 | 0.068 | 0.005 |
| UPmax(FDIAT) | Maximum rate of phytoplankton phosphorus uptake | 0.006 | 0.051 | 0.003 | 0.051 | 0.003 | 0.123 | 0.015 | 0.061 | 0.004 |
| POMDia1 | Diameter of POM particles (labile) | 0.006 | 0.053 | 0.003 | -0.033 | 0.001 | 0.009 | 0.000 | -0.140 | 0.020 |
| KOn | Half sat constant for nitrification | 0.006 | -0.027 | 0.001 | 0.063 | 0.004 | -0.131 | 0.017 | 0.040 | 0.002 |
| koNH | Nitrification rate coefficient | 0.006 | 0.028 | 0.001 | -0.090 | 0.008 | -0.115 | 0.013 | 0.027 | 0.001 |
| Topt(FDIAT) | Optimum temperature | 0.005 | 0.105 | 0.011 | -0.085 | 0.007 | -0.050 | 0.002 | 0.019 | 0.000 |
| tcSS(SSOL1) | Critical shear stress of suspended solids | 0.005 | 0.076 | 0.006 | -0.038 | 0.001 | 0.028 | 0.001 | 0.111 | 0.012 |
| KeDOC1 | Specific attenuation coefficient of DOC (labile) | 0.005 | -0.036 | 0.001 | 0.004 | 0.000 | -0.028 | 0.001 | -0.133 | 0.018 |
| tcPOM1 | Critical shear stress for POM (labile) | 0.005 | 0.028 | 0.001 | -0.065 | 0.004 | 0.117 | 0.014 | 0.032 | 0.001 |
| KP(CHLOR) | Half saturation constant for phosphorus | 0.005 | 0.049 | 0.002 | -0.034 | 0.001 | -0.075 | 0.006 | 0.103 | 0.011 |
| ws(CHLOR) | Constant settling velocity | 0.005 | 0.048 | 0.002 | -0.064 | 0.004 | -0.110 | 0.012 | 0.022 | 0.000 |
| fdom(FDIAT) | Fraction of metabolic loss rate that goes to DOM | 0.005 | -0.077 | 0.006 | 0.059 | 0.003 | 0.088 | 0.008 | -0.032 | 0.001 |
| fdom(CHLOR) | Fraction of metabolic loss rate that goes to DOM | 0.004 | 0.002 | 0.000 | -0.015 | 0.000 | 0.083 | 0.007 | 0.102 | 0.010 |
| fres(CYANO) | Fraction of respiration relative to total metabolic loss rate | 0.004 | -0.083 | 0.007 | 0.006 | 0.000 | 0.092 | 0.008 | 0.018 | 0.000 |
| UNmax(CYANO) | Maximum rate of phytoplankton nitrogen uptake | 0.003 | 0.033 | 0.001 | -0.039 | 0.001 | -0.103 | 0.011 | 0.023 | 0.001 |
| diaSS(SSOL1) | Diameter of suspended solid particles | 0.003 | 0.011 | 0.000 | 0.046 | 0.002 | 0.096 | 0.009 | 0.043 | 0.002 |
| IK(CHLOR) | Parameter for initial slope of P/I curve | 0.003 | -0.012 | 0.000 | -0.071 | 0.005 | 0.040 | 0.002 | 0.061 | 0.004 |
| Pmax(FDIAT) | Maximum growth rate | 0.003 | 0.046 | 0.002 | -0.048 | 0.002 | -0.061 | 0.004 | 0.043 | 0.002 |
| KSOs | Half sat constant for DO sediment flux | 0.002 | -0.023 | 0.001 | 0.030 | 0.001 | 0.004 | 0.000 | -0.092 | 0.009 |
| Kep(CYANO) | Specific attenuation coefficient of phytoplankton | 0.002 | -0.064 | 0.004 | -0.038 | 0.001 | 0.055 | 0.003 | 0.032 | 0.001 |
| POP1max | Max transfer of POPL-DOPL | 0.002 | -0.033 | 0.001 | 0.012 | 0.000 | -0.016 | 0.000 | 0.082 | 0.007 |
| Tsta(FDIAT) | Standard temperature | 0.002 | -0.073 | 0.005 | 0.022 | 0.000 | -0.047 | 0.002 | 0.012 | 0.000 |
| IK(CYANO) | Parameter for initial slope of P/I curve | 0.002 | -0.062 | 0.004 | 0.027 | 0.001 | 0.040 | 0.002 | 0.010 | 0.000 |
| SedPorosity | Sediment porosity | 0.001 | -0.032 | 0.001 | -0.016 | 0.000 | -0.059 | 0.003 | 0.016 | 0.000 |
| fdom(CYANO) | Fraction of metabolic loss rate that goes to DOM | 0.001 | 0.043 | 0.002 | -0.039 | 0.002 | -0.036 | 0.001 | 0.015 | 0.000 |
| KN2 | Half sat constant for denitrification | 0.001 | -0.045 | 0.002 | 0.040 | 0.002 | 0.014 | 0.000 | -0.031 | 0.001 |
| UNmax(FDIAT) | Maximum rate of phytoplankton nitrogen uptake | 0.001 | -0.022 | 0.000 | -0.002 | 0.000 | -0.019 | 0.000 | -0.062 | 0.004 |
| KeSS(SSOL2) | Specific attenuation coefficient of suspended solids | 0.001 | -0.008 | 0.000 | 0.026 | 0.001 | -0.011 | 0.000 | -0.052 | 0.003 |

### Model scenario results - external load change

External loading scenarios showed considerable potential for the improvement of water quality by reducing catchment loads. The model predicted that the trophic state of the lake could be reduced from a TLI prior to substantial alum dosing (4.8 in 2008) to 4.29 (Figure 19).

The lowest load scenarios (i.e., loads less than sustainable targets) had the greatest impact on the TN and TP TLI components, reducing them to below TLI 4 for TN and Secchi disk, however TLc (chl *a*) was only reduced to a minimum of 5.32 from the current simulated condition of 5.65 (S22-PP). Similarly, TLs, derived from Secchi disk depth, was relatively insensitive to changes in external loading, likely due to the fact that TSS concentrations were not modified under scenarios, so the main factor influencing TLs would be chl *a* concentrations.

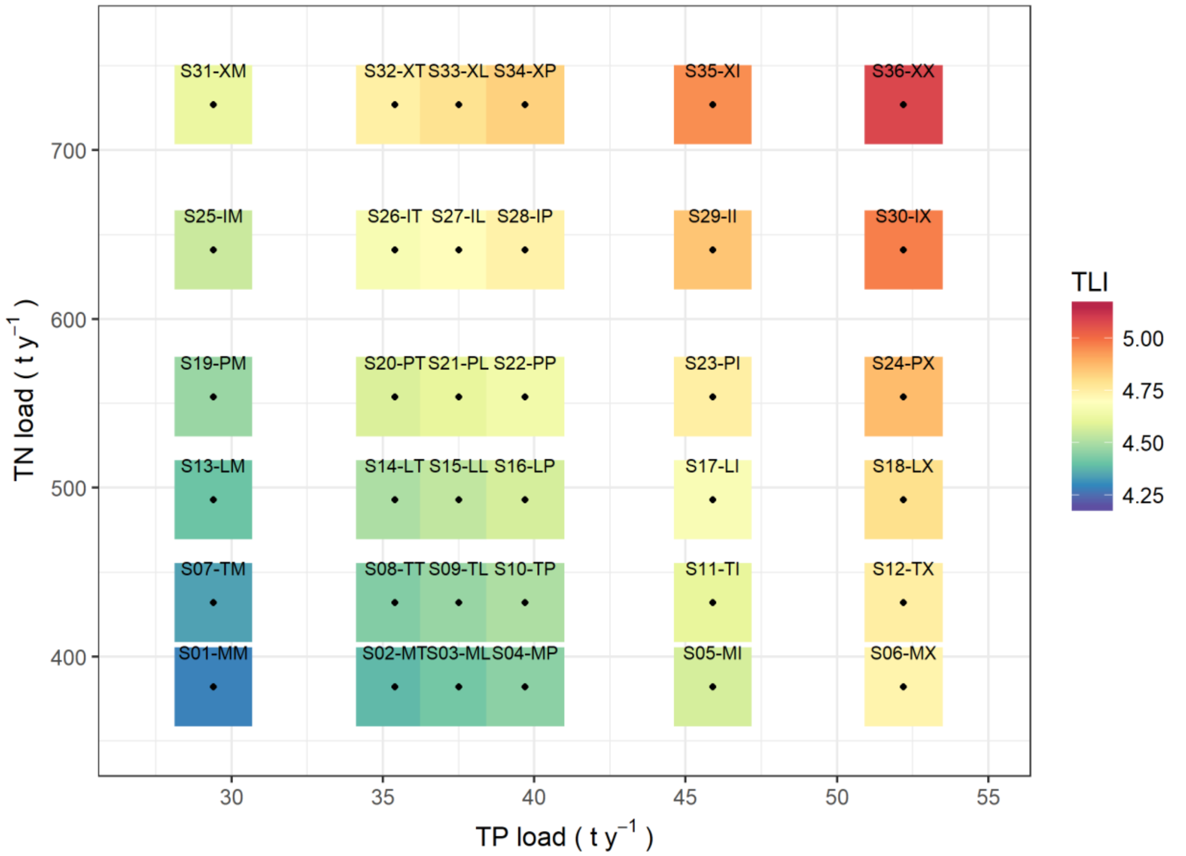


Figure 19. Modelled TLI under 36 scenarios covering various combinations of catchment TN and TP loading (NSP loads). Results presented demonstrate the influence of TP and TN loading (t y$^{-1}$) on TLI over the scenario period from July 2000 to June 2007. The lowest simulated TLI was 4.29 (S01-MM) and the highest simulated TLI was 5.08 (S36-XX). Each point and coloured tile on the plot represents one nutrient loading scenario corresponding to its position on the X and Y axes and the colour (not the size) of the tile shows the resulting simulated TLI. The last two letters of each scenario name represent the N and P load, respectively, where: M = minimum load, and T=Target load, L = lower load, P = Present load (2001–2007), I = Increased load, and X = Maximum load.

It should be noted that TLI values are on a log scale, and relatively small changes in TLI generally represent more substantial changes in the constituent variables. Although the range of TLI values simulated among all scenarios was small, approximately 1 TLI unit between minimum and maximum loading, these changes represent large changes in lake nutrient concentrations (Figure 20). Simulated mean TN concentrations for all scenarios ranged from 0.28 to 0.63 g $m^{-3}$, TP 0.022 to 0.079 g $m^{-3}$, total chlorophyll *a* 17.1 to 26.7 mg $m^{-3}$ and Secchi 2.5 to 3.2 m.

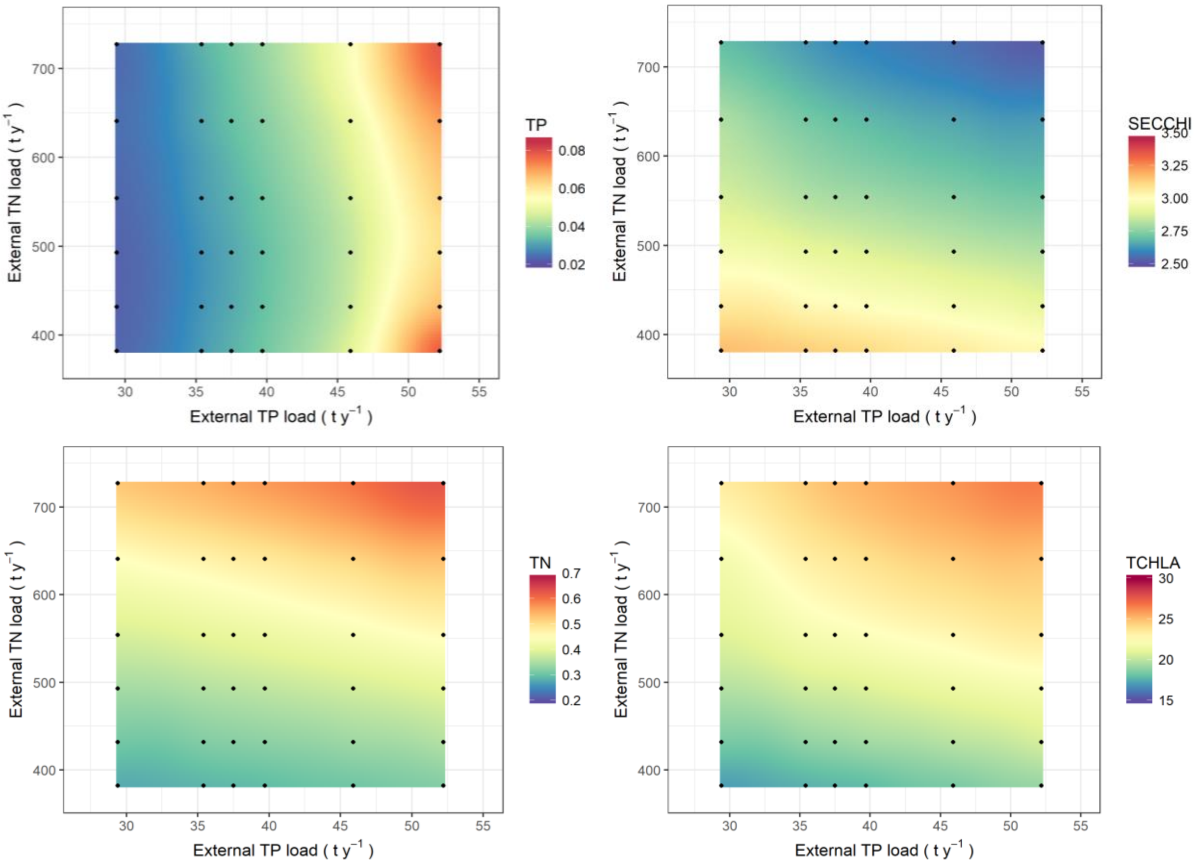


Figure 20. Simulated mean TP, TN, chlorophyll *a* and Secchi depth, plotted as a function of external N and P load (NSP loads) for each of 36 loading scenarios for the period from July 2000 to June 2007.

Internal loading from lake sediments was projected to change by greater relative change than external loading, likely as a result of the non-linear relationships used to derive likely sediment oxygen demand under the various model scenarios (see Methods for calculation of SOD). Figure 21 shows simulated lake TLI (on the colour scale) resulting from all 36 scenarios of N and P loading combinations, for external, internal, and total (external + internal) load. These results show the outcomes of the specified dynamics, based on the stated assumptions coupling internal loads to external loading, and suggest that internal load dynamics represent an important component of the overall lake response to external loading. A detailed comparison of all components of 'total load' (including dissolved nutrients) to the lake for N and P under all 36 model scenarios is provided in Appendix 2.

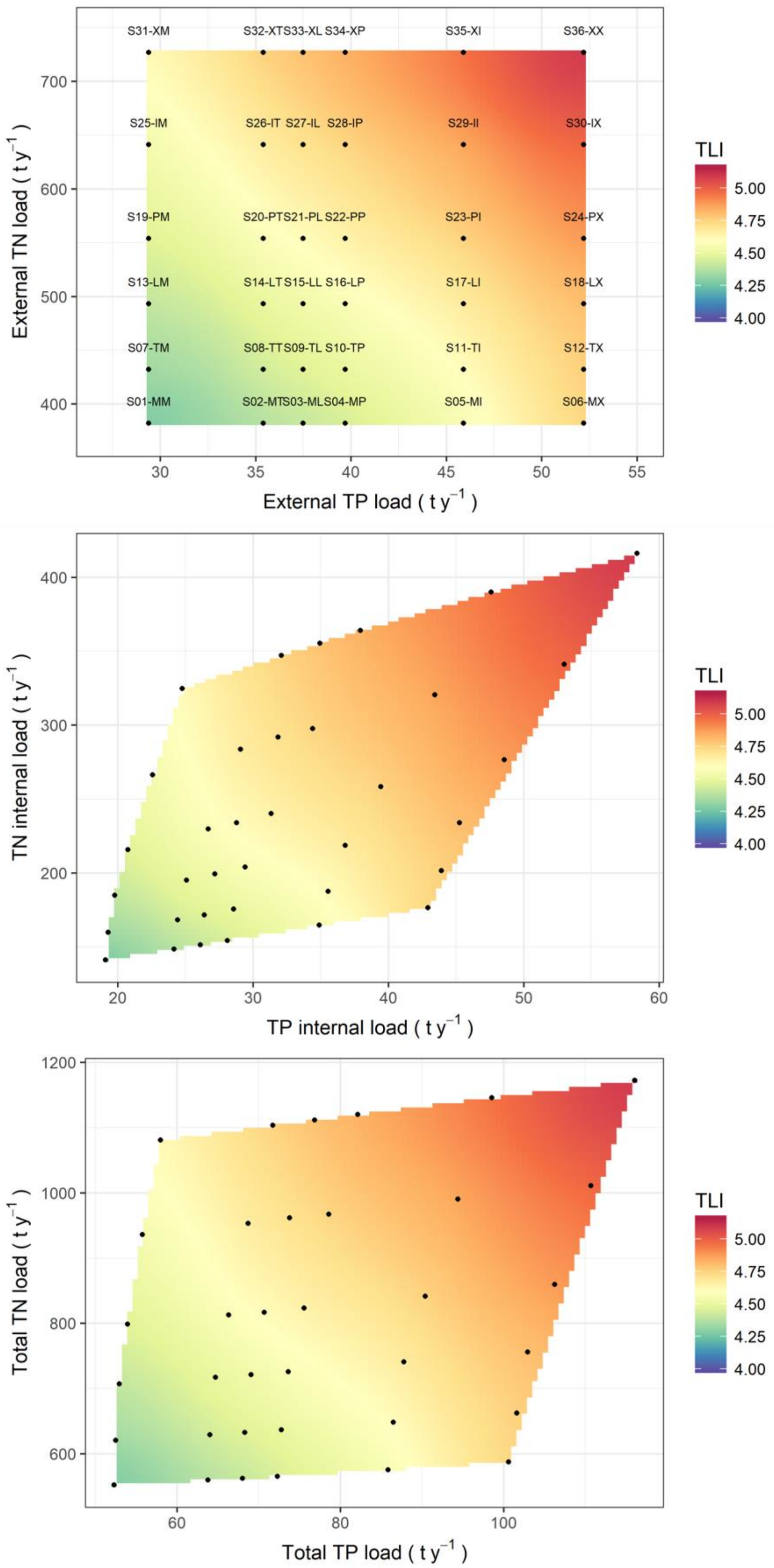


Figure 21. Simulated average TLI as a function of external load (no storm particulates), internal load and total load (external plus internal) over the scenario period from July 2000 to June 2007. The last two letters of each scenario in the upper panel represent the N and P load, respectively, where: M = minimum load, and T=Target load, L = lower load, P = Present load (2001–2007), I = Increased load, and X = Maximum load.

### Internal load reduction scenarios

A marked change in lake trophic status following alum dosing of the Utuhina and Puarenga streams is evident from in-lake observations from approximately 2006 (Figure 22). The greatest reduction was to phosphate and total phosphorus, however total nitrogen also decreased from the late-2000s. This change may be related to a reduction in total phytoplankton production (as evidenced by lower chlorophyll *a* measurements), which in turn results in less organic matter, and lower oxygen demand in the hypolimnion and sediments. The flocculating action of alum is likely to contribute, enhancing sedimentation of particulate material, including nutrients, from the water column and decreasing bioavailability of nutrients on organic matter in sediments. In support of this, exchangeable Al has been shown to be the primary control on organic matter accumulation in New Zealand soils (Percival et al. 2000). This could potentially decrease the severity and duration of anoxia, in turn decreasing sediment ammonium releases. The same feedback loop would apply to P release, however, alum may have additional effects by direct adsorption in the water column, should in-stream DRP concentrations be insufficient to exhaust the binding capacity of the alum being dosed.

Modelled water quality in Lake Rotorua is sensitive to changes in maximum potential release rates within DY-CD. The marked decrease in nutrient concentrations and chl *a* post-2010 has been attributed to alum dosing not only influencing inflow concentrations of phosphate in the two streams that are dosed, but also to the potential in-lake effects described above. Accordingly, we ran scenarios to simulate reduced sediment N and P release rates (Table 1, Figure 22), intended to indicate the degree to which alum dosing may have disrupted internal nutrient release processes, particularly since dose rates were increased substantially from 2010. Results from these scenarios suggest that the in-lake effects of alum are likely similar in magnitude to the effects of reducing the internal load of nitrogen and phosphorus by c. 50% (e.g., c. 16 t $yr^{-1}$ for phosphorus). This reduction is greater than the largest simulated external load reduction (S1-MM scenario reduces TP load by c. 10 t $yr^{-1}$). These simulations also indicate that chl *a* concentration would not be reduced to post-2010 levels without a 50 to 75% average reduction in internal loading of both and P.

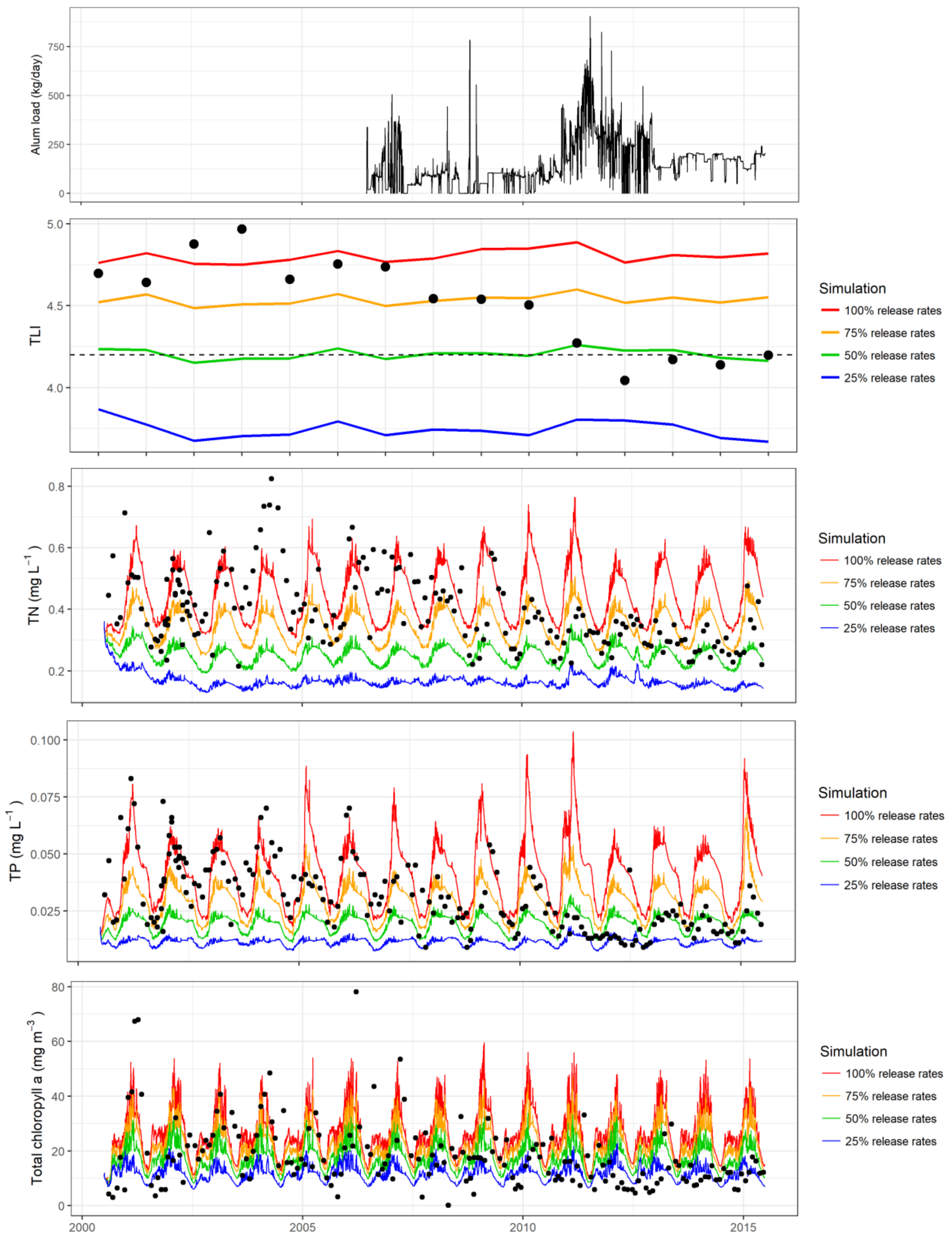


Figure 22. Alum dose rate (top panel, expressed as kg Al per day). The lower four panels show water quality observations (black dots) along with simulated values for four different scenarios of sediment nutrient flux rates. ‘100% release rates’ represents the ‘baseline’ calibrated DY-CD lake model, and other scenarios represent reductions in modelled sediment nutrient flux rates. Simulations showed that reduced sediment P release in the model produced a better goodness-of-fit to the observations post-2010, and this is consistent with the hypothesis that such an effect is induced by alum entering the lake (although it does not rule out other potential controls on sediment-P release).

## Discussion

The present application of the DYRESM-CAEDYM lake model builds upon a series of lake modelling studies aimed at understanding processes in Lake Rotorua and supporting management decisions in the lake and catchment (Rutherford et al. 1996; Burger et al. 2008; Hamilton et al. 2012; Hamilton et al. 2015; Abell et al. 2015). The calibrated model incorporates latest best-estimates of daily hydraulic and catchment loading (McBride et al., 2018). Error statistics for key state variables in the present DY-CD application were comparable and in many cases superior to, previously published applications of the model to Rotorua and other lakes (e.g., Burger, 2006; Gal et al., 2009; Özkundakci et al., 2011; Trolle et al., 2011). Improvements in model performance owe somewhat to the better characterisation of daily variation in stream discharge and nutrient concentrations (see McBride et al. 2018 for detailed description of inflow time-series generation). Simulation of water temperature and stratification dynamics was excellent (e.g., surface waters; $r = 0.99$), as was the representation of oxygen dynamics in bottom waters. Capturing the dynamics of these variables is critical to accurate estimation of internal loading (due to the dependence of key sediment nutrient fluxes on concentrations of dissolved oxygen and other oxidised compounds such as nitrate). Nutrient dynamics in surface and bottom waters were generally well-simulated, particularly pulses of N and P associated with releases from bottom sediments during stratification. Similar to previous DY-CD studies of Lake Rotorua, the model did not reproduce well the very high TN and total chlorophyll concentrations associated with well-documented and severe blooms of *Dolichospermum planktonicum* in the early 2000s, although it should be noted that very high chlorophyll values (>50 mg $m^{-3}$) were observed by BoPRC on only three occasions over this period. Note that the 1-D discretization used in DY-CD it not able to reproduce complex spatial heterogeneity of blooms, such as buoyant accumulations advected by surface currents. Improvement of the application of nitrogen fixation algorithms within CAEDYM could partially address this shortcoming in the course of future work. The calibrated model simulated cyanobacteria as the dominant phytoplankton taxa, consistent with previous assessments (e.g., Smith et al. 2016).

The target TLI stipulated by the Lake Rotorua Action plan is 4.2. Model simulations presented here for scenarios of catchment nutrient load reduction suggest considerable potential for water quality improvement through catchment load management. Simulations projected that the trophic state of the lake could decrease from the TLI before substantial alum application (TLI = 4.8 in 2008) to approximately 4.3, or near the target of 4.2. This change in TLI may appear to be modest but is associated with the logarithmic nature of the TLI constituent equations, and absolute reductions in TP, TN and chl *a* were large. For example TP and TN mean concentrations were reduced by c. 75%. Furthermore, although TLI was reduced to c. 4.3 (i.e., failed to meet the target) under the lowest loading scenarios, historical loading and TLI estimates (McBride et al., 2018) showed that catchment N load prior to 1970 was likely to have been lower than the target load of 435 t N $y^{-1}$, and lake TLI may possibly have been > 4.2 during the late-1960s or even earlier (Figure 3). Scenarios of increased loading suggested that substantial further degradation could occur if present trends of increases in the catchment nitrogen load are left unchecked. We note here that when uncertainty in model estimations is accounted for, the TLI target likely falls within the bounds of model uncertainty for scenarios with substantial load reductions.

Simulations presented here showed that only the synergistic effect of combined reductions in both TP and TN load was able to reduce TLI to near-target levels. Management of only one of these nutrients (i.e., either TN or TP) failed to result in meeting or getting close to the TLI target. This finding is consistent with previous research, including other DY-CD applications, and with the general scientific consensus that dual nutrient management and reduction is needed for effective lake restoration (Abell et al., 2010, 2015, Hamilton et al. 2015, Smith et al. 2016). It also aligns with recent bioassay studies that show that dual nitrogen and phosphorus additions most strongly stimulate increases in phytoplankton biomass (Smith et al., 2016).

Whole lake experiments have demonstrated that harmful algal blooms are stimulated more by combined N and P enrichment than by enrichment of only one nutrient (Paerl et al., 2016), and that a “P-only” or “N-only” paradigm is not appropriate. For Lake Rotorua, phytoplankton N limitation was described over a considerable time period (Vincent, 1981a, 1981b; White, 1983a; White et al., 1985; Abell et al., 2010), generally to a much greater extent than observed in many other lakes around the world. There is growing evidence, however that in Lake Rotorua, varying nutrient loading patterns and the application of alum have produced periods of co-limitation by N and P, reflective of increased water column P scarcity post-alum application. For example post 2010 there has been a reduction of bioavailable phosphate (DRP) observed in monthly water quality sampling data, alongside abundant surface DIN at times (McBride et al. 2018). Table 8 shows that modelled nutrient limitation of phytoplankton did change in response to scenario catchment nutrient loading. A 3-D model (e.g., ELCOM-CAEDYM; see Abell et al. 2013) could be used to assess the spatial variation limitation.

We calibrated and validated the DY-CD model for the period 2001-2007 in order to avoid the potential confounding influence of alum dosing from 2006 onwards (with much increased alum dose rates from 2010, see Figure 22). When the calibrated model was run from 2001-2017, this ‘baseline’ model greatly overestimated nutrient and chlorophyll concentrations post-2007 (i.e., the ‘alum period’, see Figure 22, purple line). In this sense, the baseline model over the period 2007-2017 provides an estimate of what water quality in Lake Rotorua may have been over the past 10 years if alum dosing were never commenced. Because the calibrated model accounts for changes to external loading and climate at daily time scale over the pre- and post-alum periods, these simulations provide weight to the proposition that alum dosing of Puarenga and Utuhina Streams has positively impacted lake water quality over and above any benefit arising solely from adsorbing DRP in those streams. In fact, although both dosing rates and water quality varied widely from 2007-2017, our simulations showed on balance that alum dosing had a similar effect to reducing modelled nutrient release rates (i.e., internal loads) by c. 50%. When comparing internal and external loads, it should also be noted that internal loads are dominated by dissolved, bioavailable nutrients, whereas catchment loads contain relatively more particulate and organic species of which only some fraction becomes bioavailable over time.

Overall, the observed improvement in TLI since 2007 and comparison with model results suggest that Rotorua has possibly avoided a regime shift which may be evident as a rapid shift to a super- or even hypertrophic status. We speculate that TLI values above 5.0 could lead to a persistent turbid, phytoplankton dominated state. One of the problems with a regime shift is that there can be a 'hysteresis' between the degradation phase and the rehabilitation phase which has been well demonstrated in shallow lakes (Scheffer, 2004). The problem of hysteresis occurs when a system of interacting ecological and physical processes act to stabilise a regime of a turbid, phytoplankton dominated state, with strong resistance against a return to the historic clear-water, macrophyte dominated state (Janse et al., 2008). Internal loading of phosphorus from sediments will potentially result in reduction of effectiveness of external load reductions in the short-term, and if there was a turbid phytoplankton-dominated state then it would tend to reinforce this state such that there would be a strong hysteresis. The lag time for sediments to reach equilibrium has been estimated at between 10 and 15 years (Jeppesen et al., 2005), but sometimes persisting for more than 20 years for internal loading of phosphorus (Søndergaard et al., 2003). This means that water quality improvement derived from external load reductions may not be seen for a decade or more. Trolle et al. (2011) showed progressive change in sediment nutrient concentrations which could impact nutrient fluxes and internal loading. In shallow lakes internal load can be of a similar magnitude to external load (Søndergaard et al., 1999) and this has also been shown to be the case for Lake Rotorua in the present study, and historically (Hamilton et al., 2004; Burger, 2006). However, the relatively rapid changes in Rotorua's water quality post-2010 suggest either that the recovery process has played out faster than expected, or that restoration resulting from caps on agricultural loads and alum dosing commenced during a period when the lake was 'flickering' on the verge of a regime shift (Wang et al., 2013) and have returned Rotorua to the more stable state represented by the TLI 4.2 target.

### Study limitations, recommendations and future work

The present study highlights multiple opportunities to progress the scope and robustness of catchment and lake modelling for Lake Rotorua. Here, we summarise potential additions to the lake modelling framework in order to better support accurate and effective management decisions for the catchment and lake. Improvements we can identify as priorities for development over the next 5 year cycle under LR M2 are:

1. Modelling of climate change effects on load delivery and in-lake processes.
2. Model ensemble, including multiple models with different process representations of internal load (nutrient recycling). This approach would generate a wider, potentially more realistic, range of potential outcomes of management strategies.

3. Modelling additional food web groups and higher trophic levels
4. More detailed uncertainty estimation, including Monte Carlo (MC) or Bayesian Markov Chain Monte Carlo techniques applied to parameter values, to further improve confidence in outputs by generating an envelope of expected outcomes.
5. Coupled catchment-lake modelling, including a catchment model such as the Soil and Water Assessment Tool (SWAT), capable of modelling the role of contaminant mobilisation and delivery as well as associated mitigation options (Me et al. 2018).

### *1. Simulation of climate change effects*

Climate change effects were not simulated in this study. Climate models project increases in air temperature of 0.7 to 3.7°C by 2110 as a result of increasing atmospheric $CO_2$ concentrations, dependent on emission pathways (i.e., the extent of mitigation actions taken) and the climate model used (Intergovernmental Panel on Climate Change Fifth Assessment report).

Climate change is also likely to alter seasonal patterns of rainfall, increase frequency of extreme rainfall events, and potentially alter the frequency El Niño-Southern Oscillation (ENSO) events.

There is evidence that climate change is likely to enhance eutrophication in mesotrophic and eutrophic lakes due to physiochemically- and biologically-induced internal loading increases (Paerl et al. 2011; Jeppesen et al., 2014), and warmer water temperatures favouring the production of bloom-forming phytoplankton (Hamilton et al. 2012). Previous studies on the influence of climate change in Lake Rotorua indicate that TLI could increase by 0.13 TLI units, therefore the TLI target would not be met under the maximum nutrient reductions within the present study (Hamilton et al., 2012). The scale of these changes should necessitate that their impacts are built into future plans for nutrient load controls. Lower nutrient loads will need to be prescribed in a future warmer world to meet the current ecological targets represented by the TLI. Furthermore, cyanobacteria populations are expected to increase in a warming climate (Vincent, 2009) due to physiological adaptations that may assist them to grow more rapidly and maintain biomass (Carey et al., 2012). Observations of changes in phytoplankton species composition (e.g., the marked reduction in percentage of cyanobacteria biovolume following alum dosing: Smith et al. 2016) could be used to test theories like climate change effects and N vs P limitation on cyanobacteria populations, and help to better inform modelling efforts in this area.

### *2. Ensemble modelling*

Ensemble modelling refers to the use of two or more related but independently developed process-based models in order to provide estimated ranges (or combined averages) for state variables of interest, and to reduce uncertainty in model estimates by producing an envelope of model outputs. When carried through to scenario simulations, ensemble approaches can improve confidence in model projections where outputs among various models are in agreement, and/or highlight areas of uncertainty where output diverges. For example, within climate change modelling, the aim of ensemble modelling is to deal with uncertainties in the global climate system, and improve the robustness of future climate projections. Model ensembles can inform relevant policy information, by providing probability distribution functions of outcomes for various management actions.

When applied to lake systems, ensemble modelling has the advantage that some uncertainties in individual model-estimated outcomes can be reduced by presenting the mean and range of ensemble model outputs (Mooij et al., 2010; Trolle et al., 2014). However, ensemble modelling of lake ecosystems has rarely been completed, potentially due to constraints related to resources and availability of expert modelling knowledge (Trolle et al., 2014).

Two related but independent lake models that are potential candidates for ensemble modelling are the General Lake Model - Aquatic Ecodynamics (GLM-AED) and General Ocean Transport Model - Framework of Aquatic Biogeochemical Models - PCLake (GOTM-FABM-PCLake). Both models are open source (DY-CD is only partially open).

GOTM is a 1-D hydrodynamics model that has been coupled to a complex ecological model (PClake; Hu et al., 2016, Fig. 4) via the Framework for Aquatic Biogeochemical Models (FABM, Bruggeman and Bolding, 2014). FABM is an open source FORTAN based code that facilitates the coupling of hydrodynamic and ecological models. FABM-PCLake is a redesigned version of the original PCLake (Figure 23), which was a 0-dimensional 'box model' in its original implementation (Janse et al., 1995). In contrast to DYCD and GLM, GOTM uses a fixed depth vertical layer structure (though not necessarily equally spaced, depending on model configuration).

GLM (Hipsey et al., 2017) is a 1-D hydrodynamics model that simulates vertical stratification and mixing, and accounts for the effect of inflows/outflows, surface heating and cooling. AED is a biogeochemical model similar in many respects to CAEDYM (Figure 24). GLM was developed in partnership with the Global Lake Ecological Observatory Network (GLEON), and couples with the Framework for Aquatic Biogeochemical Models (FABM), but can also run independently of FABM. GLM applies a flexible Lagrangian layer structure (generalised coordinates) similar to DY-CD (Hamilton & Schladow, 1997).

Initial scoping of the application of GLM-AED and GOTM-FABM-PCLake to Lake Rotorua has been completed. We have been able to translate DYCD inputs to PCLake and GLM, successfully run these models and compare output to field data. We foresee considerable work involved in calibrating these candidate models for routine application to Lake Rotorua. (e.g., Nielsen et al., 2014). However, initial testing has revealed that both candidate models are an order of magnitude faster in terms of runtime relative to DY-CD. This opens up opportunities for Monte Carlo based routines for auto-calibration and uncertainty analysis. Auto-calibration methods are currently under development both at the University of Waikato and abroad. For example Markov Chain Monte Carlo (MCMC) has previously applied to GLM (Bruce et al., 2018) and auto-calibration routines have previously been applied to PCLake. There are substantial advantages resulting from MCMC or other inverse parameterisation techniques, including the ability to better assess parameter uncertainty and bias.

In terms of lake process representation and algorithm choice, GLM-AED is closely related to DY-CD, sharing much of its structure and many of its algorithms. By contrast, GOTM-PCLake was developed entirely independently of DY-CD and many of its process representations differ markedly. Importantly, PCLake has a functioning dynamic sediment module, and GLM-AED has a dynamic sediment module that is optionally activated (but which has not yet been often utilised). Dynamic sediment modelling is advantageous because real-world sediment nutrient release rates vary in response to changes in sediment composition, which result from changes in external (catchment) nutrient loads and in-lake processes over interannual to decadal time-scales. Therefore, models with dynamic sediment flux rates may be capable of superior performance when applied to changing and/or productive lakes (i.e., where internal loading is substantial and variable) over multi-year timescales. Calibration of dynamic sediment nutrient flux rates represents an additional challenge compared with conventional (~static-flux) biogeochemical models such as CAEDYM, however, long-term simulations of Lake Rotorua provide an ideal test case for dynamic sediment representation within lake models, particularly to align with measurements of sediment composition through time (Trolle et al. 2011). The responsiveness of internal loads to a range of factors is an appropriate subject for further research, including different species contributing external loads (i.e., dissolved versus particulate), the historical effects of alum dosing, and climate change. Regular and continued sediment composition studies will be informative to future modelling efforts.

### 3. Modelling additional food web groups and higher biology

The present version of DY-CD does not simulate the potential ecologically structuring role of aquatic macrophytes, which may be enhanced with lower concentrations of phytoplankton increasing available light for macrophyte growth. Increased macrophyte coverage has the potential to reduce sediment and particulate nutrient resuspension, and lock up or store sediment and water column nutrients. These synergistic effects could amplify changes in TLI in response to loading and water quality changes. The structural role of littoral vegetation and the associated food web of this region, is increasingly recognised as having important lake-wide effects (Stewart et al., 2017), and is likely especially important for Rotorua, where the shallow gradient of lake shores results in a large proportion of the lake bottom lying within the euphotic zone.

Further, zooplankton and other groups of the lake food chain (i.e., higher biology) were not simulated, meaning the model does not account for top-down control on phytoplankton (and hence nutrient cycling), other than through an increased model coefficient for phytoplankton respiration, representative of zooplankton predation. This could be a long-term objective and might help to further refine model outputs. The model PCLake is well-suited to simulation of higher biology, and an important evolution of Lake Rotorua modelling could include, for example, grazing by zooplankton, simulation of effects of thermal regime on trout populations under a changing climate, and the establishment and likely effects of invasive fish incursions such as the brown bullhead catfish (*Ameirurus nebulosis*).

### *4. Uncertainty and sensitivity analysis*

Complex ecological models such as DY-CD generally provide a single deterministic outcome to a given set of input conditions, and, due to the observation error and noise, calibrated parameters are usually quite uncertain (i.e., there are typically multiple sets of parameters that would lead to a similar or better model performance than the chosen calibrated parameter set). This uncertainty is propagated to model state variable estimation, e.g., TLI. Therefore, rigorous uncertainty analysis is required to present not only multiple solutions to calibrated models, but propagate these solutions (or realizations) to scenarios, creating probability distribution functions of outcomes. This will ultimately lead environmental managers to assess the risk involved in the simulation of scenarios that are designed to support management decisions. Rigorous uncertainty analysis would strengthen the application of predictions made by environmental models in supporting policy development (Özkundakci et al., 2018).

The present study includes a simple global sensitivity analysis, however there is considerable scope to refine this process, including the application of state of the art parameterisation/uncertainty techniques such linear-based, first-order, second-moment (FOSM) uncertainty analyses (White et al., 2016), also known as Bayes linear theory, and Markov Chain Monte Carlo. Uncertainty analysis can be used for a model to indicate both the sensitivity of model output to single coefficients, and/or sensitivity to synergistic effects of changes in multiple coefficients, as well as indicating the degree of uncertainty in overall model performance. Sensitivity analyses for multiple (ensemble) models could be used to describe a yet broader envelope of potential outcomes.

### *5. Coupled catchment-lake models*

The present application of DY-CD to Lake Rotorua was configured using daily stream inflow volume and quality estimated from empirical observations (see McBride et al., 2018). Some previous applications of DY-CD to Rotorua have used output from the catchment nutrient routing model ROTAN (see, for example, Hamilton et al., 2012). An advantage of this coupled catchment-lake modelling approach is that catchment models can be used to simulate management scenarios on the landscape which can be carried through to lake simulations. ROTAN, however, is at present limited to an annual time-step, and is capable of simulating nitrogen but not phosphorus. Therefore, for the present study, we chose to use empirically-derived stream inputs, and instead used ROTAN scenario simulations (Rutherford et al., 2016) upon which the relative change in load from present model input was able to be derived (see Methods).

Future work could re-focus on coupling the present, improved DY-CD lake model with existing or novel catchment model applications. A time-resolved version of ROTAN could be reinstated, and perhaps augmented to include representation of phosphorus and sediment. Alternatively, the catchment model SWAT has previously been applied successfully to the Puarenga sub-catchment and coupled to an earlier version of the DY-CD lake model (see Me et al., 2015, 2018). The SWAT model simulates dissolved and particulate N and P, as well as suspended sediment (SS), and has recently been coupled to the dynamic groundwater model MODFLOW. Extension of SWAT simulations to whole-catchment scale for Rotorua, would provide ample opportunity for modelling the role of contaminant mobilisation and delivery as well as associated mitigation options, and carrying these through to projecting in-lake effects.

## Conclusions

This project represents an evolution of model application to Rotorua refined over the past 12 years and through multiple studies (Burger et al., 2008; Hamilton et al. 2012; Hamilton et al. 2015; Abell et al., 2015). Nevertheless, considerable scope exists to improve the robustness and utility of model simulations of management strategies for Lake Rotorua, through the application of multiple models, coupled catchment models, inclusion of climate change scenarios, and thorough sensitivity analyses and auto-calibration techniques. Model executables and configuration files for Lake Rotorua are maintained by The University of Waiakto, and the present model could be updated at any time using updated inflow and meteorological data as these become available. Improved knowledge of biological and physical processes may enable better calibration of model parameters, however we consider present model performance to be such that changes to simulation performance would be relatively minor. While at present we believe DY-CD to be the best suited model to the current application, DY-CD is not unique in its capacity, and if the need arises other models could be applied.

Detailed modelling of Lake Rotorua catchment inputs and in-lake processes has shown that substantial reductions in external loading will be required to achieve similar water quality to that achieved over the past 10 years of active management in Rotorua. Modelling results support the assertion that alum dosing has disrupted in-lake degradation processes leading to a reduction of both nutrients and phytoplankton in the lake over the past 10 years. These simulations reinforce the urgent need to work towards established nutrient targets, and the importance of combined reductions in both nitrogen and phosphorus loads to achieve the TLI target of 4.2.

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# Appendix 1: calibrated model parameters

Table 5. 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 | - | 0.08 | Patten et al. (1975) |
| Potential energy mixing efficiency | - | 0.2 | Spigel et al. (1986) |
| Shear production efficiency | - | 0.3 | Spigel et al. (1986) |
| Wind stirring efficiency | - | 0.23 | Spigel et al. (1986) |
| Vertical mixing coefficient | - | 500 | Yeates & Imberger (2003) |
| Effective surface area coefficient | $m^2$ | $1.0 \times 10^7$ | Standard value |

Table 6. 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 | $g\ m^{-2}\ d^{-1}$ | 3.07 | 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 | $g\ m^{-2}\ d^{-1}$ | 0.4 | |
| Oxygen half-saturation constant for release of ammonium from bottom sediments | $g\ m^{-3}$ | 3.0 | |
| Maximum potential $NO_3$ release rate | $g\ m^{-2}\ d^{-1}$ | -0.15 | |
| Oxygen half-saturation constant for release of nitrate from bottom sediments | $g\ m^{-3}$ | 2.0 | |
| 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* | | *Cyanobacteria, Chlorophytes, Diatoms* | |
|---|---|---|---|
| Maximum potential growth rate at 20°C | $d^{-1}$ | 0.55, 1.00, 1.50 | Robson & Hamilton (2004) |
| Irradiance parameter non-photoinhibited growth | µmol $m^{-2}$ $s^{-1}$ | 380, 50, 20 | Robson & Hamilton (2004) |
| Half saturation constant for phosphorus uptake | mg $L^{-1}$ | 0.006, 0.005, 0.004 | Trolle et al. (2008) |
| Half saturation constant for nitrogen uptake | mg $L^{-1}$ | 0.01, 0.12, 0.18 | Trolle et al. (2008) |
| Minimum internal nitrogen concentration | mg N (mg chl a)$^{-1}$ | 4.5, 3.0, 4.0 | Schladow & Hamilton (1997) |
| Maximum internal nitrogen concentration | mg N (mg chl a)$^{-1}$ | 14, 12, 12 | Schladow & Hamilton (1997) |
| Maximum rate of nitrogen uptake | mg N (mg chl a)$^{-1}$ $d^{-1}$ | 14, 12, 12 | Schladow & Hamilton (1997) |
| Minimum internal phosphorus concentration | mg P (mg chl a)$^{-1}$ | 4.5, 3.5, 3.0 | Schladow & Hamilton (1997) |
| Maximum internal phosphorus concentration | mg P (mg chl a)$^{-1}$ | 1.2, 0.8, 0.8 | Schladow & Hamilton (1997) |
| Maximum rate of phosphorus uptake | mg P (mg chl a)$^{-1}$ $d^{-1}$ | 0.3, 0.15, 0.2 | Schladow & Hamilton (1997) |
| Temperature multiplier for growth limitation | - | 1.07, 1.04, 1.04 | Schladow & Hamilton (1997) |
| Standard temperature for growth | °C | 20, 10, 7 | Gal et al. (2009) |
| Optimum temperature for growth | °C | 22, 20, 8 | Gal et al. (2009) |
| Maximum temperature for growth | °C | 33, 35, 9.5 | Gal et al. (2009) |
| Respiration rate coefficient | $d^{-1}$ | 0.051, 0.06, 0.07 | Schladow & Hamilton (1997) |
| Temperature multiplier for respiration | - | 1.04, 1.08, 1.08 | Schladow & Hamilton (1997) |
| Fraction of respiration relative to total metabolic loss rate | - | 0.6, 0.9, 0.9 | |
| Fraction of metabolic loss rate that goes to DOM | - | 0.01, 0.1, 0.05 | |
| Constant settling velocity | m $s^{-1}$ | 0.25x10-4,-0.23x10-6, -0.3x10-5 | Burger et al. (2007a) |

# Appendix 2: comparison of total load components for all 36 model scenarios

Table 7. Summary scenario results including all components of external and internal nutrient loading, phytoplankton nutrient limitation function coefficients resulting from each scenario (whereby 0 = full limitation and 1 = no limitation), as well as trophic lake index (TLI) including TLn (total nitrogen), TLp (total phosphorous), and TLc (total chlorophyll *a*), TLs (Secchi).

| | Model input – nutrient loads | | | | | | | | | | Model output – water quality | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Nitrogen (t $y^{-1}$) | | | | | Nitrogen (t $y^{-1}$) | | | | | Nutrient limitation | | Trophic Level Index | | | | |
| Scenario | Catchment DIN load | Catchment TN load (NSP) | Catchment TN load (All) | Internal N load | Catchment + internal N load | Catchment DRP load | Catchment TP load (NSP) | Catchment TP load (All) | Internal P load | Catchment + internal P load | Phyto. N limitation (0 to 1) | Phyto. P limitation (0 to1) | TLn | TLp | TLc | TLs | TLI |
| **S01-MM** | 370.48 | 382.00 | 410.92 | 141.13 | 552.04 | 29.40 | 33.20 | 19.11 | 22.41 | 52.31 | 0.68 | 0.76 | 3.75 | 4.18 | 5.34 | 3.89 | 4.29 |
| **S02-MT** | 370.48 | 382.00 | 410.92 | 148.62 | 559.53 | 35.40 | 39.64 | 24.16 | 26.76 | 63.80 | 0.69 | 0.82 | 3.79 | 4.48 | 5.37 | 3.90 | 4.38 |
| **S03-ML** | 370.48 | 382.00 | 410.92 | 151.46 | 562.38 | 37.50 | 41.93 | 26.10 | 28.31 | 68.03 | 0.70 | 0.83 | 3.81 | 4.58 | 5.38 | 3.90 | 4.42 |
| **S04-MP** | 370.48 | 382.00 | 410.92 | 154.28 | 565.20 | 39.70 | 44.22 | 28.10 | 29.85 | 72.32 | 0.70 | 0.85 | 3.82 | 4.69 | 5.39 | 3.91 | 4.45 |
| **S05-MI** | 370.48 | 382.00 | 410.92 | 164.64 | 575.55 | 45.90 | 50.96 | 34.90 | 34.40 | 85.86 | 0.71 | 0.89 | 3.88 | 5.06 | 5.42 | 3.92 | 4.57 |
| **S06-MX** | 370.48 | 382.00 | 410.92 | 176.39 | 587.30 | 52.20 | 57.70 | 42.91 | 38.95 | 100.61 | 0.73 | 0.91 | 3.93 | 5.61 | 5.46 | 3.94 | 4.73 |
| **S07-TM** | 416.17 | 432.00 | 460.94 | 159.75 | 620.69 | 29.40 | 33.20 | 19.30 | 22.41 | 52.50 | 0.71 | 0.73 | 3.88 | 4.16 | 5.42 | 3.92 | 4.34 |
| **S08-TT** | 416.17 | 432.00 | 460.94 | 168.31 | 629.25 | 35.40 | 39.64 | 24.42 | 26.76 | 64.06 | 0.72 | 0.80 | 3.92 | 4.45 | 5.44 | 3.93 | 4.44 |
| **S09-TL** | 416.17 | 432.00 | 460.94 | 171.61 | 632.55 | 37.50 | 41.93 | 26.40 | 28.31 | 68.33 | 0.72 | 0.81 | 3.94 | 4.55 | 5.45 | 3.94 | 4.47 |
| **S10-TP** | 416.17 | 432.00 | 460.94 | 175.55 | 636.49 | 39.70 | 44.22 | 28.57 | 29.85 | 72.79 | 0.73 | 0.83 | 3.96 | 4.65 | 5.47 | 3.94 | 4.51 |
| **S11-TI** | 416.17 | 432.00 | 460.94 | 187.49 | 648.43 | 45.90 | 50.96 | 35.55 | 34.40 | 86.51 | 0.74 | 0.87 | 4.01 | 4.99 | 5.49 | 3.96 | 4.61 |
| **S12-TX** | 416.17 | 432.00 | 460.94 | 201.52 | 662.46 | 52.20 | 57.70 | 43.92 | 38.95 | 101.61 | 0.75 | 0.90 | 4.07 | 5.44 | 5.53 | 3.97 | 4.75 |
| **S13-LM** | 471.97 | 493.00 | 522.04 | 184.89 | 706.93 | 29.40 | 33.20 | 19.77 | 22.41 | 52.97 | 0.75 | 0.71 | 4.03 | 4.15 | 5.49 | 3.96 | 4.41 |
| **S14-LT** | 471.97 | 493.00 | 522.04 | 195.22 | 717.26 | 35.40 | 39.64 | 25.09 | 26.76 | 64.73 | 0.75 | 0.77 | 4.07 | 4.44 | 5.53 | 3.97 | 4.50 |
| **S15-LL** | 471.97 | 493.00 | 522.04 | 199.32 | 721.36 | 37.50 | 41.93 | 27.17 | 28.31 | 69.10 | 0.76 | 0.80 | 4.09 | 4.54 | 5.53 | 3.98 | 4.54 |
| **S16-LP** | 471.97 | 493.00 | 522.04 | 203.93 | 725.96 | 39.70 | 44.22 | 29.42 | 29.85 | 73.64 | 0.76 | 0.81 | 4.11 | 4.65 | 5.54 | 3.98 | 4.57 |
| **S17-LI** | 471.97 | 493.00 | 522.04 | 218.61 | 740.65 | 45.90 | 50.96 | 36.83 | 34.40 | 87.78 | 0.77 | 0.86 | 4.17 | 4.96 | 5.57 | 4.00 | 4.67 |
| **S18-LX** | 471.97 | 493.00 | 522.04 | 233.98 | 756.02 | 52.20 | 57.70 | 45.25 | 38.95 | 102.94 | 0.79 | 0.90 | 4.22 | 5.35 | 5.60 | 4.01 | 4.80 |
| **S19-PM** | 527.77 | 554.00 | 583.13 | 215.68 | 798.81 | 29.40 | 33.20 | 20.76 | 22.41 | 53.96 | 0.79 | 0.70 | 4.18 | 4.18 | 5.55 | 3.99 | 4.47 |
| **S20-PT** | 527.77 | 554.00 | 583.13 | 229.80 | 812.93 | 35.40 | 39.64 | 26.68 | 26.76 | 66.32 | 0.79 | 0.76 | 4.23 | 4.48 | 5.60 | 4.01 | 4.58 |
| **S21-PL** | 527.77 | 554.00 | 583.13 | 233.85 | 816.98 | 37.50 | 41.93 | 28.78 | 28.31 | 70.71 | 0.79 | 0.78 | 4.25 | 4.58 | 5.61 | 4.02 | 4.61 |
| **S22-PP** | 527.77 | 554.00 | 583.13 | 240.09 | 823.22 | 39.70 | 44.22 | 31.34 | 29.85 | 75.56 | 0.79 | 0.80 | 4.27 | 4.68 | 5.62 | 4.02 | 4.65 |
| **S23-PI** | 527.77 | 554.00 | 583.13 | 258.27 | 841.40 | 45.90 | 50.96 | 39.45 | 34.40 | 90.41 | 0.81 | 0.85 | 4.33 | 5.00 | 5.64 | 4.03 | 4.75 |
| **S24-PX** | 527.77 | 554.00 | 583.13 | 276.56 | 859.69 | 52.20 | 57.70 | 48.57 | 38.95 | 106.26 | 0.82 | 0.89 | 4.39 | 5.39 | 5.67 | 4.05 | 4.87 |
| **S25-IM** | 606.76 | 641.00 | 669.61 | 266.33 | 935.93 | 29.40 | 33.20 | 22.59 | 22.41 | 55.79 | 0.84 | 0.69 | 4.37 | 4.23 | 5.59 | 4.01 | 4.55 |
| **S26-IT** | 606.76 | 641.00 | 669.61 | 283.62 | 953.23 | 35.40 | 39.64 | 29.08 | 26.76 | 68.72 | 0.84 | 0.76 | 4.43 | 4.54 | 5.66 | 4.04 | 4.67 |
| **S27-IL** | 606.76 | 641.00 | 669.61 | 291.77 | 961.38 | 37.50 | 41.93 | 31.86 | 28.31 | 73.79 | 0.84 | 0.78 | 4.45 | 4.66 | 5.67 | 4.05 | 4.71 |
| **S28-IP** | 606.76 | 641.00 | 669.61 | 297.69 | 967.30 | 39.70 | 44.22 | 34.40 | 29.85 | 78.62 | 0.84 | 0.80 | 4.47 | 4.75 | 5.69 | 4.06 | 4.74 |
| **S29-II** | 606.76 | 641.00 | 669.61 | 320.51 | 990.12 | 45.90 | 50.96 | 43.44 | 34.40 | 94.40 | 0.85 | 0.85 | 4.53 | 5.10 | 5.72 | 4.07 | 4.85 |
| **S30-IX** | 606.76 | 641.00 | 669.61 | 341.22 | 1010.83 | 52.20 | 57.70 | 52.99 | 38.95 | 110.69 | 0.86 | 0.89 | 4.58 | 5.50 | 5.73 | 4.08 | 4.97 |
| **S31-XM** | 685.75 | 727.00 | 756.09 | 324.70 | 1080.79 | 29.40 | 33.20 | 24.78 | 22.41 | 57.98 | 0.87 | 0.68 | 4.56 | 4.21 | 5.66 | 4.05 | 4.62 |
| **S32-XT** | 685.75 | 727.00 | 756.09 | 347.13 | 1103.21 | 35.40 | 39.64 | 32.10 | 26.76 | 71.74 | 0.87 | 0.76 | 4.61 | 4.59 | 5.71 | 4.08 | 4.75 |
| **S33-XL** | 685.75 | 727.00 | 756.09 | 355.39 | 1111.48 | 37.50 | 41.93 | 34.93 | 28.31 | 76.87 | 0.87 | 0.78 | 4.64 | 4.70 | 5.73 | 4.09 | 4.79 |
| **S34-XP** | 685.75 | 727.00 | 756.09 | 363.93 | 1120.02 | 39.70 | 44.22 | 37.93 | 29.85 | 82.15 | 0.87 | 0.80 | 4.66 | 4.82 | 5.75 | 4.09 | 4.83 |
| **S35-XI** | 685.75 | 727.00 | 756.09 | 389.84 | 1145.93 | 45.90 | 50.96 | 47.60 | 34.40 | 98.56 | 0.88 | 0.85 | 4.72 | 5.20 | 5.77 | 4.11 | 4.95 |
| **S36-XX** | 685.75 | 727.00 | 756.09 | 416.06 | 1172.14 | 52.20 | 57.70 | 58.38 | 38.95 | 116.07 | 0.88 | 0.89 | 4.78 | 5.63 | 5.80 | 4.12 | 5.08 |

# Appendix 3: schematic diagrams of lake models

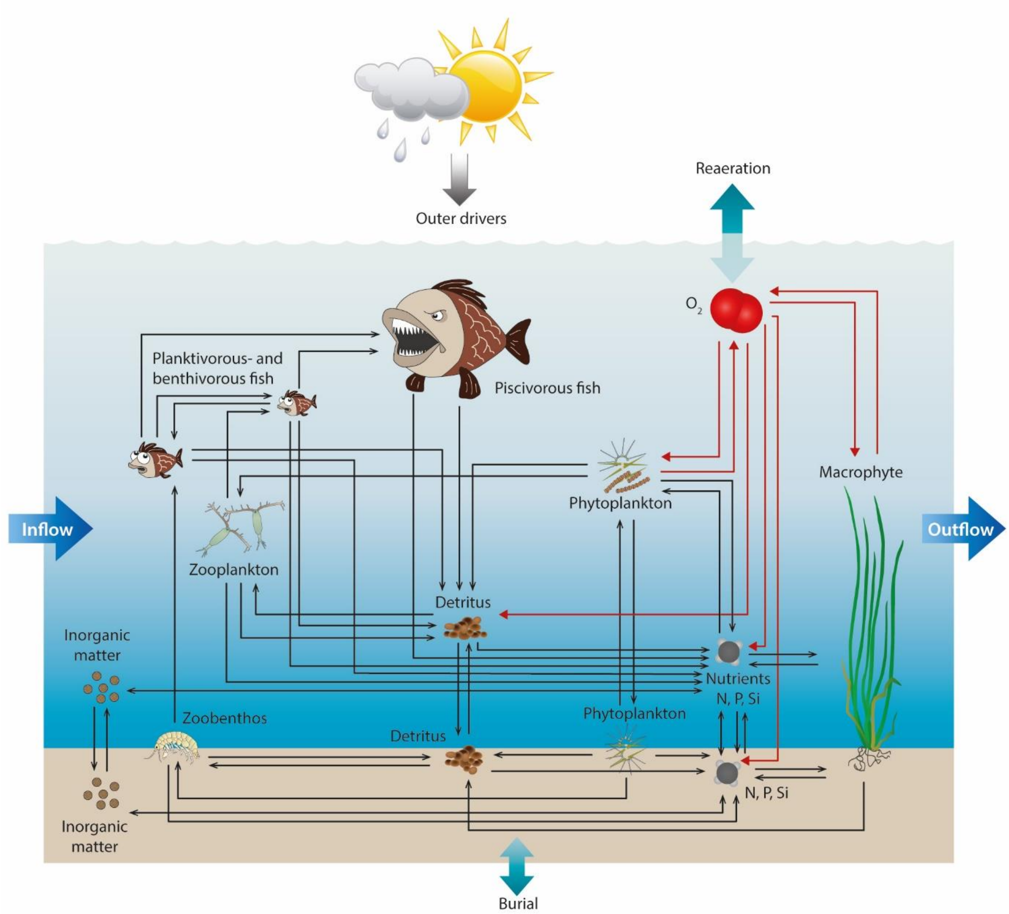


Figure 23. Conceptual model of ecological state variables simulated by PCLake (from Hu et al. 2016).

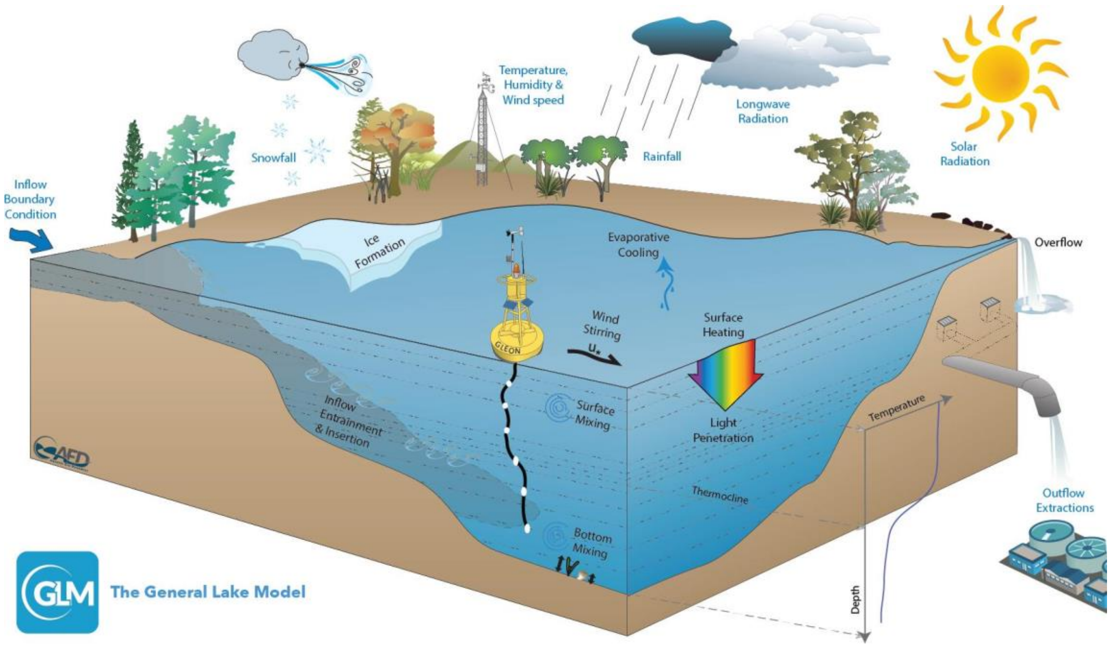


Figure 24. Conceptual model of physical state variables simulated by GLM (Hipsey et al. 2017).