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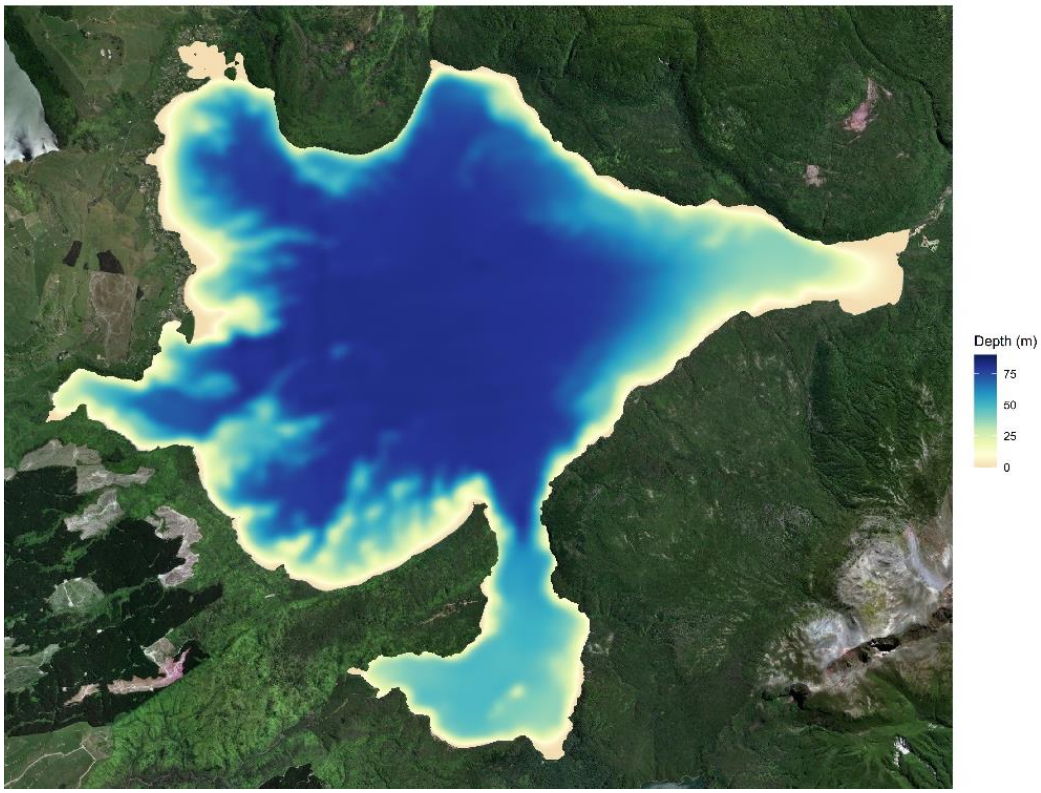
Assessing effects of changes to nutrient loads on Lake Tarawera water quality:

Model simulations for 2010 to 2020



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Executive Summary

Lake Tarawera is a nationally significant lake that is highly valued by tangata whenua, local residents and the regional community. Monitoring shows that lake water quality does not presently meet the target, based on a Trophic Level Index (TLI) value of 2.6 identified in the Tarawera Lakes Restoration Plan[1]. Between 2010 and 2020 annual observed TLI was frequently as high as 2.9. Managing nutrient loads to the lake is necessary to achieve desired lake water quality and interim nutrient reduction targets have been established. An objective of this study was to evaluate these targets by improving estimates of 'sustainable nutrient loads', which are the external loads of nitrogen and phosphorus that would result in meeting the TLI target.

To estimate sustainable loads, a one-dimensional lake ecosystem model was configured for Lake Tarawera for the period July 2010 to June 2020. The model comprised a hydrodynamic model (DYRESM) that was coupled to a water quality model (CAEDYM). The model was configured with 12 separate inflows, including surface streams, groundwater springs around the edge of the lake, and groundwater discharge to the bed of the lake. Groundwater discharge to the bed of the lake is greater than all surface discharges combined, and was estimated with reference to groundwater modelling conducted by GNS Science[2]. Inflow water quality was configured based on measurements collected at multiple (>30) inflows during 2006–2020. Nutrient concentrations in groundwater were based on volume-weighted average concentrations of measurements in surface waters and assumptions about groundwater concentrations. This approach is consistent with earlier nutrient budget studies, although there is considerable uncertainty regarding nutrient concentrations in groundwater, particularly from geothermal sources. For context, the assumed groundwater nitrogen concentration (as nitrate) used in the model was ~15% lower than the concentration estimated by White *et al.* (2016), although the higher value is based on steady state conditions, which have yet to be realised due to lags in groundwater transit.

The average annual TN load to the lake represented in the model was 112.7 t/y, while the annual TP load to the lake represented in the model was 13.03 t/y. These average external nutrient loads were calculated using the most current data available and are ~15–20% higher than the estimated loads presented in the Tarawera Lakes Restoration Plan. These differences partly reflect revised (higher) estimates of groundwater inflow. The external nitrogen load estimated in this study was 5% (6 t N/y) higher than the load recently estimated by McBride *et al*. (2020)[3], whereas the phosphorus load estimated in this study was 18% (2.3 t P/y) higher than the load estimated in that study. McBride *et*

[1] BoPRC (Bay of Plenty Regional Council). 2015. Tarawera Lakes Restoration Plan. Whakatāne, New Zealand.

[2] White, P., C. Tschritter, M. Toews and A. Lovett. 2016. Nitrogen Discharge from the Groundwater System to Lakes and Streams in the Greater Lake Tarawera Catchment. GNS Science Consultancy Report 2015/108.

[3] McBride, C.G., A. MacCormick and P. Verburg. 2020. Estimated Catchment Loads of Nitrogen and Phosphorus to the Rotorua Te Arawa Lakes. Environmental Research Institute Report prepared for Bay of Plenty Regional Council. The University of Waikato. 93 p.

al. (2020) used a different method (land use export coefficient modelling) than this study, and therefore the similarity between the two independent sets of estimates provides some confidence in the accuracy of the estimates. Confidence in the estimates developed in this study and by McBride *et al*. (2020) is considered similar, and therefore managers should consider these two studies in combination to understand the ranges of external nutrient loads that are plausible.

The model was calibrated using an automated procedure to effectively examine the parameter space. Manual calibration was then undertaken to further refine parameter values, prior to validation with an independent dataset. The model satisfactorily simulated phytoplankton dynamics and the key nutrient cycling processes in Lake Tarawera. Simulated daily mean concentrations of TN, TP and chlorophyll *a* were used to calculate annual TLI_3 values. These are analogous to standard Trophic Level Index values, except Secchi depth was not included in the calculation because this variable was not explicitly simulated with the model. Analysis showed that error due to omission of Secchi depth data from the calculation was negligible, and therefore TLI_3 values predicted with the model were considered interchangeable with values of the standard four-variable TLI. For the baseline simulation period of 2010–2020, the model predicted TLI_3 with a mean absolute error of 0.15 TLI_3 units. The difference between the mean modelled and measured values of TLI_3 for the simulation period was 0.08 TLI_3 units.

Thirty-six nutrient load scenarios were simulated, with the aim of identifying sustainable levels of nutrient input that would result in achieving the TLI target. Scenarios were configured by adjusting the 'manageable' portions of the external TN and TP loads to the lake by factors between 0.6 and 1.2. The 'manageable' portions were assumed to comprise total external loads, minus loads from atmospheric deposition and geothermal inflows. Maximum sediment nutrient release rates were adjusted by the same factors to represent changes to internal loads. Results showed that maintaining current nitrogen loads (112.7 t/y) and reducing the external phosphorus load by an average of 1.2 t P/y would achieve a 10-year average TLI_3 of 2.57, which is just less than the target of 2.6. This scenario is based on the current interim target for the lake and corresponds to a 13% reduction to the 'manageable' portion of the external phosphorus load, or 9% of the total external phosphorus load. This scenario–named '1N:0.87P'–was configured by also reducing maximum sediment phosphorus release rates by 13% to reflect reductions to internal loads that are assumed to occur once benthic sediment nutrient stores have equlibrated to reductions in external loads. Additional permutations of nitrogen and phosphorus load reductions that are predicted to comply with the target were identified and tabulated in Appendix D.

Scenario results therefore indicate that the current interim nutrient management targets to reduce phosphorus loads by 1.2 t P/y while holding nitrogen loads constant are expected to improve lake water quality and achieve the TLI target. This outcome needs to be considered in the context of the projected increases to groundwater nitrogen concentrations that are described by GNS researchers (White *et al.* 2016), as this indicates that actions to manage nitrogen (as well as phosphorus) in the wider catchment are nonetheless required to meet the target.

Sensitivity analyses were undertaken to examine interactions with climate change and uncertainty in sediment nutrient release rates. A key finding was that increased water temperatures associated with

climate change are expected to increase thermal stability of the lake and increase the duration of the stratified period, likely resulting in substantive yet uncertain effects on lake water quality. Sediment nutrient release sensitivity analysis involved simulating all scenarios but leaving maximum sediment nutrient release rates unchanged from the baseline configuration to represent expected conditions before internal loads have equilibrated to external load reductions. These sediment nutrient release sensitivity analyses showed that the load scenario results were relatively insensitive to the rate of decline in sediment nutrient release rates. This reflects that internal loads were estimated to comprise minor components of the total nutrient loads to the lake (5% of total loads for nitrogen and 15% of total loads for phosphorus). Nitrogen fixation by cyanobacteria was simulated within the lake model and was estimated to contribute ~20% of the external nitrogen load; this modelled estimate is substantially higher than a previous estimate based on empirical observations, and the role of nitrogen fixation in the lake nitrogen budget is uncertain. There is also considerable uncertainty regarding the hydrology and chemistry of groundwater fluxes to the bed of the lake and further work to quantify these fluxes should be considered a priority.

The nutrient load scenario results (matrices) produced in this study provide a tool for lake managers to efficiently evaluate the potential outcomes of nutrient management in the catchment on lake water quality. Simulations presented here suggest that current catchment nutrient management targets are appropriate for achieving target water quality in the lake. Further, the lake model provides a decision support tool to assist Bay of Plenty Regional Council with ongoing lake and catchment management. The model has been configured to potentially incorporate inputs from future groundwater and surface hydrology modelling to evaluate the effects of land use change scenarios on lake water quality.

Acknowledgments

Terry Beckett led the sampling programme of lake inflows. David Hamilton (Griffith University) contributed to an earlier stage of the study and Paul White (GNS Science) provided details regarding GNS Science groundwater modelling. Andy Bruere (BoPRC) and Alistair MacCormick provided support and direction. João Braga (Ecofish Research Ltd.) supported with preparing figures. DYRESM-CAEDYM was developed at the Centre for Water Research at the University of Western Australia.

Table of Contents

Executive Summary iii
Acknowledgments vi
Table of Contents vii
List of Tables viii
List of Figures viii
List of Appendices 9
Appendix A Inflow Water Quality Data 9
Appendix B Model Configuration Details 9
Appendix C Calibrated Parameter Values and Model Performance 9
Appendix D Detailed Scenario Results 9
Introduction 10
Background and Objectives 12
Lake and Catchment Description 12
Lake Water Quality 13
Previous Monitoring and Research 17
Lake Nutrient Loads and Targets 19
Study Objectives 19
Methods 20
Model Selection 20
Model Overview 20
Model Application 23
Model Scenarios 24
TLI3 vs TLI 25
Sensitivity Analysis 26
Climate Change 26
Sediment Nutrient Release 27
Results and Discussion 28
Nutrient Loads 28
Model Performance 32
Nutrient Load Scenario Results 35
Sensitivity Analysis 37
Climate Change 37
Sediment Release Rates 40
Uncertainties 41
Conclusion 43
References 45

List of Tables

Table 1. Morphometric characteristics of Lake Tarawera (BoPRC 2015). 12

Table 2. Overview of the Trophic Level Index (TLI) classification scheme (Burns et al. 1999). 14

Table 3. Comparison of annual average nutrient loads to Lake Tarawera estimated in this study, with loads estimated in previous studies. The standard deviations of estimates from comparable studies are provided to indicate variance. 32

List of Figures

Figure 1. Annual Trophic Level Index (TLI) values for Lake Tarawera. 15

Figure 2. Lake Tarawera water quality data. Total nitrogen and phosphorus concentrations were measured by NIWA at the lake outflow; chlorophyll *a* and Secchi depth measurements were collected in the lake by BoPRC. 16

Figure 3. Comparison of paired concentrations of total nitrogen (TN) and total phosphorus (TP) measured by Bay of Plenty Regional Council (BoPRC) in Lake Tarawera surface water, and by NIWA at the lake outlet. BoPRC measurements have been adjusted based on McBride and Baisden (2019) to reduce suspected laboratory error. 17

Figure 4. Conceptual diagrams of the cycling of nitrogen (A) and phosphorus (B) within the water quality model (CAEDYM). DONL, labile dissolved organic nitrogen; PONL, labile particulate organic nitrogen; DOPL, labile dissolved organic phosphorus; POPL, labile particulate organic phosphorus. 22

Figure 5. Summary of inflows (blue) and outflows (yellow) included in the lake model. Vertical lines show ±1 standard deviation. See Appendix B for details of all inflows (n=12) included in the model. 23

Figure 6. Nutrient load scenarios. 25

Figure 7. Measured values of TLI and TLI_3. 26

Figure 8. Mean annual nutrient loads for all inflows included in the model. See Figure 9 for a summary. 30

Figure 9. Summary of mean annual nutrient loads for categories of inflows. See Figure 8 for loads for individual inflows. 31

Figure 10. Simulated water temperature. 33

Figure 11. Simulated chlorophyll *a* concentration. 34

Figure 12. Modelled and measured values of TLI_3 and the three component indices, 2011–2020. 34

Figure 13. Summary of nutrient loading scenario results, for a) external loading only and b) external plus internal loading for nitrogen and phosphorus. Each dot represents a nutrient loading scenario representing changes to the manageable portions of the external nitrogen and phosphorus loads. The interpolated coloured

background represents the average three-parameter Trophic Level Index for a 10-year model simulation (2011–2020). The blue isobar represents the TLI target of 2.6 (note the target is set using a four-parameter TLI). See Appendix D for similar plots for each TLI component. 36

Figure 14. Lake surface water temperatures for the baseline simulation and the four climate change sensitivity scenarios. Simulated concentrations of chlorophyll *a* associated with each phytoplankton group, total chlorophyll *a*, total nitrogen and total phosphorus for the baseline simulation and the four climate change (cc) sensitivity scenarios: 1) 'low' – mean increase to baseline air temperatures of 0.26°C (RCP2.6); 2) 'medium' – mean increase to baseline air temperatures of 1.31°C (RCP6.0); 3) 'high' – mean increase to baseline air temperatures of 2.26°C (RCP8.5), and; 4) 'hist' – mean air temperature 0.61°C cooler than the model baseline period (2010–2020), corresponding to historical conditions during 1971–1990. 38

Figure 15. Water column state for the baseline simulation and the four climate change (cc) sensitivity scenarios: 1) 'low' – mean increase to baseline air temperatures of 0.26°C (RCP2.6); 2) 'medium' – mean increase to baseline air temperatures of 1.31°C (RCP6.0); 3) 'high' – mean increase to baseline air temperatures of 2.26°C (RCP8.5), and; 4) 'hist' – mean air temperature 0.61°C cooler than the model baseline period (2010–2020), corresponding to historical conditions during 1971–1990. 38

Figure 16. Simulated values of each component of the three-parameter Trophic Level Index (TLI_3) and combined TLI_3 for the baseline simulation and the four climate change (cc) sensitivity scenarios: 1) 'low' – mean increase to baseline air temperatures of 0.26°C (RCP2.6); 2) 'medium' – mean increase to baseline air temperatures of 1.31°C (RCP6.0); 3) 'high' – mean increase to baseline air temperatures of 2.26°C (RCP8.5), and; 4) 'hist' – mean air temperature 0.61°C cooler than the model baseline period (2010–2020), corresponding to historical conditions during 1971–1990. 39

Figure 17. Sediment release sensitivity results: plot shows nutrient loading scenario results (TLI_3); however, maximum potential sediment nutrient release rates have not been adjusted by the nutrient load factors. Thus, comparing these results with Figure 13 shows the sensitivity of scenario results to whether or not maximum sediment nutrient release rate is varied when configuring scenarios. Each dot represents a nutrient loading scenario comprising changes to the manageable portions of the external nitrogen and phosphorus loads. The interpolated coloured background represents the average three-parameter Trophic Level Index for a 10-year model simulation (2011–2020). The blue isobar represents the TLI target of 2.6 (note the target is set using a four-parameter TLI). See Appendix D for similar plots for each TLI component. 40

List of Appendices

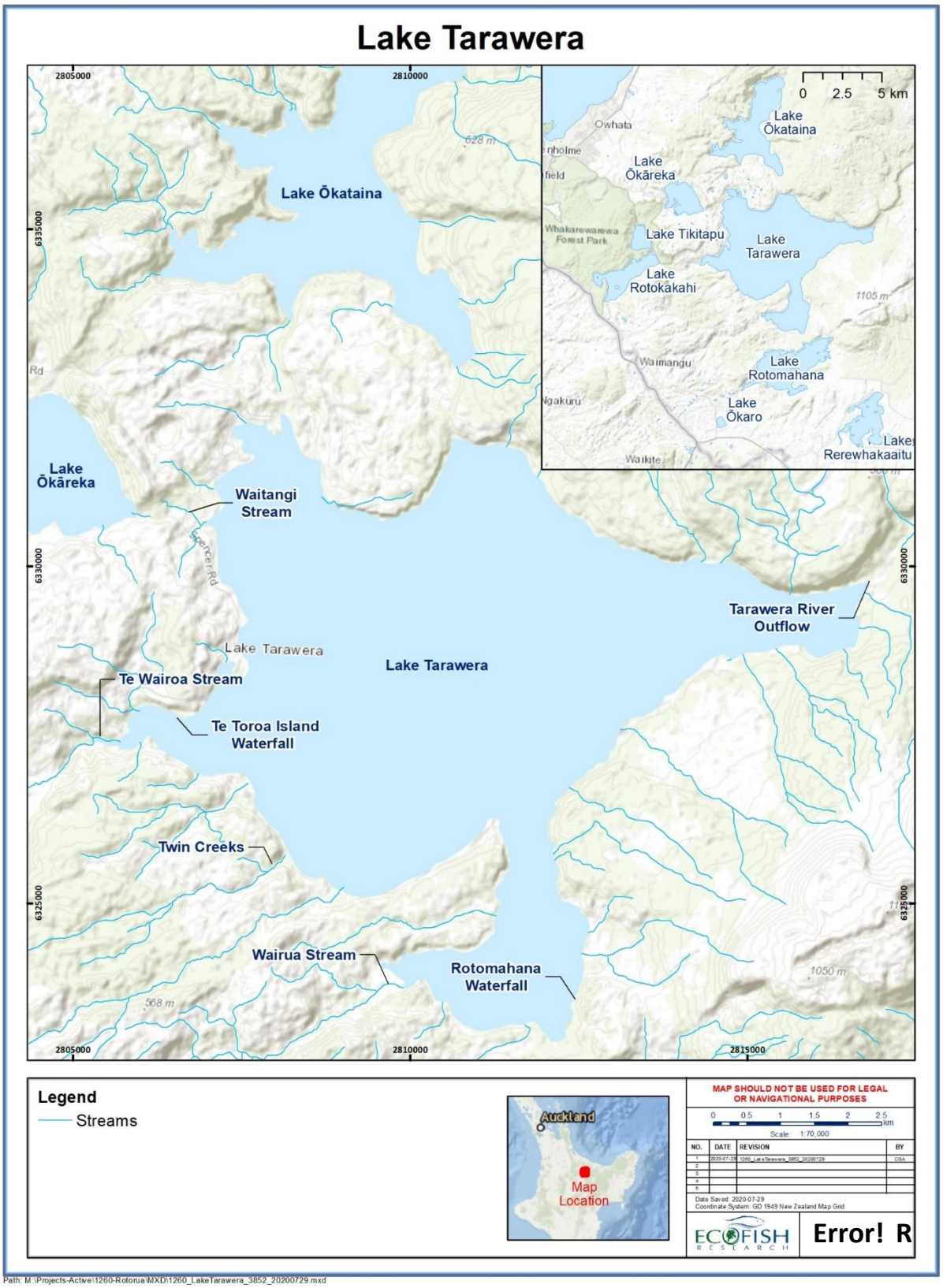
Appendix A	Inflow Water Quality Data
Appendix B	Model Configuration Details
Appendix C	Calibrated Parameter Values and Model Performance
Appendix D	Detailed Scenario Results

Introduction

Lake Tarawera (**Error! Reference source not found.**) is highly valued by tangata whenua, local residents a nd the regional community. Monitoring shows that lake water quality does not meet the trophic level index (TLI) target for the lake of 2.6, as defined in the Bay of Plenty Regional Natural Resources Plan (BoPRC 2008a). This target was based on observed water quality in 1994 (BoPRC 2008a). The target is also prescribed in the Tarawera River Catchment Plan, which requires water in the lake to be "managed in its Natural State" with "No increase in nitrogen and phosphorus levels relative to the total nitrogen and total phosphorus levels measured in the lake in 1994" (BoPRC 2004)

In 2015, the Tarawera Lakes Restoration Working Party developed the Tarawera Lakes Restoration Plan, which recognised a need to manage nutrient loads to the lake to improve water quality (BoPRC 2015). Managing nutrient loads to the lake is necessary to achieve desired lake water quality. This plan specified interim reduction targets for nitrogen (N) and phosphorus (P) that were considered "the absolute minimum required to improve water quality". The restoration plan highlighted a need to undertake further research to refine these nutrient load reduction targets based on identifying sustainable nutrient loads, which are the loads necessary to achieve the TLI target.

Bay of Plenty Regional Council (BoPRC) engaged the Environmental Research Institute, University of Waikato to undertake the Lake Tarawera Modelling Project to support decision making regarding lake and catchment management. This report describes the development and application of a lake model to estimate sustainable nutrient loads to Lake Tarawera.



Map 1. Lake Tarawera inner catchment. Inset shows seven lakes in the wider catchment that drain into Lake Tarawera.

Map 2

Background and Objectives

Lake and Catchment Description

Lake Tarawera is situated in the in the Taupo Volcanic Zone, 12 km to the south-east of the city of Rotorua, Bay of Plenty (Map 1). Three core iwi and hapū identify Lake Tarawera as within their rohe: Te Arawa, Tūhorangi Tribal Authority and Ngāti Rangitihii (BoPRC 2015). Te Arawa own the lake bed.

Lake Tarawera is a large and deep lake (Table 1). The lake catchment has complex volcanic geology, which includes large aquifers that exert a major control on fluxes of water and nutrients to the lake (Tschritter and White 2014). The lake is monomictic and therefore remains vertically mixed through the winter (July through September) and thermally stratifies in spring to autumn. The lake has high cultural and recreational values and provides a high-quality trophy trout fishery. Based on surveys of Submerged Plant Indicators (Burton and Clayton 2015), Lake Tarawera has a stable community of macrophytes that is in moderate ecological condition, although the submerged vegetation community is dominated by hornwort (*Ceratophyllum demersum*), an invasive species. This species colonised Lake Tarawera between 1988 and 2005, contributing to a decline in the ecological condition of the macrophyte community (Burton and Clayton 2015).

Table 1. Morphometric characteristics of Lake Tarawera (BoPRC 2015).

Variable	Value
Lake area	41.4 km^2
Catchment area (inner)	144.7 km^2
Mean depth	50 m
Maximum depth	88 m
Elevation	298 m a.s.l.

Lake Tarawera has an 'inner' (surface) catchment of 144.7 km^2 that comprises the area that drains directly into the lake via surface or groundwater. The inner catchment predominantly consists of native forest (62%), with smaller areas of pasture (mixed sheep/beef/deer/lifestyle; 19%) and exotic plantation forest (15%) (BoPRC 2015). There are 391 houses around the lakeshore, of which 75% are only used intermittently as holiday accommodation (BoPRC 2015).

Lake Tarawera also has an 'outer' or 'greater' catchment that includes lakes that drain into Lake Tarawera. There are seven lakes in the wider catchment: Ōkataina, Ōkāreka, Tikitapu, Rotokakahi, Rotomahana, Rerewhakaaitu and Ōkaro (see inset in Map 1). As described in Dada et al. (2016), lakes Rotokakahi and Ōkāreka flow to Tarawera via the Te Wairoa and Waitangi streams[4], respectively, and lakes Tikitapu, Ōkataina and Rotomahana are connected to Lake Tarawera by subsurface flow paths. There is also an overflow control structure that permits surface flow from Rotomahana to Lake

[4] BoPRC manages the flow from Lake Ōkāreka to Lake Tarawera by pipeline and valve.

Tarawera when water level at Rotomahana is high. The maximum flow through the structure is limited to 62,200 m^3/d with a maximum rate of 0.72 m^3/s, although this flow is not monitored regularly and there is no clear relationship between Rotomahana water level and the volume of this discharge. Lakes Ōkaro and Rerewhakakaaitu drain into Rotomahana and are therefore indirectly connected to Lake Tarawera.

Lake Water Quality

Since 2000, BoPRC has sampled Lake Tarawera water quality each month at a central lake site ('Site 5'), while NIWA also samples the Tarawera River immediately (<1 km) downstream of the lake outlet as part of the National River Water Quality Network (NRWQN) monitoring programme. Lake sampling occurred less consistently prior to 2000, as part of other programmes. The current programme includes monitoring indicators of eutrophication, which is caused by excessive inputs of nutrients that promote growth of plants, including both phytoplankton (microscopic plants suspended in the water column) and macrophytes (larger aquatic plants). The primary nutrients of concern are nitrogen and phosphorus. Symptoms of eutrophication include reduced water clarity; depletion of dissolved oxygen in bottom waters, particularly during stratification; unsightly blooms of cyanobacteria that may produce toxins and/or odours, and; extirpation of species requiring less productive waters (Carpenter *et al.* 1998).

The primary metric used by BoPRC to monitor trophic status is the Trophic Level Index (TLI; Table 2), which integrates annual mean measurements of Secchi depth and concentrations of total nitrogen (TN), total phosphorus (TP) and chlorophyll *a* (chl *a*) (Burns *et al.* 1999). Since 1991, annual TLI values have ranged from 2.6 to 3.0 for years with complete data, generally corresponding to the high end of the oligotrophic range (Figure 1). These conditions correspond to an exceedance of the current TLI target of 2.6 in the Bay of Plenty Regional Natural Resources Plan (BoPRC 2008b & subsequent amendments), which is based on a three-year average TLI value and approximates conditions that occurred in 1994 (McIntosh 2012). Since 2010 (the modelling period used in this study), annual TLI values have ranged from 2.6 to 2.9, and there is no clear trend over the 30-year period of record (Figure 1). Values for individual TLI components show that values for the TN component (TL_N) are relatively low, while values for the TP component (TL_P) are relatively high (see coloured dots in Figure 1).

Measurements used to calculate the TLI values in Figure 1 are shown in Figure 2. Measurements of all four TLI components were collected in the lake by BoPRC. Nutrient chemistry data collected in the lake that are used for this study have been adjusted using equations developed by McBride and Baisden (2019) to reduce suspected error due to analytical interference related to the geothermally-influenced water chemistry, as evident from inconsistent results among laboratories (McBride and Baisden 2019). For context, TN and TP measurements are shown in Figure 3, compared with paired values of measurements collected by NIWA at the NRWQN site at the lake outlet. Values seem to match most closely after 2010 (the period considered in this study), particularly for TN. Some TP measurements collected by BoPRC are higher than paired measurements collected by NIWA; these values may be biased high, reflecting that adjustments to the data have not fully corrected for the suspected analytical interferences. Nonetheless, the full adjusted BoPRC dataset was used in this study

to quantify model performance, and this approach ensures consistency with the dataset that is currently used by BoPRC to evaluate lake water quality. Nitrogen availability exerts an important control on phytoplankton biomass accumulation in Lake Tarawera. A study in 1982 showed that the lake was nitrogen limited in winter, with nutrients considered not limiting in the other three seasons (White *et al.* 1985). The lake was then shown to be primarily nitrogen limited again in summer 1985 (White *et al.* 1986). Similarly, bioassay experiments in 2004–2005 showed primary nitrogen limitation was present during stratified and unstratified periods, with nitrogen fixation by cyanobacteria also measured in summer and autumn (Baldwin 2009). While these historical studies have shown the presence of primary nitrogen limitation in the lake, changes to phosphorus loads also have potential to affect lake water quality; it is notable that several of the bioassay experiments reported in the studies cited above showed a higher increase in phytoplankton biomass when nitrogen and phosphorus were added in combination, compared with adding nitrogen alone (e.g., Baldwin 2009). Since 2000, the mean ratio of TN to TP concentrations (TN:TP) is approximately 12. Such low to moderate TN:TP values are commonly a feature of lakes where both nitrogen and phosphorus control phytoplankton biomass accumulation (e.g. White 1983).

Table 2. Overview of the Trophic Level Index (TLI) classification scheme (Burns et al. 1999).

Trophic state	Trophic Level Index	Productivity	Perceived water quality
Ultra–microtrophic	0–1	Very low	Excellent
Microtrophic	1–2	↓	↓
Oligotrophic	2–3		
Mesotrophic	3–4		
Eutrophic	4–5		
Supertrophic	5–6		
Hypertrophic	6–7	Very high	Very poor

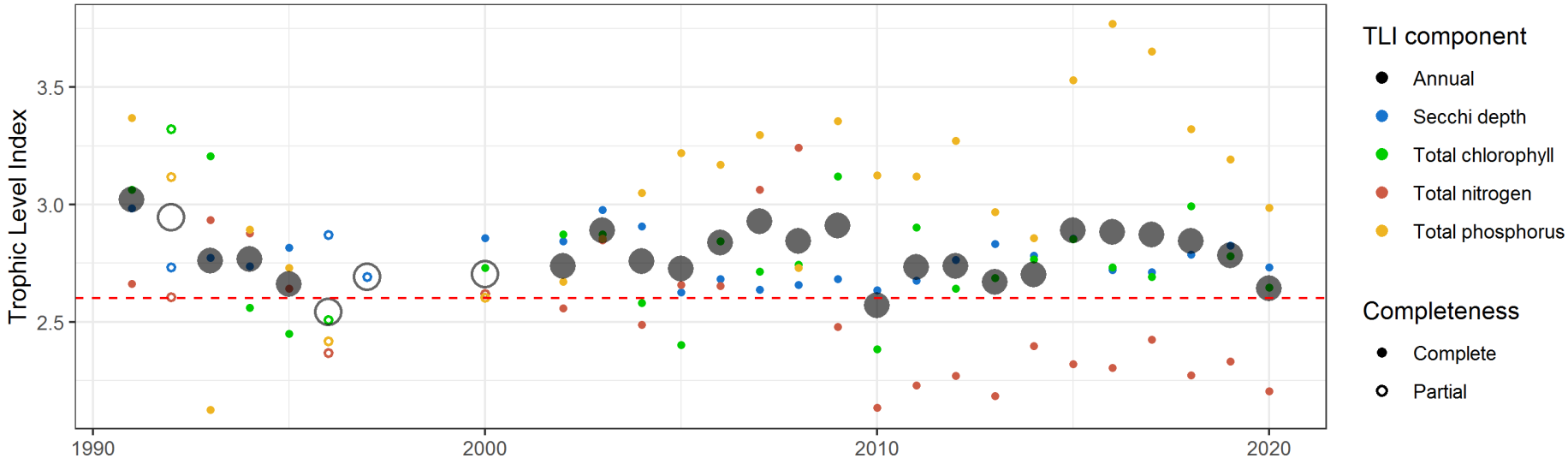


Figure 1. Annual Trophic Level Index (TLI) values for Lake Tarawera.

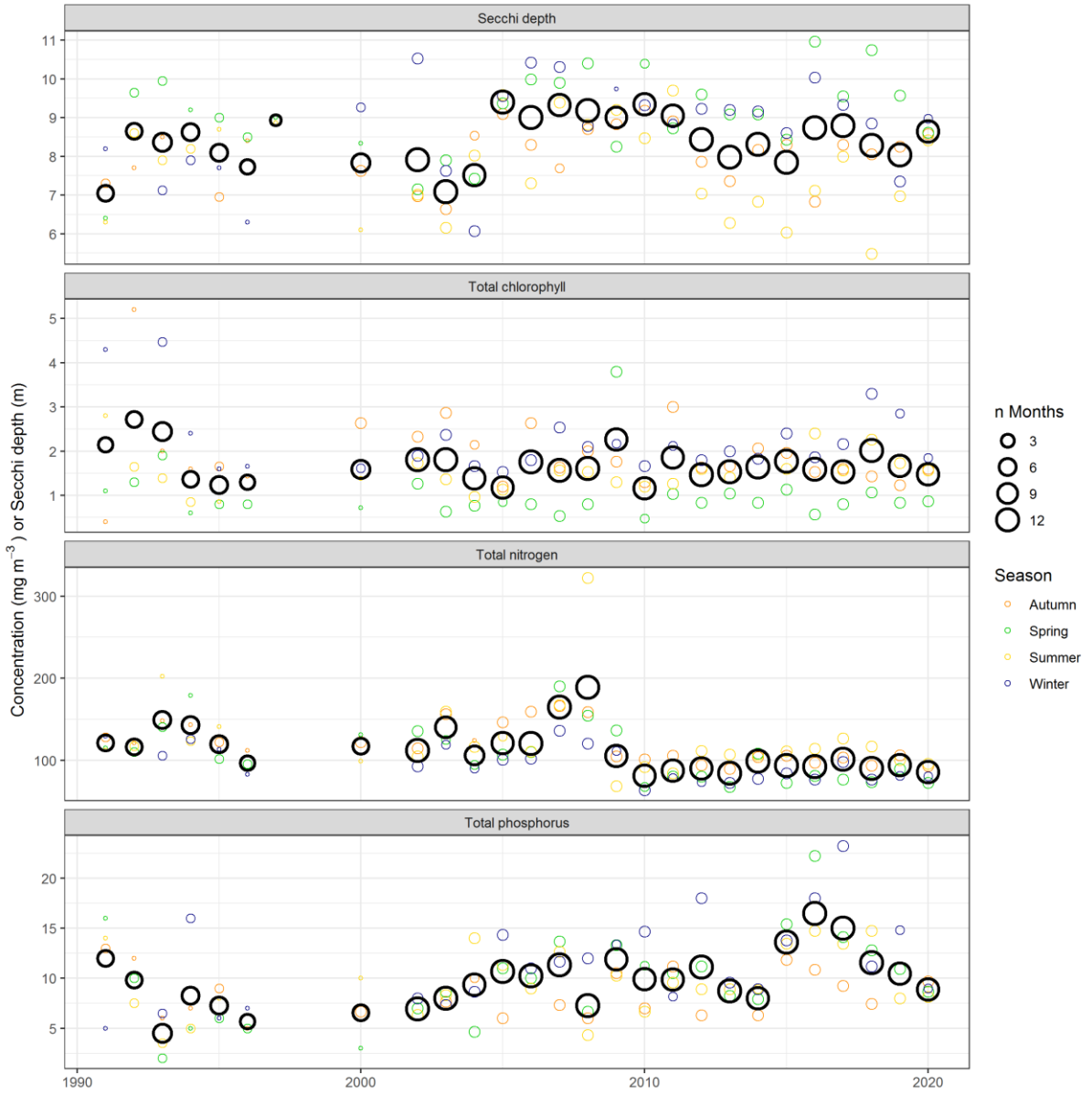


Figure 2. Lake Tarawera water quality data. Total nitrogen and phosphorus concentrations were measured by NIWA at the lake outflow; chlorophyll *a* and Secchi depth measurements were collected in the lake by BoPRC.

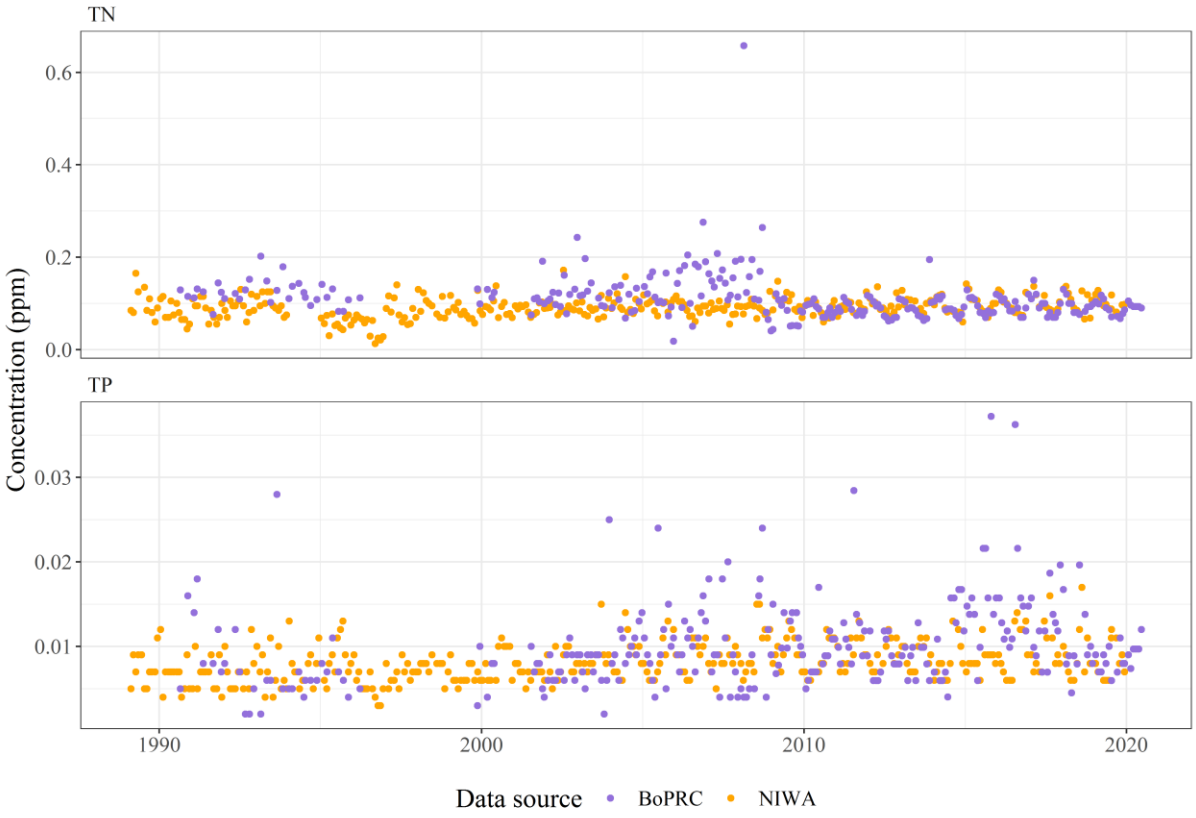


Figure 3. Comparison of paired concentrations of total nitrogen (TN) and total phosphorus (TP) measured by Bay of Plenty Regional Council (BoPRC) in Lake Tarawera surface water, and by NIWA at the lake outlet. BoPRC measurements have been adjusted based on McBride and Baisden (2019) to reduce suspected laboratory error.

Previous Monitoring and Research

This lake modelling study drew on the results of several previous monitoring and research programmes:

Groundwater monitoring/modelling: GNS Science [5] led a series of detailed studies to improve understanding of groundwater interactions with Lake Tarawera (Bruere and White 2016). These studies have been part of a three-phase programme: 1) a drilling programme to measure aquifer properties; 2) developing a geological model to identify key aquifers (Tschritter and White 2014), and; 3) developing a groundwater flow model. The groundwater flow model provides steady-state predictions of fluxes of water and nitrate in the groundwater to the lake, in addition to lakes and streams within the greater catchment (White *et al.* 2016).

Inflow water quality sampling: Since 2006, Mr T. Beckett has led a programme to monitor nutrient concentrations in lake inflows, with sampling also undertaken by BoPRC of the Waitangi Stream. Monitoring has been undertaken at quarterly frequency with laboratory analysis conducted at the

[5] The Institute of Geological and Nuclear Sciences Limited

University of Waikato, BoPRC's laboratory and Hill Laboratories. Discharge has also been measured at a subset of inflows as part of this programme. Historical lake inflow data are summarised in Appendix A.

Tarawera River (outlet) sampling: As described above, NIWA samples the Tarawera River immediately (<1 km) downstream of the lake outlet as part of the NRWQN monitoring programme. NRWQN methods are described in Smith and McBride (1990) and Smith et al. (1996).

Monthly lake water quality sampling: BoPRC has sampled Lake Tarawera each month since 2000 at a central lake site to measure a standard suite of limnological variables. Methods are generally consistent with Burns et al. (2000). A surface integrated sample has typically been collected to a depth of 17 m, although this depth has varied since 2016 (depths of 8 m, 10 m and 14 m are reported in files provided by BoPRC). Samples have also consistently been collected at a depth of 60 m, while near-bottom samples have also been intermittently collected from a depth of 81 m.

As noted in the

Lake Water Quality section above, some nutrient concentration data collected during this programme have low accuracy, likely due to analytical interference caused by the unusual chemistry of Lake Tarawera (high silica and arsenate). Since 2010, improvements in nitrogen measurement methods have seemingly addressed a prior issue with erroneously high values, although unadjusted phosphorus concentrations measured since 2010 are still prone to inaccuracy (McBride and Baisden 2019). In consultation with BoPRC's Lake Water Quality Technical Advisory Group, the University of Waikato led a review of BoPRC's lake water quality data to evaluate the potential to apply adjustments to the existing data to improve accuracy. This included reviewing interlaboratory comparisons of measurements of dissolved and total phosphorus in Lake Tarawera water samples that were collected from July 2017 to January 2018 and analysed at five laboratories (McBride and Baisden 2019). Based on this review, regression equations were developed to improve the accuracy of historical phosphorus measurements. As described above, the adjusted BoPRC dataset was used to prepare figures and quantify model performance for this report, although NIWA's NRWQN nitrogen and phosphorus data collected at the lake outlet were considered in addition to the BoPRC dataset during model calibration as part of a diligent approach to model development.

Lake monitoring buoy: The University of Waikato maintains a buoy that has monitored lake water quality since 2009. The buoy includes a climate station and measures water temperature at a range of depths. Monitoring has been intermittent at the site due to challenges posed by the exposed and deep nature of the site.

Nutrient budget studies: Multiple nutrient budgets studies of Lake Tarawera have been undertaken that provide information about estimated fluxes of water and nutrients to the lake (Hamilton *et al.* 2006; McIntosh 2012; Dada *et al.* 2016; McBride *et al.* 2020). Most recently, McBride *et al*. (2020) estimated external loads to 13 Te Arawa lakes including Lake Tarawera. The authors used a modified export coefficient approach that was informed by BoPRC's Overseer® modelling outputs. External loads estimated in that study (106.8 t N/y and 10.7 t P/y) provide a valuable reference point to compare to loads that are estimated in this study using a more empirical approach based on available field data.

Lake Nutrient Loads and Targets

Previous nutrient budget studies have shown that the majority of the TP load, and a major component of the TN load to the lake, are derived from geothermal inflows and other lake catchments in the greater watershed (Hamilton *et al.* 2006; Dada *et al.* 2016). Within the inner catchment, important nutrient sources that are manageable include farmland and wastewater from septic tanks (Dada *et al.* 2016).

Failure to meet the TLI target resulted in establishment of the Tarawera Lakes Restoration Working Party and development of the Tarawera Lakes Restoration Plan (BoPRC 2015). This plan identified a need to control loads of nitrogen and phosphorus to the lake to achieve desirable water quality and contribute to achieving the TLI target. The restoration plan recognised that further research was required to calculate the current nutrient loads to the lake, and to estimate 'sustainable' nutrient loads, which correspond to the loads of nitrogen and phosphorus that would result in achieving the TLI target. The differences between current loads and sustainable loads may therefore represent viable nutrient load reduction targets to achieve desired water quality. While further research is being undertaken, the restoration plan identified interim targets: 1) to ensure that no increase to annual TN loads occurs, and; 2) to reduce the annual TP load to the lake by at least 1,200 kg.

Since the restoration plan was developed, research has been undertaken to refine estimates of current nutrient loads to Lake Tarawera, as well as to improve the accuracy of lake water quality data (see

Previous Monitoring and Research section). However, to estimate sustainable loads, it is necessary to predict how changing nutrient loads will affect lake water quality. Lake ecosystem modelling provides a tool that can support identification of sustainable nutrient loads, and has been used to inform management of other Te Arawa lakes (e.g., Hamilton *et al.* 2012, 2015).

Study Objectives

The objectives of this study are to:

1. Develop a one-dimensional (1-D) lake model for Lake Tarawera that simulates historical water quality.
2. Simulate a range of nutrient loading scenarios expressed as changes in TN and TP loads to the lake.
3. Analyse scenario results to estimate sustainable nutrient loads of TN and TP that will comply with the TLI target for the lake. Use these results to estimate the differences between current loads and sustainable loads to define potential nutrient reduction targets.
4. Evaluate monitoring data and modelling results to identify key sources of uncertainty that could be addressed to better understand how catchment management can affect lake water quality.

The remainder of this report describes the methods, results and conclusions of the lake modelling study.

Methods

Model Selection

The 1–D model DYRESM–CAEDYM was used for this study. DYRESM–CAEDYM is the most widely–cited aquatic ecosystem model in the scientific literature (Trolle *et al.* 2012). The model has been applied to several lakes in New Zealand, including Rotorua (Burger *et al.* 2008; McBride *et al.* 2019), Lake Ōkaro (Özkundakci *et al.* 2011), Rotoehu, Lake Ōkareka and Te Waihora/Lake Ellesmere (Trolle *et al.* 2011), as well as dozens of other lakes globally. Selecting DYRESM–CAEDYM meant that this study could benefit from the extensive body of previous work that has been undertaken to configure and calibrate the model in similar lakes.

The model comprises a hydrodynamic model (DYRESM[6]) that is coupled to a water quality model (CAEDYM[7]). DYRESM predicts the vertical variations of temperature and density in lakes such as Lake Tarawera that have relatively simple morphometry and satisfy the 1–D assumption. CAEDYM can be used to model a wide range of biogeochemical state variables such as nutrient concentrations and phytoplankton abundance. The models are process–based and are thus primarily based on representations of functional (rather than empirical) relationships between state variables. Both DYRESM and CAEDYM were developed at the Centre for Water Research (CWR) in Western Australia. Details of the model conceptualisations and equations are available in the 'science manuals' (Hipsey and Hamilton 2008; Imerito 2015)

Such process–based modelling allows a wide range of variables to be simulated at high temporal resolution to provide detailed understanding of major processes in the lake. The use of process–based models provides greater certainty in the results for scenarios that differ from the current state, compared to the use of empirical (i.e., statistical) relationships, which are generally invalid outside the bounds of the data used to develop the model. A constraint of such process–based models, however, is that they are 'data hungry': they require information for a large number of forcing variables including meteorology, hydrology and catchment water chemistry variables, in addition to field measurements of simulated variables to assist model calibration. The extent of data available for Lake Tarawera was considered adequate to develop a satisfactory model, particularly in light of recent groundwater modelling GNS (White *et al.* 2016). Nonetheless, there are some notable data gaps – uncertainties associated with these gaps, in addition to options to address them, are described in the Uncertainties section.

Model Overview

DYRESM simulates multiple horizontal water column layers of variable thickness that change dynamically to accommodate changes in lake volume and vertical density gradients. DYRESM is primarily affected by surface exchanges of heat, mass and momentum, and resolves the vertical

[6] DYnamic REservoir Simulation Model

[7] Computational Aquatic Ecosystem Dynamics Model

distributions of temperature, salinity and density in lakes and reservoirs (Hamilton and Schladow 1997).

CAEDYM simulates fluxes that regulate biogeochemical variables such as nutrient concentrations and phytoplankton biomass (Figure 4). The model includes representations of cycling processes for carbon, nitrogen, phosphorus, dissolved oxygen and inorganic suspended sediments (Hipsey *et al*. 2013). The state variables that are simulated within CAEDYM can be adjusted depending on the study objectives and the availability of measured data for calibration. The following three generic groups of phytoplankton were represented in CAEDYM for this study: freshwater diatoms, chlorophytes and cyanobacteria. Phytoplankton growth depends on nutrient availability and temperature. For each model time step, growth rate (μ; d^{-1}) for each phytoplankton group was estimated with CAEDYM as[8]:

$$\mu = \mu_{max} \times min[f(I), f(N), f(P), f(Si)] \times f_{T1}(T)$$

where μ_{max} (d^{-1}) is maximum growth rate at 20 °C; f(I), f(N), f(P) and f(Si) represent limitation by light, nitrogen, phosphorus and silica (diatoms only) respectively, and; $f_{T1}(T)$ is a temperature function which allows the maximum growth rate at temperature of T_{opt} and prevents growth at temperature > T_{max}. Nutrient limitation was represented using the Droop (1974) equations, which allow for dynamic internal nutrient (N and P) stores that vary within the bounds of assigned minimum and maximum values. Phytoplankton nutrient uptake is a function of ambient concentrations and assigned maximum uptake rates. Nutrient limitation depends on internal stores, with only a single nutrient potentially limiting growth at any one time. Nitrogen fixation was simulated in cyanobacteria, as described in the Results and Discussion.

Sediment nutrient release (DRP and NH_4-N) was simulated, with fluxes related inversely to dissolved oxygen concentrations. Sediment resuspension was not simulated, reflecting the generally deep and steep-sided nature of Lake Tarawera.

Higher-level fauna and macrophytes were not considered. In particular, this meant that zooplankton grazing was not simulated. This omission was justified on the basis that zooplankton grazers are generally depauperate in New Zealand lakes (e.g., Bayly 1995), no zooplankton data were available, and the effect of zooplankton grazing could be partly implicitly represented by calibrating parameters that control other processes that reduce phytoplankton biomass accumulation (respiration and settling).

The model was configured using daily data relating to hydrology (Figure 5), meteorology and water chemistry of 12 inflows. Details are provided in Appendix B about model configuration, including details of the water balance and estimation of external nutrient loads.

[8] From Equation 6.1 in Hipsey *et al*. (2013)

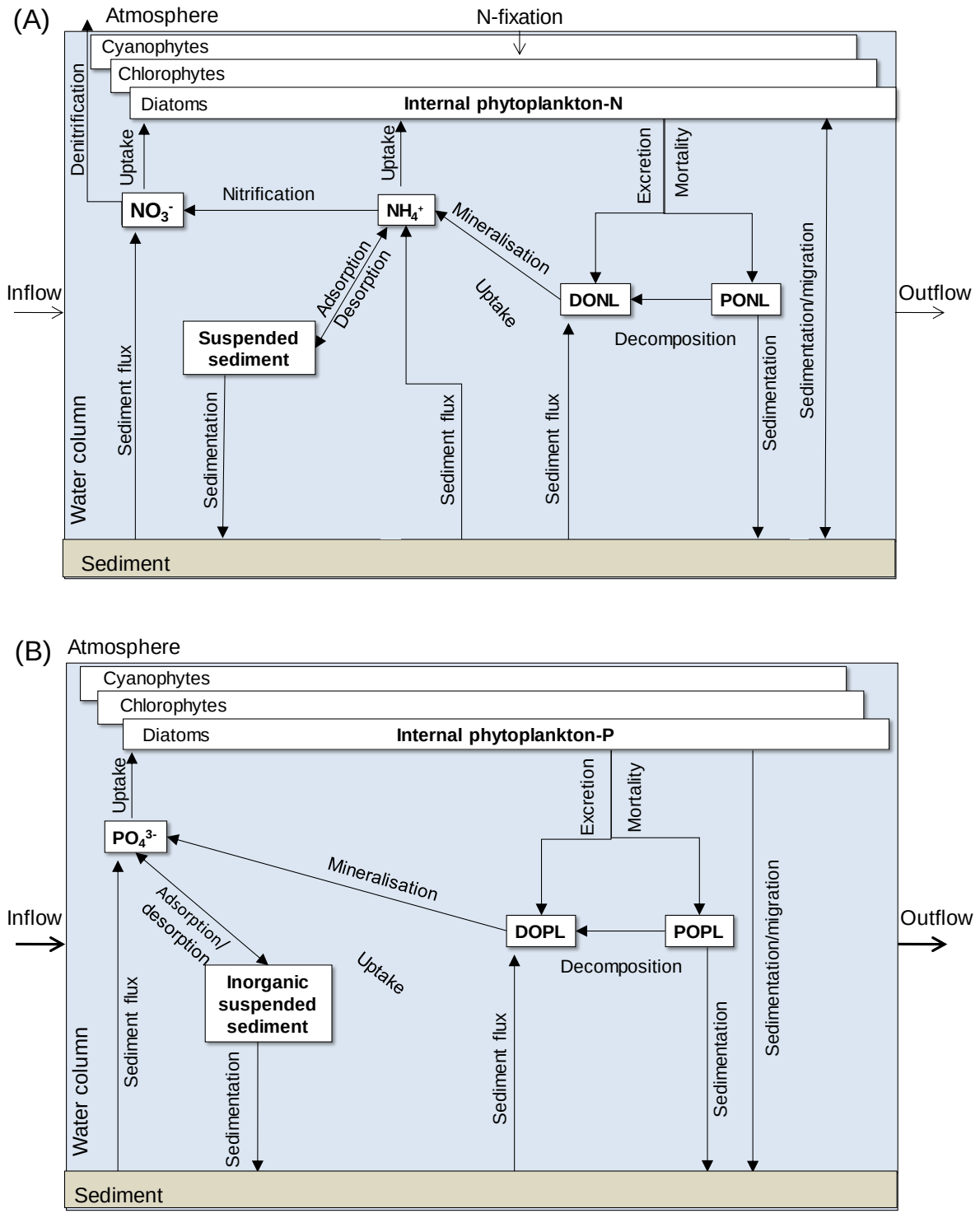


Figure 4. Conceptual diagrams of the cycling of nitrogen (A) and phosphorus (B) within the water quality model (CAEDYM). DONL, labile dissolved organic nitrogen; PONL, labile particulate organic nitrogen; DOPL, labile dissolved organic phosphorus; POPL, labile particulate organic phosphorus.

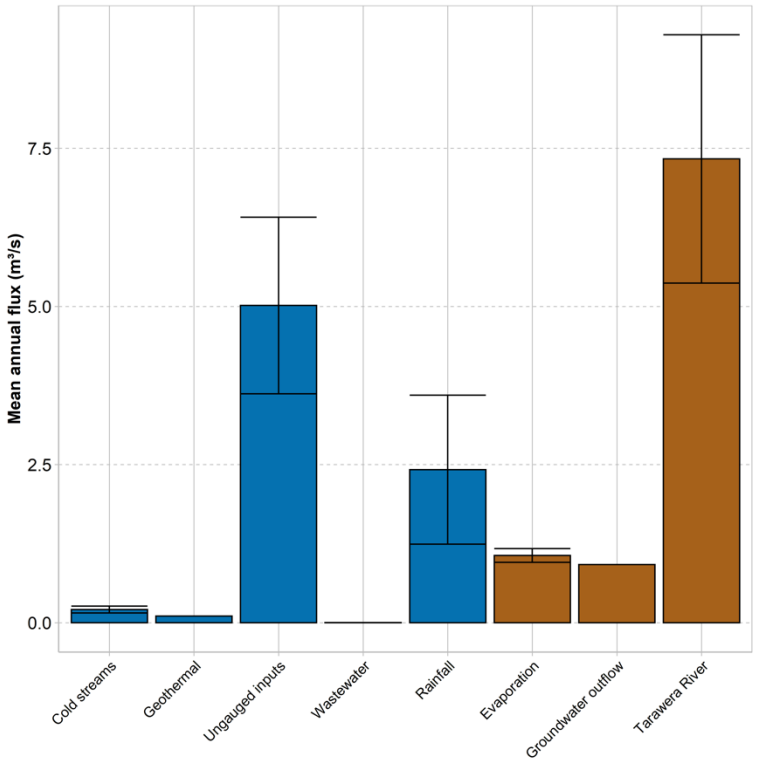


Figure 5. Summary of inflows (blue) and outflows (yellow) included in the lake model. Vertical lines show ±1 standard deviation. See Appendix B for details of all inflows (n=12) included in the model.

Model Application

A ten-year baseline period of 2010–2020 was simulated. Results were analysed based on "TLI years", not calendar years, to reflect that the annual TLI is calculated using data collected between 1 July and 30 June. Therefore, the simulation period spanned from 1 July 2010 to 30 June 2020.

It was desirable to select a baseline period that spanned multiple years to encompass a range of background environmental conditions (meteorology and hydrology) that can affect lake water quality. The calibration period was 2010–2015 and the validation period was 2015–2020. The model was calibrated by adjusting key parameter values to optimise model performance statistics (see below) for the calibration period, with simulations typically run for short periods (2–3 years) during the calibration period to refine parameter values. Values were adjusted within the range of literature values, with particular reference to values used in other modelling studies of Te Arawa lakes.

To effectively examine the parameter space, an automated calibration procedure was developed using the R statistical computing environment (R Core Team 2020). This procedure allowed multiple model simulations to be run with different sets of parameter values that were randomly modified within user-defined ranges. Model performance was then compared among multiple simulations based on evaluating the Pearson correlation coefficient (a measure of how well the model reproduced observed trends) and the normalised root-mean-square-error (a measure of how well the model reproduced the magnitude of observations). The automated calibration procedure followed a sequential approach, with physical processes first calibrated by evaluating error in water temperature

predictions after adjusting DYRESM parameters and applying minor adjustment factors (typically ±5%) to meteorological variables. Model simulation of bottom water dissolved oxygen and nutrient concentrations was then calibrated, followed by surface water variables including nutrients and chlorophyll *a*. Manual calibration was then necessary to further refine parameter values.

Model performance for the calibration and validation periods was quantified by comparing modelled and measured values of the following variables: temperature, dissolved oxygen, nutrients (nitrogen and phosphorus fractions) and chl *a*. Comparisons were made with monthly measurements collected by BoPRC at a central lake site ('Site 5') and at the lake buoy (temperature only). Model results were compared with results for surface integrated samples (generally 0–17 m depth) and samples collected from within a depth range of 60–65 m, as well as 80–85 m. Model performance was evaluated based on visual inspection of time series graphs to evaluate trends, and was assessed quantitatively by calculating mean error, mean absolute error and root-mean-square-error, as well as normalised versions of these statistics that were reduced to a common scale by dividing by the mean of observations (see Bennett *et al.* 2013 for detailed methods for calculating performance statistics).

The model was run with a time step of one hour, with results output at daily frequency. Water quality parameters were initialised based on the most recent monitoring data that corresponded to the start date. Version 3.1.0 of DYRESM-CAEDYM was used.

Model Scenarios

Multiple scenarios of varying nutrient loads were simulated with the aim of identifying 'sustainable' nutrient loads corresponding to the loads of nitrogen and phosphorus that would result in achieving the TLI target of 2.6. Scenarios were configured by adjusting the 'manageable' portions of the external TN and TP loads to the lake by factors between 0.6 and 1.2. The manageable loads were considered to be the total loads, minus atmospheric deposition and geothermal inputs. The factors were applied to concentrations of nitrogen and phosphorus fractions in the inflow files, with the volume (discharge) of each inflow left unchanged. This approach resulted in changing the average annual loads for the model simulation period by the respective factors – note that changes to individual annual loads in each scenario were variable as the discharge of each inflow varied among years. Further, maximum sediment oxygen consumption and nutrient release rates were also modified by the respective factors to reflect assumed changes in internal loads. In total, 36 nutrient load scenarios were identified, as summarised in a 6 × 6 matrix (Figure 6).

All other forcing data were unchanged during scenario analysis. For each scenario, annual TLI_3 values were calculated based on modelled chl *a*, TN and TP concentrations averaged across depths from 0–17 m, which corresponds to the depth range that is typically sampled by BoPRC to collect an integrated surface sample (see

Previous Monitoring and Research).

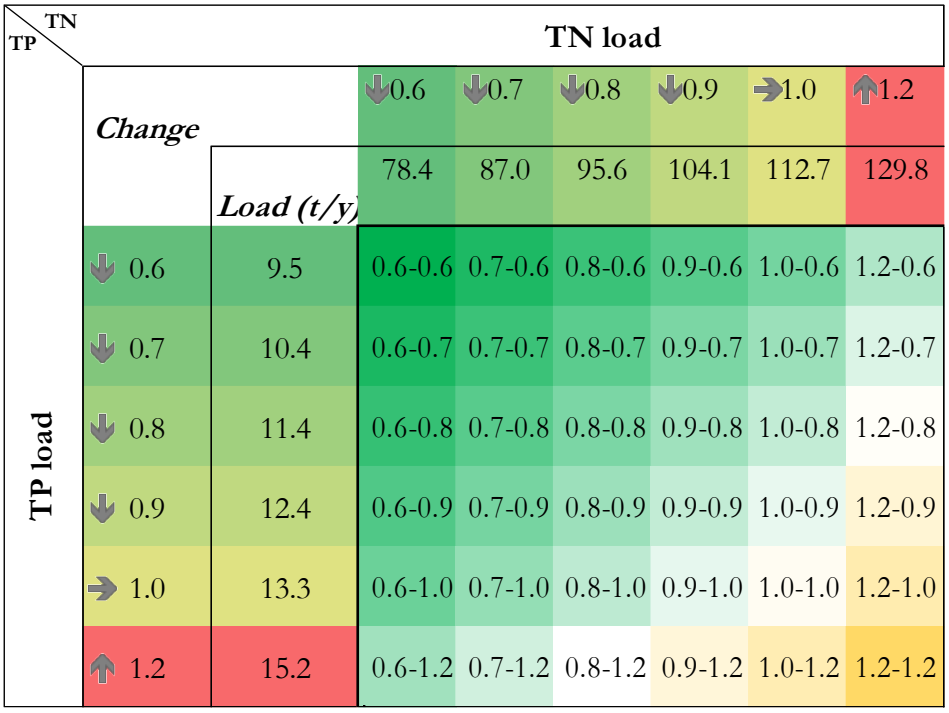


Figure 6. Nutrient load scenarios.

TLI3 vs TLI

The model was used to calculate TLI_3, which is a three-parameter version of the TLI metric that is calculated without Secchi depth data. This is because the lake model does not explicitly simulate Secchi depth. This potentially causes inconsistency when making comparisons to the TLI target for the lake (2.6), as this is based on the standard four-parameter TLI. To examine the potential error associated with this uncertainty, measured values of TLI and TLI_3 (2000—2018) were compared: this comparison showed that paired values of the two metrics were very similar (Figure 7)[9]. On average, paired values were only 1.1% different, with absolute differences ranging from 0.006 to 0.071 units. These differences are much smaller than model error. Accordingly, model predictions in TLI_3 units are compared directly with the TLI target, with no correction deemed necessary.

[9] This analysis was completed using NIWA's NRWQN data. The conclusion from this analysis is not expected to change if were to be completed with BoPRC's adjusted lake water quality dataset (unavailable when this analysis was completed).

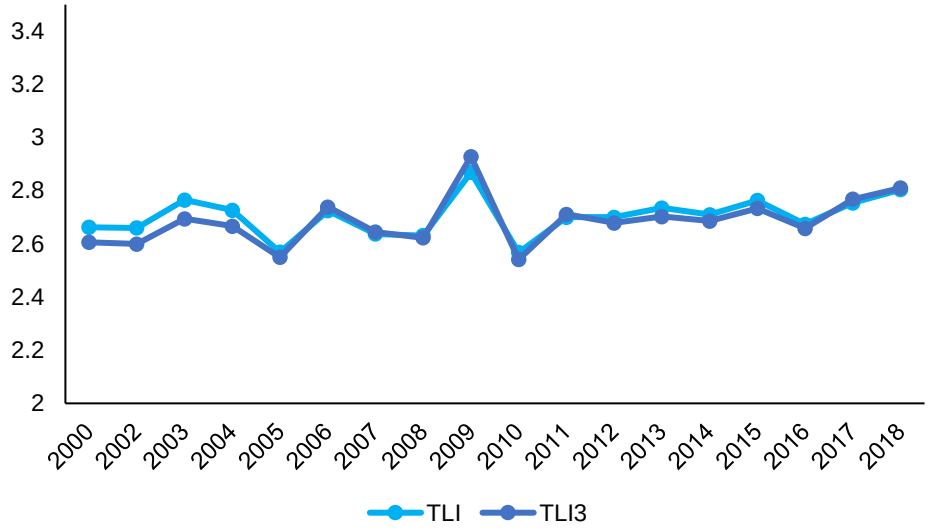


Figure 7. Measured values of TLI and TLI_3.

Sensitivity Analysis

Two sets of sensitivity analyses were completed to examine how model results could be affected by climate change and changes to nutrient releases from lakebed sediments.

Climate Change

To examine the sensitivity of model results to climate change, climate change sensitivity scenarios were developed based on evaluating NIWA (2016) climate change projections and using the midpoint of Tauranga and Taupō as a good representation of Lake Tarawera's climate for 2020–2100. A lack of variation was identified for precipitation and other parameters, so only air temperature changes were considered using the following three climate change scenarios used by the Intergovernmental Panel on Climate Change (IPCC):

- RCP2.6: this corresponds to a "stringent mitigation" scenario and is the least extreme of four key scenarios used by climate scientists to project changes to climate. This IPCC scenario assumes net negative greenhouse gas emissions by 2100 and is considered likely (medium confidence) to result in global warming that remains below 2°C above pre-industrial temperatures (IPCC 2015).
- RCP6.0: this corresponds to an intermediate IPCC scenario. For this IPCC scenario, there is high confidence that global warming is likely to exceed 2°C above pre-industrial temperatures by 2100 (IPCC 2015).
- RCP8.5: this is the most extreme of four key scenarios used by climate scientists. This IPCC scenario assumes "very high" greenhouse gas emissions and there is high confidence that global warming is likely to exceed 2°C above pre-industrial temperatures by 2100 (IPCC 2015).

Projected air temperatures for all IPCC scenarios are similar until 2050, after which the projections diverge. Projections for the RCP2.6 IPCC scenario then remain relatively steady through to 2100, while projections for the RCP8.5 IPCC scenario continue to increase through to 2100. Projections for the

RCP6.0 scenario are intermediate. Based on this, four climate change sensitivity scenarios were identified:

1. Low: this scenario corresponds to the mean air temperatures for the RCP2.6 IPCC scenario for the period 2050–2100. This scenario involves a mean increase to baseline air temperatures of 0.26°C (a 0.61°C increase relative to the 1971–1990 mean).
2. Medium: this scenario corresponds to the mean air temperatures for the RCP6.0 IPCC scenario for the period 2090–2099. This scenario involves a mean increase to baseline air temperatures of 1.31°C (a 1.92°C increase relative to the 1971–1990 mean).
3. High: this scenario corresponds to the mean air temperatures for the RCP8.5 IPCC scenario for the period 2090–2099. This scenario involves a mean increase to baseline air temperatures of 2.26°C (a 2.87°C increase relative to the 1971–1990 mean).
4. Historical: this scenario corresponds to historical conditions during 1971–1990, when mean air temperature was 0.61°C cooler than the model baseline period (2010–2020). This scenario was included as a point of reference to examine how model sensitivity to projected future climate change compares with recent historical changes.

The four climate change sensitivity scenarios above were configured by applying constant offsets to daily mean air temperatures in the meteorology file. Inflow water temperatures were unchanged for these sensitivity analyses; this was justified on the basis that separate sensitivity analyses showed that model results were not sensitive to changes in inflow temperatures within plausible ranges. This reflects that surface stream inflow volumes are particularly small in Lake Tarawera (Figure 5) and therefore they have limited potential to affect the lake heat budget.

Sediment Nutrient Release

Sediment nutrient stores are conceptualised in CAEDYM to be constant through time. When configuring the nutrient load scenarios, maximum sediment nutrient release rates were adjusted by the respective external load change factors for each scenario to reflect the expectation that sediment nutrient releases (internal loads) would change in response to changes to external loads (see

Model Scenarios above). There is uncertainty regarding the assumption that changes to sediment nutrient release rates would be directly related to changes in external loads; however, this assumption was considered reasonable for Lake Tarawera – note that as an oligotrophic lake, internal loads in Lake Tarawera are smaller and less influenced by complex feedbacks with redox state than eutrophic lakes such as Rotorua. Thus, over longer time periods such as 10–15 years, benthic sediment nutrient release (internal loads) would be expected to decline in response to external load reductions as sediment nutrient stores equilibrate to lower rates of nutrient settling (Jeppesen *et al.* 2005).

Nonetheless, there is expected to be a lag before benthic sediment nutrient stores equilibrate to changes in external loads. Thus, over shorter time periods following nutrient load reductions, sediment nutrient release relates may remain elevated until benthic sediment nutrient stores decline. To evaluate the sensitivity of scenario results to this lag, sediment nutrient release sensitivity analysis was undertaken by simulating the nutrient load scenarios but leaving maximum sediment nutrient

release rates unchanged. Thus, the results of this analysis indicate expected water quality in the short term (e.g., <5 years) following changes to external nutrient loads.

Results and Discussion

Nutrient Loads

In total, estimated annual average external nutrient loads to Lake Tarawera were 112.7 t N/y and 13.03 t P/y (Figure 8, Figure 9, Table 3). The total external nitrogen load was higher than the load estimated in previous studies, whereas the total phosphorus load was towards the upper end of the range estimated by previous comparable studies (Table 3). Based on the calibrated baseline model (Appendix D), estimated average internal nitrogen load was 6.27 t N/y (5% of total loads), whereas estimated average internal phosphorus load was 2.35 t N/y (15% of total loads).

For context, the standard deviation of external loads estimated in comparable studies was 9% of the estimated external TN load and 19% of the estimated external TP load. The external nitrogen load estimated in this study was 5% (6 t N/y) higher than the load recently estimated by McBride *et al*. (2020), whereas the phosphorus load estimated in this study was 18% (2.3 t P/y) higher than the load estimated in that study. McBride *et al*. (2020) used a different method (land use export coefficient modelling) than this study, and therefore the similarity between the two independent sets of estimates provides some confidence in the accuracy of the estimates. Unlike McBride *et al*. (2020), external loads were estimated in this study using an empirical approach that used all suitable and current field data to estimate loads separately for each lake inflow. Lake nutrient budgets derived using this method are generally considered more accurate than those based on general land use export coefficients because data specific to the study system are used. However, given the high uncertainty in calculating groundwater nutrient loads (discussed in *Uncertainties* section below), confidence in the estimates developed in this study and by McBride *et al*. (2020) can be considered similar, and therefore managers should consider these two studies in combination to understand the ranges of external nutrient loads that are plausible. The study by McBride *et al*. (2020) has an advantage that estimates can be directly compared with the other Te Arawa Lakes that they studied. However, this study provides greater detail about how loads vary among sources such as stream inflows and groundwater, as well as providing estimates of internal loads, which cannot be derived from export coefficient modelling.

'Ungauged inputs' is the inflow that contributes the largest nutrient loads to the lake (Figure 8, Figure 9). This inflow is estimated to comprise ~90% groundwater and contributes an average of 50.75 t N/y and 6.69 t P/y to the lake, representing 45% of the total nitrogen load and 51% of the total phosphorus load.

Cold streams are estimated to be the second-largest source of nitrogen to the lake and the third-largest source of phosphorus (Figure 9). Geothermal inflows are estimated to be the second-largest source of phosphorus to the lake but a lesser source of nitrogen (Figure 9). Estimated mean annual nutrient loads from all cold streams were 31.9 t N/y and 2.30 t P/y, whereas estimated mean annual nutrient loads for the two geothermal inflows ('observed' and 'diffuse') combined were 2.8 t N/y and 2.39 t P/y (Figure 9). For context, the total geothermal nitrogen and phosphorus loads estimated in this study are approximately half of those presented in the Tarawera Lakes Restoration Plan (6.3 t N/y

and 5 t P/y); however, there is high uncertainty regarding this component as a substantial portion of the geothermal load is expected to enter diffusely via the lake bed and cannot be directly measured.

Atmospheric deposition to the lake surface is estimated to be an important source of nutrients (24.2 t N/y and 1.37 t P/y), contributing ~10% (phosphorus) to ~20% (nitrogen) of external nutrient loads. Estimated mean annual nutrient loads from wastewater were 2.84 t N/y and 0.284 t P/y (Figure 8, Figure 9).

For current land use under steady-state conditions, White et al. (2016) estimated that the annual nitrogen load to Lake Tarawera from the greater catchment but excluding the lake surface is 138.2 t N/year (Table 4.9 in White *et al.* 2016). Adding this value to the current estimate of nitrogen load in rainfall to the lake surface (24.9 t N/y; **Error! Reference source not found.**Figure 9) yields a TN l oad for current land use under steady-state conditions of 163.1 t N/y. This load is 45% higher than the current (non-steady-state) annual nitrogen load estimated in this study, thus indicating that there may be a major 'load to come' of ~50 t N/y to the lake as groundwater nitrogen concentrations increase to equilibrate with loads from existing land use pressures. This prediction depends on the accuracy of estimates of current and future groundwater nitrogen concentrations, which are highly uncertain.

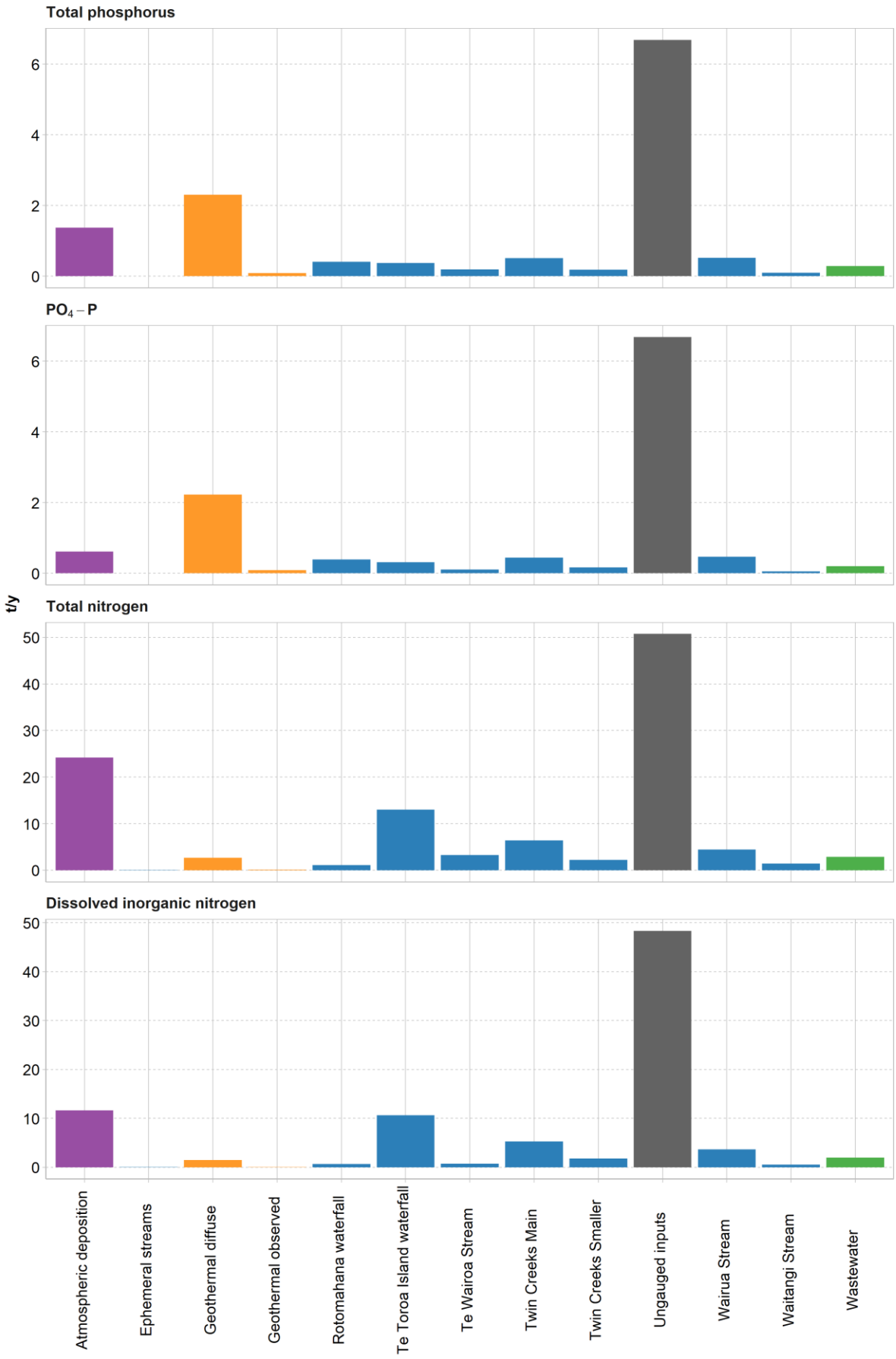


Figure 8. Mean annual nutrient loads for all inflows included in the model. See Figure 9 for a summary.

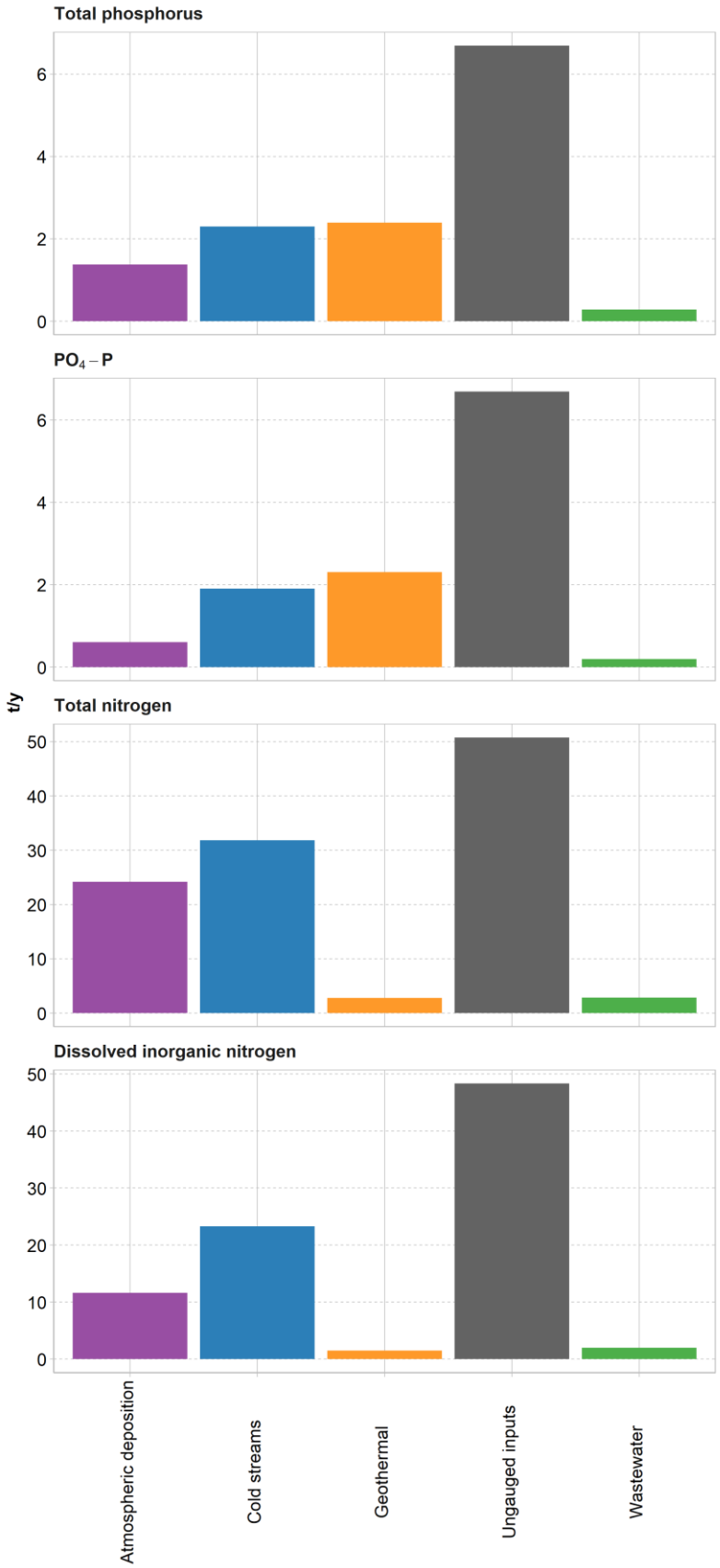


Figure 9. Summary of mean annual nutrient loads for categories of inflows. See Figure 8 for loads for individual inflows.

Table 3. Comparison of annual average nutrient loads to Lake Tarawera estimated in this study, with loads estimated in previous studies. The standard deviations of estimates from comparable studies are provided to indicate variance.

Study	TN (t/y)	TP (t/y)	Comments
This study	112.7	13.03	Average for 2010–2020
McBride et al. (2020)	106.8	10.70	Estimated for current (2020) conditions using a modified land use export coefficient approach, informed by Overseer®
Dada et al. (2016)	85.0	10.6	Partly based on land use export coefficients
Table 4.9 in White et al. (2016)	138.2	NA	For current land use but based on a steady-state loading, after groundwater concentrations have equilibrated to current land use. Does not include atmospheric deposition.
Tarawera Lakes Restoration Plan (BoPRC 2015)	94.9	11.39	Inner catchment based on Hamilton (2014); outer catchment calculated separately. Geothermal P load assumed to be 5 t/yr but is "an estimate only"
Hamilton et al. (2006)	84.6	10.41	Based on land use export coefficients
Hamilton et al. (2006)	96.2	12.7	Based on stream sampling
Donovan and Donovan (2003)	91.4	17.53	Geothermal P loads assumed to be 12 t/yr but recognises that they could be in the range 4-20 t/yr
Standard deviation (n= 7)	*10.6*	*2.5*	*Estimates by White et al. (2016) omitted from calculation as they are not directly comparable*

Model Performance

Appendix C presents calibrated DYRESM and CAEDYM parameter values, and details of model performance. These details include model performance statistics for the calibration and validation periods, in addition to time series plots that compare trends in modelled and measured values for a range of biological, physical and chemical variables across multiple depths.

In summary, the lake model accurately reproduced the physical dynamics of the lake, including the monomictic mixing regime of Lake Tarawera (Figure 10). Further, the model satisfactorily simulated measured nutrient concentrations and phytoplankton biomass (chlorophyll *a*). Simulated phytoplankton phenology was consistent with expectations, namely dominance of diatoms in the winter, the highest relative composition of cyanobacteria in the summer, and less pronounced seasonality for chlorophytes, albeit with typically highest Chlorophyta abundance in spring and summer (Jolly and Chapman 1977). Model results show a weak deep chl *a* maximum at a depth of

~20 m in some years (Figure 11), as has been observed in field studies (Hamilton *et al.* 2010; Kpodonu 2016).

The model satisfactorily simulated measured TLI_3 values (Figure 12). The overall mean of modelled annual TLI_3 values for the study period was 2.68, compared to an overall mean of measured TLI_3 of 2.76, indicating that the model slightly under-estimated TLI_3 on average. Mean absolute error was 0.15 TLI_3 units when all available data were considered (n = 11 years). Error was greatest for 2015, when the model under-predicted TLI_3 by approximately 0.3 TLI_3 units, primarily reflecting under prediction of TP concentrations. As discussed in the

Lake Water Quality section, there is particular uncertainty regarding the phosphorus measurements in the adjusted BoPRC dataset that was used, and it is notable that several values of TP concentrations were higher than values measured in the comparable NRWQN dataset, particularly around 2015–2017 (Figure 3). Thus, model performance quantified here reflects error in the model, but likely also some error in the phosphorus measurements.

The satisfactory model results described above indicate that the model adequately simulates phytoplankton ecology and the key nutrient cycling processes in Lake Tarawera. These results provide confidence in using the model for scenario analysis to examine projected nutrient load reductions.

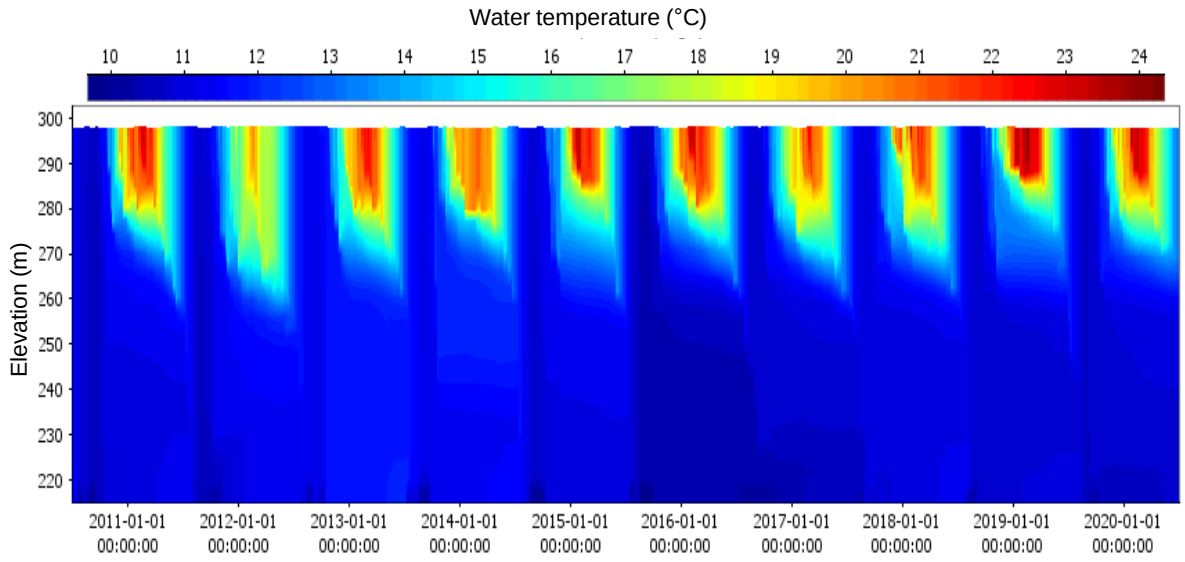


Figure 10. Simulated water temperature.

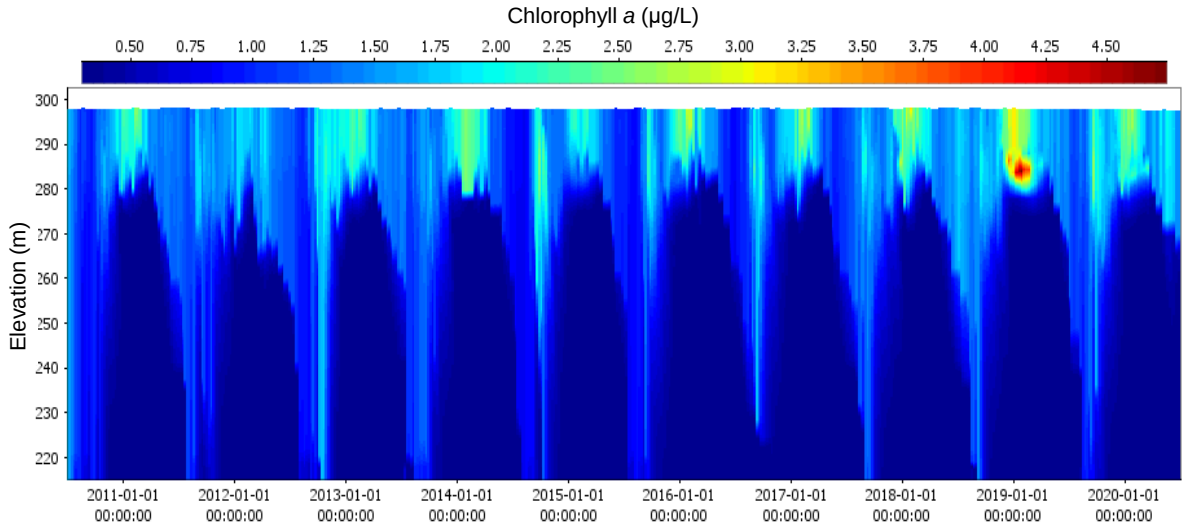


Figure 11. Simulated chlorophyll *a* concentration.

Figure 12. Modelled and measured values of TLI_3 and the three component indices, 2011–2020.

Nutrient Load Scenario Results

Nutrient load scenario results are summarised in Figure 13, which presents matrices that show a 'heat map' of 10-year averge simulated TLI_3. Plots are shown separately that quantify the external loads and total (internal plus external) loads for each scenario. The blue contour line on each plot shows the range of scenarios that correspond to the TLI target of ≤2.6. Thus, multiple permutations of changes to nitrogen and phosphorus loads correspond to the TLI target, reflecting the sensitivity of TLI to changes in either nitrogen or phosphorus loads.

Assuming nitrogen loads remain unchanged (i.e., '1N'), model results indicate that a reduction of just over 10% to the manageble portion of the phosphorus load is necessary to achieve a 10-year average TLI_3 that meets the target of 2.6. Specifically, the '1N-0.9P' scenario results in a 10-year average TLI_3 of 2.61, whereas the '1N-0.8P' scenario results in a 10-year average TLI_3 of 2.51 (see table in Appendix D). Thus, the 1N-0.9P scenario corresponds to average TLI_3 that exceeds the target by 0.1 units, although it should be noted that this difference approximates the model error, i.e., mean absolute error of 0.15 when expressed to two decimal places.

Respectively, these two scenarios correspond to total external phosphorus load reductions of 0.93 t P/y and 1.85 t P/y, or reductions to total external phosphorus loads of 7% and 14%, i.e., including the 'non-manageable' components. For further context, this range of load reductions corresponds to 40–81% of the estimated phosphorus load from cold streams, or 14–28% of the estimated phosphorus load from ungauged inflows that are assumed to predominantly comprise groundwater to the bed of the lake.

Further, a '1N-0.87P' scenario was simulated separately to directly examine the suitability of the current interim targets to reduce phosphorus loads by 1.2 t P/y while holding nitrogen loads constant. This scenario corresponds to 1.2 t P/y external phosphorus load reduction (the interim target) and yields average TLI_3 of 2.57, just below the TLI target of 2.6

All scenario results are tabulated in Appendix D, which also includes heat map plots for each TLI_3 component.

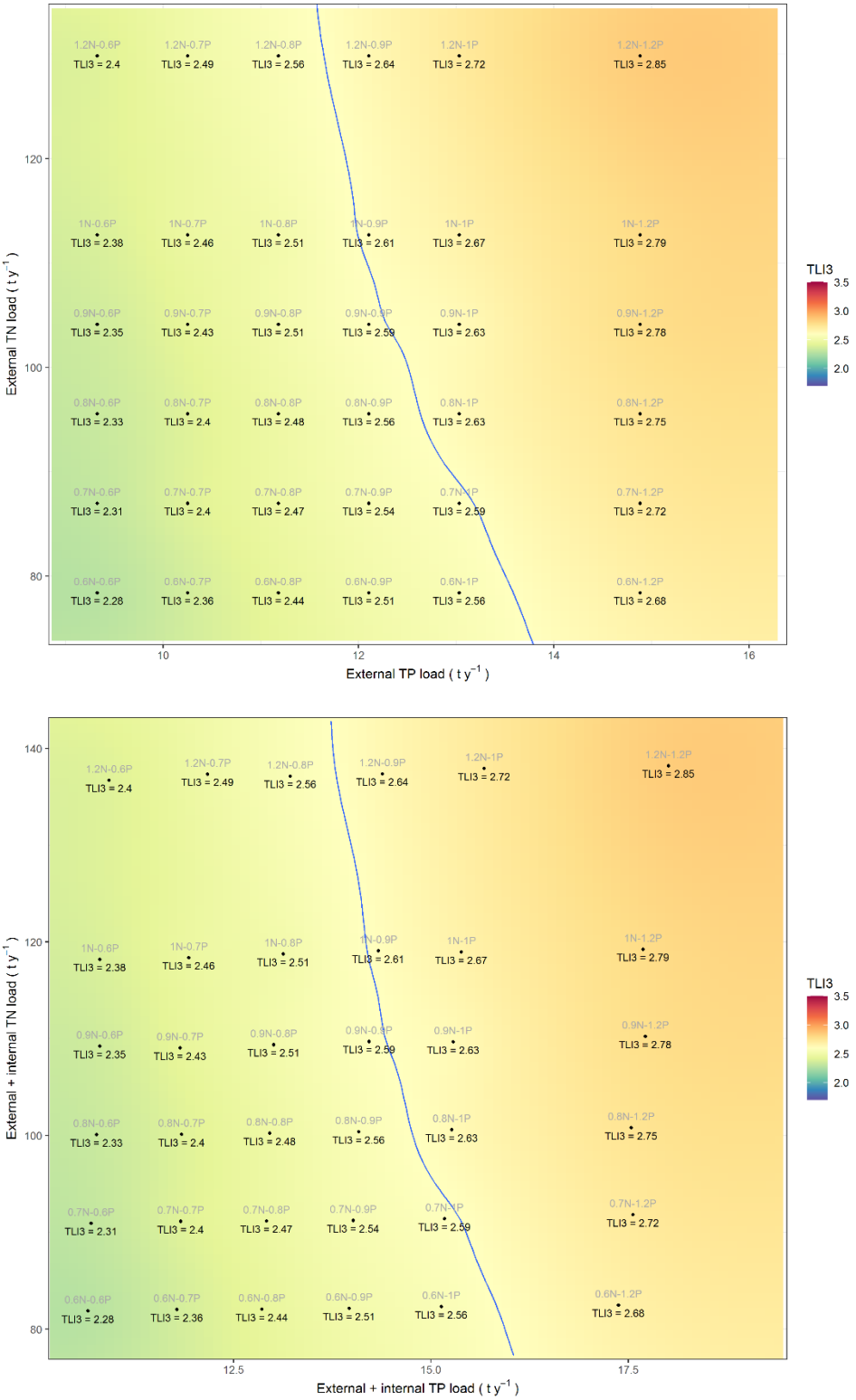


Figure 13. Summary of nutrient loading scenario results, for a) external loading only and b) external plus internal loading for nitrogen and phosphorus. Each dot represents a nutrient loading scenario representing changes to the manageable portions of the external nitrogen and phosphorus loads. The interpolated coloured background represents the average three-parameter Trophic Level Index for a 10-year model simulation (2011–2020). The blue isobar represents the TLI target of 2.6 (note the target is set using a four-parameter TLI). See Appendix D for similar plots for each TLI component.

Sensitivity Analysis

Climate Change

The results for climate change sensitivity scenarios show progressively warmer surface water temperatures with increasing magnitude of climate change (Figure 14). A consequence of this warming is a projected increase in water column stability with increasing water temperature, as reflected in the results by increases to the duration of the stratified period, particularly for the 'medium' and 'high' climate change sensitivity scenarios (Figure 15). Limitations of the model mean that there is uncertainty regarding the precise details of these projected changes to water column stability, e.g., the 'tipping point' that could cause the lake to fail to mix is uncertain. For example, changes to wind conditions in the Lake Tarawera catchment in response to climate change are uncertain and were not accounted for in these model simulations. Similarly, a temporary disequilibrium between cooler groundwater inputs and recent warming of surface waters could further enhance stability. Nonetheless, the increase in thermal stability that is projected by the model is consistent with climate change research that shows that increased thermal stability is generally projected for lakes under a warming climate (e.g., Bayer *et al.* 2013), with warm monomictic lakes potentially transitioning to meromictic lakes (albeit generally at lower latitudes than Lake Tarawera; Woolway and Merchant 2019). The consequence of no mixing for two or three years would be suboxia or anoxia in bottom waters, likely causing high phosphorus release from the Fe-rich sediment (Trolle et al 2011).

Projected changes to TLI_3 variables for the climate change sensitivity scenarios show that relative differences among scenarios are highly variable among years. This seems to reflect contrasting responses among TLI variables to climate change scenarios; e.g., compare annual TL_P values in Figure 16 (most frequently highest for the 'high' sensitivity scenario) with annual TL_N values (variable but frequently lowest for the 'high' sensitivity scenario). Mechanisms underlying these variable responses are uncertain but the responses of TLI variables to climate change are expected to be strongly influenced by the physical changes described above, as supported by Ryan et al. (2006) who showed that variability in mixing regime has a dominant role in controlling the phytoplankton community in North Island New Zealand lakes. Further climate change sensitivity analysis results are provided in Appendix D.

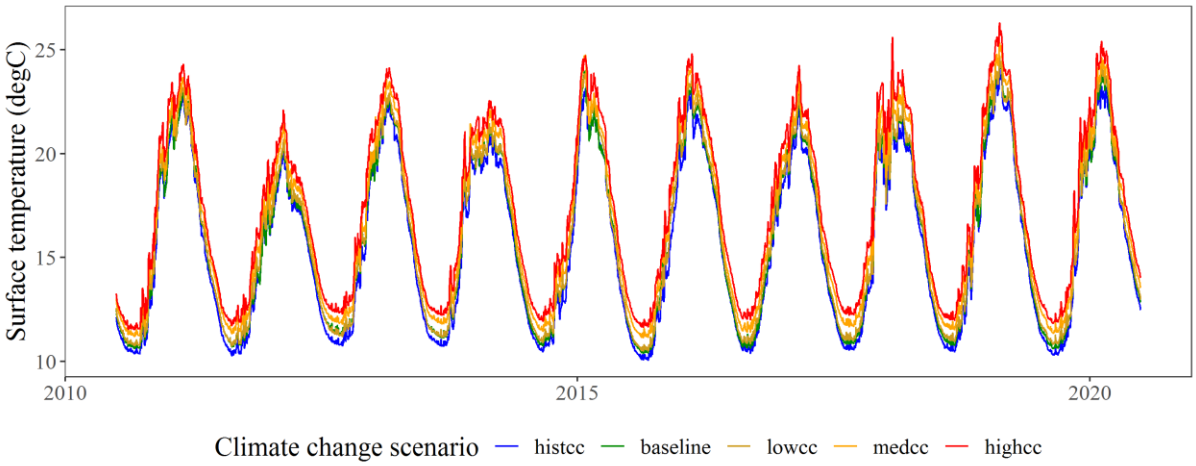


Figure 14. Lake surface water temperatures for the baseline simulation and the four climate change sensitivity scenarios. Simulated concentrations of chlorophyll *a* associated with each phytoplankton group, total chlorophyll *a*, total nitrogen and total phosphorus for the baseline simulation and the four climate change (cc) sensitivity scenarios: 1) 'low' – mean increase to baseline air temperatures of 0.26°C (RCP2.6); 2) 'medium' – mean increase to baseline air temperatures of 1.31°C (RCP6.0); 3) 'high' – mean increase to baseline air temperatures of 2.26°C (RCP8.5), and; 4) 'hist' – mean air temperature 0.61°C cooler than the model baseline period (2010–2020), corresponding to historical conditions during 1971–1990.

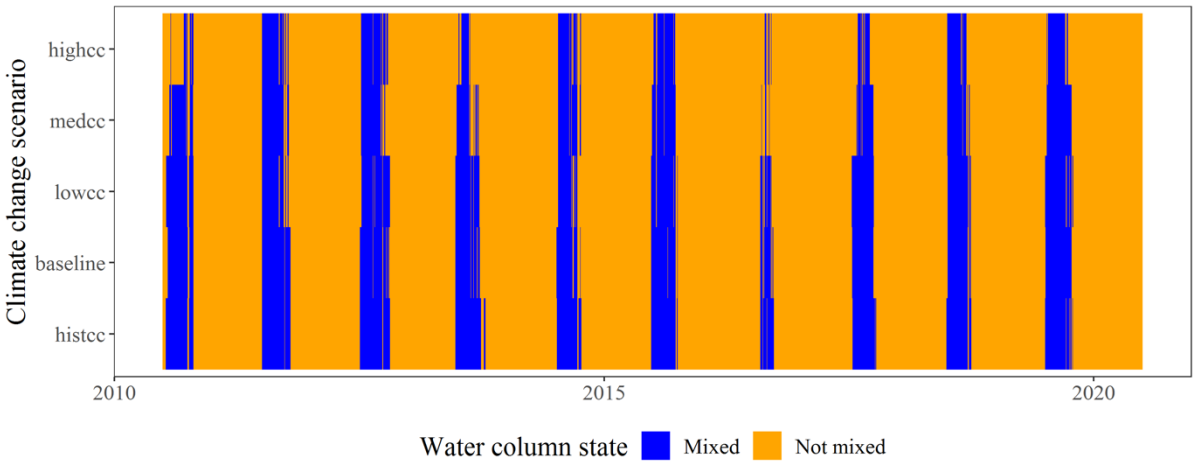


Figure 15. Water column state for the baseline simulation and the four climate change (cc) sensitivity scenarios: 1) 'low' – mean increase to baseline air temperatures of 0.26°C (RCP2.6); 2) 'medium' – mean increase to baseline air temperatures of 1.31°C (RCP6.0); 3) 'high' – mean increase to baseline air temperatures of 2.26°C (RCP8.5), and; 4) 'hist' – mean air temperature 0.61°C cooler than the model baseline period (2010–2020), corresponding to historical conditions during 1971–1990.

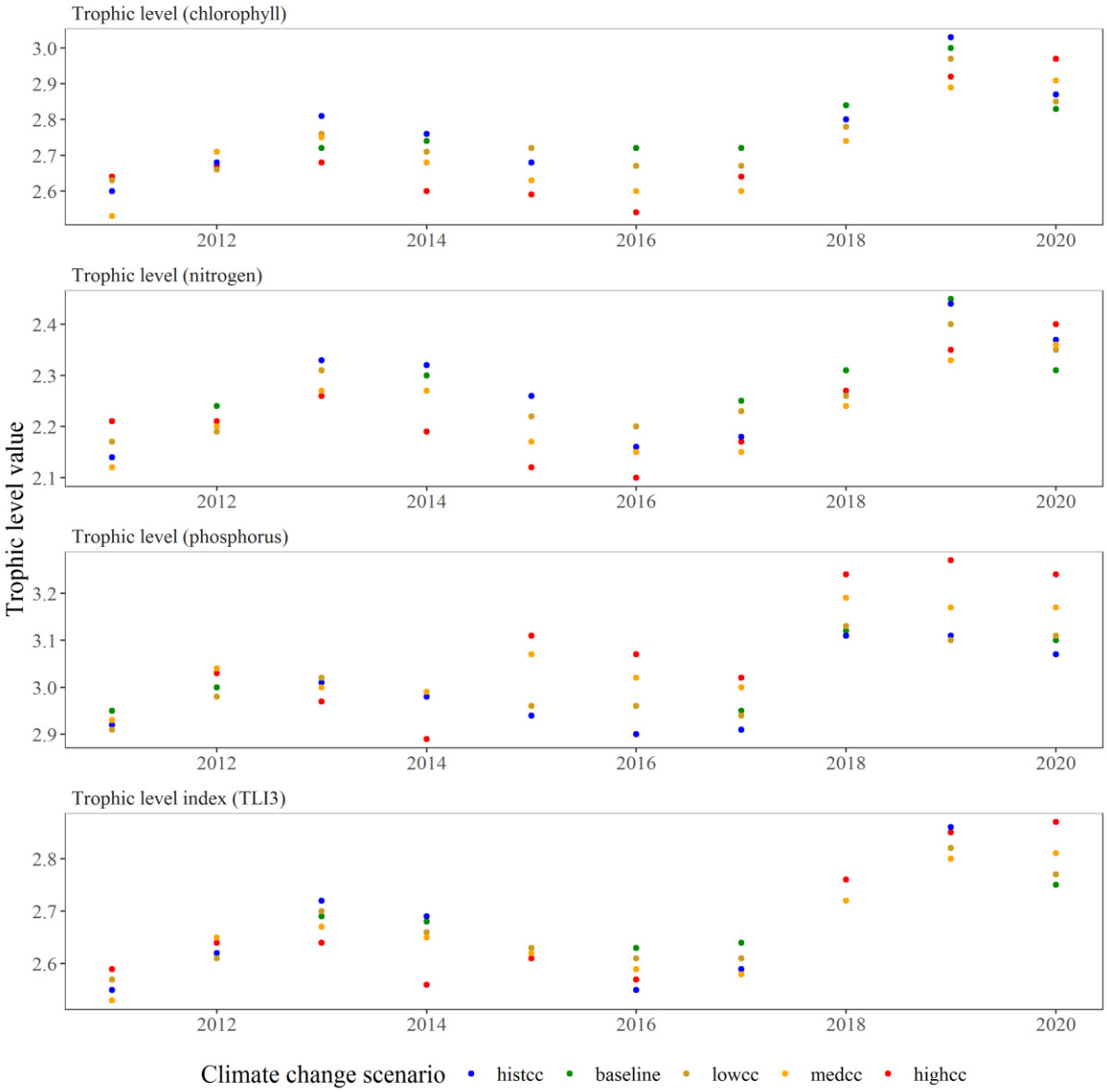


Figure 16. Simulated values of each component of the three-parameter Trophic Level Index (TLI_3) and combined TLI_3 for the baseline simulation and the four climate change (cc) sensitivity scenarios: 1) 'low' – mean increase to baseline air temperatures of 0.26°C (RCP2.6); 2) 'medium' – mean increase to baseline air temperatures of 1.31°C (RCP6.0); 3) 'high' – mean increase to baseline air temperatures of 2.26°C (RCP8.5), and; 4) 'hist' – mean air temperature 0.61°C cooler than the model baseline period (2010–2020), corresponding to historical conditions during 1971–1990.

Sediment Release Rates

The results of sediment release sensitivity scenarios are shown in Figure 17. Supplementary plots are shown in Appendix D. Results in Figure 17 are considered indicative of water quality shortly (<5 years) after nutrient load reductions, before benthic sediment nutrient release (internal loads) have equilibrated to changes in external loads. For these sensitivity scenarios, the '1N-0.9P' and '1N-0.8P' scenarios are projected to result in a 10-year average TLI_3 of 2.61 and 2.54, respectively. These values can be contrasted with the predicted 10-year average TLI_3 values of 2.61 and 2.51 for these two scenarios, when maximum sediment nutrient release rates are adjusted (see *Nutrient Load Scenario Results* above). Thus, although there is uncertainty about how sediment nutrient release rates would change following external nutrient load reductions, this analysis indicates that the rate of improvements (reductions) to lake trophic status following external load reductions is relatively insensitive to the rate of decline in sediment nutrient release rates. The results of this sensitivity analysis reflect the low maximum sediment nutrient release rates rates that were assigned during model calibration, and are representative of an oligotrophic lake.

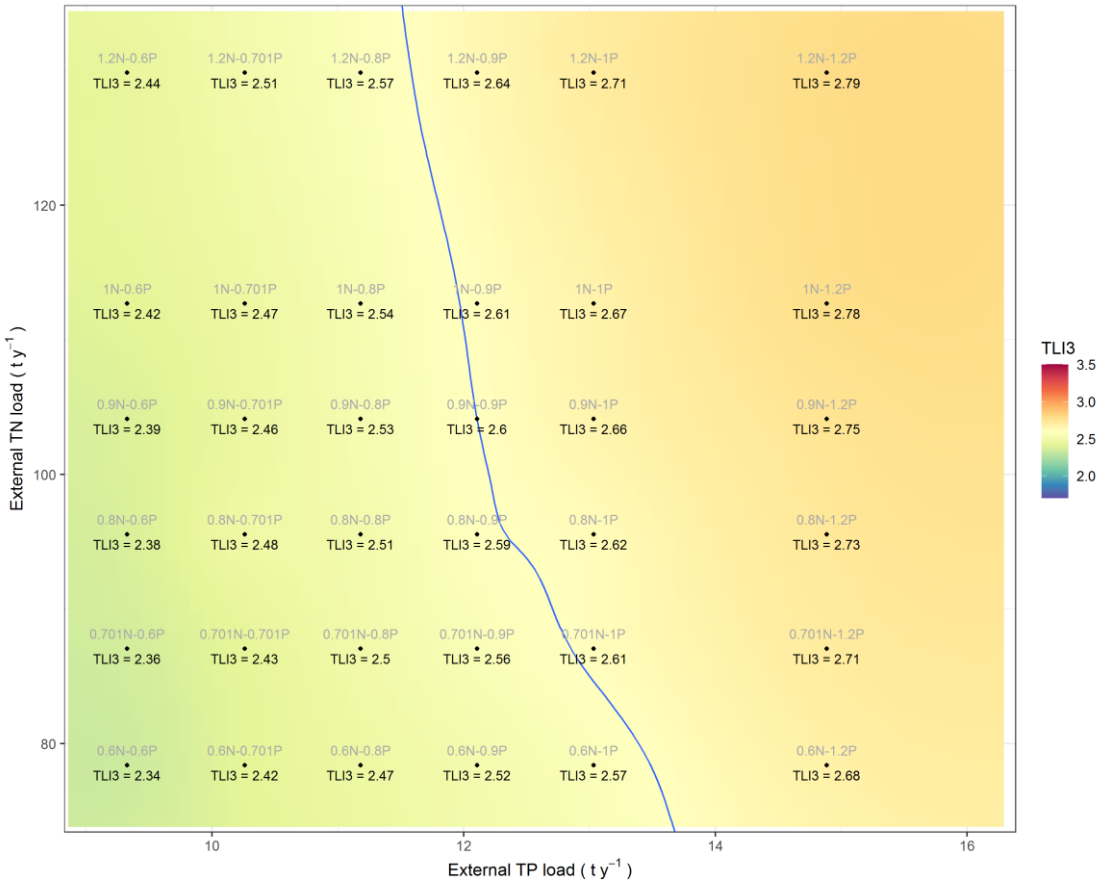


Figure 17. Sediment release sensitivity results: plot shows nutrient loading scenario results (TLI_3); however, maximum potential sediment nutrient release rates have not been adjusted by the nutrient load factors. Thus, comparing these results with Figure 13 shows the sensitivity of scenario results to whether or not maximum sediment nutrient release rate is varied when configuring scenarios. Each dot represents a nutrient loading scenario comprising changes to the manageable portions of the external nitrogen and phosphorus loads. The interpolated coloured background represents the average three-parameter Trophic Level Index for a 10-year model simulation (2011–2020). The blue isobar represents the TLI target of 2.6 (note the target is set using a four-parameter TLI). See Appendix D for similar plots for each TLI component.

Uncertainties

The complex volcanic geology of the Lake Tarawera catchment presents a challenge to lake and catchment management. Considerable work has been undertaken to improve understanding of groundwater interactions (e.g. White *et al.* 2016), yet there remains large uncertainty regarding the hydrology and chemistry of groundwater. Groundwater to the bed of the lake makes a major contribution to the water and nutrient budget, e.g., see 'ungauged inputs' in Figure 9, which predominantly comprise groundwater. Further work to quantify this flux would reduce this key source of uncertainty. This uncertainty poses challenges for setting nutrient load reduction targets that are based on reductions to existing external loads because the existing external load that enters the lake via groundwater is uncertain, and changes to this flux cannot be readily measured.

An additional uncertainty concerns the role of nitrogen fixation in the nitrogen budget for Lake Tarawera. Nitrogen fixation was simulated in the model, consistent with research that has shown that cyanobacteria comprise a major proportion of the phytoplankton assemblage during the summer, with cyanobacteria shown to contain heterocysts, which are the cells involved in nitrogen fixation (Baldwin 2009). Baldwin (2009) concluded that the overall contribution of nitrogen fixation to the lake nitrogen budget is highly uncertain but was estimated to provide between 0.04% and 8% of 'new' nitrogen input to the lake. Further, extensive model calibration undertaken for this study showed that it was essential to represent nitrogen fixation in the model to reproduce the pronounced annual 'hump' that is observed in measurements of surface total nitrogen concentrations between late spring and early autumn. Thus, it seems clear that nitrogen fixation has an important role in nitrogen cycling within Lake Tarawera, although there is uncertainty regarding the magnitude of this process and how it would interact with changes in external loads. This is pertinent because nitrogen fixation could offset external nitrogen load reductions to some extent. This has led some researchers to question the rationale for nitrogen load reductions for lakes in general (Schindler *et al.* 2008), although this view has been criticised by others who have demonstrated that nitrogen fixation cannot completely offset nitrogen load reductions and argue that nitrogen load reductions are an important component of controlling eutrophication (Scott and McCarthy 2010; Shatwell and Köhler 2019). The need for management of both nitrogen and phosphorus in Te Arawa lake catchments is recognised in regional policy (BOPRC 2008b), supported by scientific research (e.g. Smith *et al.* 2016).

Nitrogen fixation is assumed to be the main cause of the more muted response of TLI_3 to nitrogen load reductions compared to phosphorus load reductions, as evidenced by the somewhat vertical alignment of the blue line in Figure 13. Nitrogen fixation in the model was only simulated in one of the three phytoplankton groups (cyanophytes). The maximum nitrogen fixation rate (0.3 mg N/mg chl *a*/day) was set lower than the default value (2.0 mg N/mg chl *a*/day). This value was similar to (albeit higher than) the value of 0.15 mg N/mg chl *a*/day that was used in both of the two published studies that we are aware of that have simulated nitrogen fixation using CAEDYM, undertaken in Lake Kinneret, Israel (Gal *et al.* 2009) and Lake Mendota, USA (Kara *et al.* 2012). Further, a high physiological 'penalty' was assigned to nitrogen fixation by configuring the maximum phytoplankton growth rate to decline by 80% during maximum nitrogen fixation, which is higher than the penalty assigned in the two studies cited above. CAEDYM does not directly calculate the total amount of nitrogen fixed in the water column; however, a routine was developed as part of this study to quantify this – this showed that, on

average, nitrogen fixation contributed 0.068 t N/day to the lake for the baseline simulation, equating to 22% of the estimated external nitrogen load. This simulated contribution is substantially higher than Baldwin's (2009) maximum estimate of 8%. This simulated contribution is also high for oligo-mesotrophic lakes generally, although it is plausible for more productive systems. For example, Tison *et al*. (Tison *et al.* 1977) report that nitrogen fixation contributed <1% of the total annual nitrogen input from other sources in four oligo-mesotrophic lakes in the USA, whereas the contribution of nitrogen fixation in eutrophic monomictic Rotongaio (south of Taupō) was estimated to equate to 23–24% of external nitrogen loads (Viner 1985). Thus, it is possible that the contribution of nitrogen fixation was overestimated in the model, although it is notable that, despite very extensive calibration effort, it was not possible to satisfactorily reproduce the observed seasonal trends in surface TN concentrations if a lesser role was assigned to nitrogen fixation.

As a sensitivity test, nutrient load scenarios were simulated with nitrogen fixation disabled in the model. In this test, cyanophytes completely failed to grow and dissolved reactive phosphorus accumulated to erroneously high levels in surface waters as the dominance of nitrogen limitation increased in the lake. These unrealistic results support our approach of including nitrogen fixation in the model, although this sensitivity test does not provide information about the accuracy of how nitrogen fixation was configured in the model, nor whether the response of N-fixation dynamics (as modelled) to changing external nutrient loads was realistic. Were N-fixation unable to compensate for any of the shortfall in nitrogen supply under a regime of reducing external N loads, then a lesser reduction in phosphorus might be required to meet TLI targets than suggested in the present simulations of nutrient load reduction scenarios. Further context regarding the simulated role of nitrogen fixation is provided in Appendix C, and this is a candidate for additional investigation and research.

In addition to groundwater interactions and nitrogen fixation, there is uncertainty about the relative roles of key nutrient cycling processes, including settling and mineralisation of organic material. This could be partly addressed by determining organic nutrient concentrations in lake water samples, which is not routinely undertaken. More broadly, this study highlighted the importance of lake-specific biogeochemistry data for model development. To enhance confidence in model results, this study included an extensive calibration phase that was undertaken over several months and included development of sophisticated auto-calibration routines to understand how equations in the model interact. This increased confidence that satisfactory fits between modelled and measured values reflected accurate conceptualisation of underlying processes, although further lake-specific studies of limnological processes (e.g., as have been undertaken for Rotorua) would provide greater confidence in this aspect.

Ideally, uncertainty in limnologic processes in Lake Tarawera, upstream lakes, and the wider hydrologic system, including groundwater flows and geothermal activity, could be constrained in an integrated way with a linked suite of models, but existing models couple only some of these components. Given these gaps, and the uncertainties in hydrological fluxes between catchments (including Rotomahana), high effort would be required to develop such an integrated modelling approach, entailing a multi-year and multi-partner research programme.

Conclusion

A 1–D lake ecosystem model (DYRESM-CAEDYM) of Lake Tarawera was successfully configured, calibrated and validated for the period 2010–2020. The average external annual TN load to the lake represented in the model was 112.7 t/y, while the annual TP load to the lake represented in the model was 13.03 t/y. These average external nutrient loads were calculated using the most current data available and are ~15–20% higher than the estimated loads presented in the Tarawera Lakes Restoration Plan (BoPRC 2015).

Nutrient load scenarios were simulated by modifying the manageable portions of the external loads by factors in the range 0.6 to 1.2 and calculating annual TLI_3 values (i.e., TLI without including Secchi depth in the calculations). Analysis of measured data showed that it was appropriate to directly compare TLI and TLI_3 values, as paired values of the two metrics are so similar that the metrics may be considered interchangeable. Assuming nitrogen loads remain constant, model results show that reduction of 10–20% to the manageble portion of the phosphorus load is necessary to achieve a 10-year average TLI_3 that meets the target of 2.6. Specifically, the '1N-0.9P' and '1N-0.8P' scenarios resulted in 10-year average TLI_3 of 2.61 and 2.51, respectively (Figure 13). Thus, the 1N-0.9P scenario corresponds to average TLI_3 that only slightly exceeds the TLI_3 target, although this difference approximates the margin of model error. Respectively, these two scenarios correspond to total external phosphorus load reductions of 0.93 t P/y and 1.85 t P/y. These reductions of 10% and 20% respectively to the "manageable" phosphorus loads correspond to reductions to total external phosphorus loads of 7% and 14%, including the 'non-manageable' components. Over 20 additional permutations of nitrogen and phosphorus load reductions that correspond to average TLI_3 at or below the target are tabulated in Appendix D. Scenario results indicate that the current interim targets to reduce phosphorus loads by 1.2 t P/y while holding nitrogen loads constant are sufficient to achieve the TLI target. This conclusion is supported by the result of a '1N-0.87P' scenario that was run separately: this scenario corresponds to 1.2 t P/y external phosphorus load reduction (the interim target) and yields average TLI_3 of 2.57, just below the TLI target of 2.6.

Nutrient load reductions simulated in this study are in addition to those required to balance the increases in groundwater loads that are expected to occur as groundwater chemistry equilibrates to current land use and management practices. For nitrogen, the additional load 'to come' was estimated as approximately 50 t N/y, based on comparison with results of groundwater modelling led by GNS Science (White *et al.* 2016), although estimates of current and future groundwater loads are uncertain. The potential increase in phosphorus load is unknown but it is likely to be substantially less than nitrogen when expressed as a proportion of the current load, given that a lower proportion of the TP load from land use activities is expected to be transported in groundwater. This expectation is consistent with the 'protected' status of the larger catchment, which includes other lakes that buffer phosphorus inputs due to sedimentation processes. Overall, the magnitudes of groundwater inputs to the bed of the lake are uncertain, yet they are estimated to make major contributions to the water and nutrient budgets, e.g., see 'ungauged inputs' in Figure 9, which predominantly comprise groundwater. Existing measurements of the chemistry of surface inflows that included groundwater seeps (primarily collected by Mr T. Beckett) were used extensively in this study to improve the accuracy of load estimates; however, further work to quantify groundwater fluxes would reduce this

key source of uncertainty. The scope of such work could involve analysis and expansion of recently collected geochemical tracer datasets, further hydrogeology including refinements to the existing groundwater modelling led by GNS, and additional direct measurements of nutrient concentrations in groundwater wells or at lakebed groundwater seeps to quantify current concentrations of nutrients in groundwater (particularly dissolved phosphorus), which are highly uncertain yet have a large bearing on estimates of nutrient loads to the lake. The role of nitrogen fixation in the nitrogen budget for Lake Tarawera is another key uncertainty, which can be made directly or estimated from mass balance if all other inputs including groundwater are better constrained.

The nutrient load scenario results are also independent of the effects of climate change, which were examined using a sensitivity analysis of four climate change sensitivity scenarios. This sensitivity analysis showed the potential for substantial but complex effects of climate change on lake water quality. In particular, a warming climate is projected to increase water column stability, which would extend the period of stratification and could trigger biogeochemical changes such as lower dissolved oxygen concentrations in the hypolimnion. This could trigger a positive feedback cycle of enhanced phosphorus release and worsening water quality. Lake Tarawera is particularly susceptible as it has high benthic sediment phosphorus concentrations, despite its oligotrophic status (Trolle *et al.* 2008). Hypolimnetic waters currently remain oxygenated; however, annual minimum dissolved oxygen concentrations approach 3 mg/L towards the end of the stratified period (see Figure C-2 in Appendix C), and therefore further hypolimnetic oxygen decline could result in hypoxic or anoxic conditions that promote enhanced release of iron-bound phosphorus (Mortimer 1942; Boström *et al.* 1988).

The lake model provides a decision support tool to assist Bay of Plenty Regional Council with ongoing lake and catchment management. The model has been configured to readily incorporate inputs from further analysis of external loads or groundwater models, and the matrix of scenarios provides a straightforward tool for rapid scenario assessment. When combined with outputs from a catchment model, these tools can support evaluation of the effects of land use change scenarios on lake water quality.

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