Lake Rotorua Treated Wastewater Discharge: Environmental Effects Study

August 2015

DRAFT Report

Draft client report prepared for Rotorua Lakes Council

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Cover: View eastwards across Lake Rotorua.

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NON–TECHNICAL EXECUTIVE SUMMARY

Rotorua Lakes Council is undertaking a decision–making process to resolve how treated municipal wastewater from the City of Rotorua (Bay of Plenty) should be discharged after 2019, when irrigation operations at the Land Treatment System (LTS) in the Whakarewarewa Forest are scheduled to cease.

Six main options and several associated sub–options for enhanced wastewater treatment and/or land treatment were investigated. The options involve varying grades of treatment to enhance the removal of nutrients from the wastewater relative to current treatment performance. Six potential treated wastewater discharge locations have been identified from three sites which include: in the lower reach of the Puarenga Stream, on the lake shoreline near Sulphur Bay, and an offshore discharge on the lake bed 2 km to the north of the Puarenga Stream mouth.

From an environmental perspective, it is important to consider the potential impacts of discharging nutrients to the lake because additions of nutrients can cause undesirable ecological effects such as excess algae growth. Such effects are associated with a process called eutrophication. The assessment also considered potential effects related to nitrogen toxicity, dissolved oxygen and the growth of algae attached to the bed of the Puarenga Stream for a discharge to this stream, as well as impacts on the lake. Potential risks to human health risk were examined by considering summary data of projected bacteria concentrations in the treated wastewater that were provided for this initial assessment stage. A full and detailed assessment of public health risks associated with bacterial contamination was not undertaken; however, we provide details of issues that should be considered at later assessment stages.

Environmental computer modelling results showed that effects associated with lake eutrophication for each of the options would either be neutral or minor. In the lower Puarenga Stream, minor negative effects were predicted in relation to nitrogen toxicity and minor negative effects were predicted for dissolved oxygen concentrations, although this aspect of the assessment was based on a ‘worst case scenario’ of discharging treated wastewater in which there is no dissolved oxygen. Neutral effects were predicted in relation to stream algae growth. It is important to note that effects to the Puarenga Stream are limited to a short (< 2 km) section of the stream downstream of State Highway 30 where the discharge would occur.

Further environmental computer modelling was undertaken to examine how treated wastewater is expected to disperse in the lake. Wind conditions were predicted to exert a major control on how discharged treated wastewater is dispersed in the lake. Results showed that the concentration of treated wastewater in lake water would generally be low (typically <1%) throughout the lake, including near–shore areas along Rotorua City lakefront. Higher concentrations were predicted to occur in Sulphur Bay for a scenario of discharge to Puarenga Stream. Immediately offshore of the stream mouth, these may be up to ~25% but would more typically be closer to 10%. Discharge to the lake shoreline site was predicted to result in lower concentrations near the outfall because the site is nearer the mouth of Sulphur Bay, which promotes slightly increased mixing. Offshore discharge to the lake bed was predicted to result in the lowest concentrations in surface waters and at near–shore locations. Discharge at this site was, however, predicted to sometimes result in localised high concentrations (>70%) of treated

wastewater near the bed of the lake, around the discharge site. This could occur during times in the summer, when the treated wastewater is expected to be cooler than the lake water, and would therefore be initially confined to bottom waters immediately following discharge.

EXECUTIVE SUMMARY

Rotorua Lakes Council is undertaking a decision–making process to resolve how treated municipal wastewater from the City of Rotorua (Bay of Plenty) should be discharged after 2019, when irrigation operations at the Land Treatment System (LTS) in the Whakarewarewa Forest are scheduled to cease.

Six main options and several associated sub–options for enhanced wastewater treatment and/or land treatment were investigated. The options involve varying grades of treatment to enhance the removal of nitrogen and phosphorus from the wastewater relative to current treatment performance. In total, the options yield eleven permutations of final treated wastewater composition. The options are summarised below:

Table Summary of treatment options.

Option	Description	Sub-options	Details	Source
1	Base option	-	Upgrades to current tertiary treatment by addition of: flow balancing, P removal with chemical addition (alum) and UV disinfection.	Mott MacDonald (2014)
2	Base option + basic filtration	a. Disk filter b. Sand filter c. Membrane filter	Addition of filtration to remove solids, including particulate N and P.	Mott MacDonald (2014)
3	Base Option + filtration + denitrifying filter/bed	a. Denitrifying sand filter b. Sand filter + denitrifying carbon bed	Addition of filtration to remove solids, in addition to final denitrification step to convert dissolved inorganic N to atmospheric N gas.	Mott MacDonald (2014)
4	30 t N/y and 3 t P/y	-	Treatment processes configured to achieve maximum releases permitted under current Resource Consent conditions.	J. Bradley, pers. comm. 2015
5	30 t N/y and 1.5 t P/y	-	Treatment processes configured to achieve maximum N release and 50% of P release permitted under current Resource Consent conditions.	J. Bradley, pers. comm. 2015
6	Membrane bioreactor system rebuild	a b	No additional P treatment. + additional P treatment	K. Brian, pers. comm. 2015a

Six potential treated wastewater discharge locations have been identified for discharge to water:

1) three sites in the lower reach of the Puarenga Stream;
2) two sites along the shore of Lake Rotorua close to the mouth of the Puarenga Stream;
3) one offshore site on the bed of the lake 2 km north of the mouth of the Puarenga Stream.

This environmental effects study aims to inform the decision–making process by assessing effects on water quality in the Puarenga Stream and Lake Rotorua. Treated wastewater discharge contributes nutrient loads to Lake Rotorua and therefore a primary focus of the assessment involved considering effects related to eutrophication. Other issues considered were effects on stream ecosystem health related to nitrogen toxicity, dissolved oxygen concentration and periphyton proliferation. In addition, effects on human health risk were considered by examining summary statistics of projected *E. coli* concentrations (an indicator of fecal contamination) in the context of background levels in the stream.

Three main techniques were used to inform the assessment:

1) *Mass balance calculations*.

   Effects on the following environmental aspects in the Puarenga Stream were assessed in the context of Attribute State values defined in the National Policy Statement for Freshwater Management 2014: nitrate nitrogen (toxicity), ammoniacal nitrogen (toxicity), dissolved oxygen, *E. coli* and periphyton. The assessment was based on projected concentrations for each option, which are expressed as constant values.

2) *One–dimensional (1–D) lake modelling*.
   A numerical water quality model was configured to simulate the water quality effects of discharging treated wastewater, relative to a baseline period (2007–2014) that was taken to be representative of current conditions. The model was used to simulate mean annual values of a Trophic Level Index (TLI)[1] for a range of scenarios. This allowed examination of effects on lake trophic state as a consequence of changing nutrient loads to the lake. Measured and modelled concentrations of total nitrogen, total phosphorus and chlorophyll *a* were also compared with Lake Ecosystem Health Attribute State values defined in the National Policy Statement, to assess the implications of the proposals relative to defined attribute states .
3) *Three–dimensional (3–D) lake modelling.*
   A 3–D hydrodynamic model was configured to examine the mixing processes that control how simulated treated wastewater inputs are diluted and dispersed within the lake. The model was used to compare how dispersion of treated wastewater varied under different environmental conditions and with discharge simulated to the Puarenga Stream, a lake shoreline site and the proposed offshore lake bed site. One lake shoreline site was represented (Site 5), although results are expected to be consistent with those for discharge to the alternative site (Site 4) which is only ~500m to the south.

The projected discharge rate of treated wastewater is 23.81 ML/d (0.28 m^3/s). Depending on flow conditions, discharging treated wastewater to the Puarenga Stream would result in treated wastewater comprising ~ 1 – 25% of the total combined stream flow. Values at the upper end of this range would only occur during unusually low flows. Historic stream discharge data indicate that the proportion of the stream flow that would comprise treated wastewater would be < 14% for 50% of the time, and <18% for 90% of the time.

The projected nutrient concentrations in the treated wastewater are generally higher than background concentrations in the stream, although differences are small for some options (see Table 2 and Table 13 in the main report). The nutrient loads associated with each option are lower than estimated

[1] TLI_3 was used, which omits Secchi depth from calculations.

background loads in the Puarenga Stream but, in broad terms, they are comparable with loads conveyed in one of the nine major streams that flow in to Lake Rotorua. Loads are summarised below:

Table Summary of estimated annual nutrient loads in the Puarenga Stream (2007–2014) and loads associated with each treatment option.

Scenario/Option	Description	TN (t/y)		DIN (t N/y)		TP (t/y)		PO_4-P (t P/y)	
		Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.
1D_0_Stream	Baseline Puarenga Stream loads (PO_4-P attenuated by alum)	70.1	16.4	58.1	11.7	6.0	1.6	1.4	1.1
1D_0-LTS	Baseline Puarenga Stream loads with LTS loads removed	34.0	8.4	22.0	3.5	4.8	1.3	1.1	0.9
1D_0 - Alum	Baseline Puarenga Stream loads with no alum dosing	70.1	16.4	58.1	11.7	6.9	1.9	2.3	0.5
Option 1	Loads in treated wastewater	47.3	0.1	28.5	0.0	6.3	0.0	0.9	0.0
Option 2a		42.3	0.1	28.5	0.0	3.2	0.0	0.9	0.0
Option 2b		40.2	0.0	28.5	0.0	1.7	0.0	0.9	0.0
Option 2c		38.0	0.0	28.5	0.0	0.9	0.0	0.9	0.0
Option 3a		22.9	0.0	11.2	0.0	1.7	0.0	0.9	0.0
Option 3b		31.6	0.0	19.9	0.0	1.7	0.0	0.9	0.0
Option 4		30.0	0.0	28.5	0.0	3.0	0.0	0.9	0.0
Option 5		30.0	0.0	28.5	0.0	1.5	0.0	0.9	0.0
Option 6a		30.7	0.0	22.6	0.0	3.0	0.0	3.0	0.0
Option 6b		30.7	0.0	22.6	0.0	1.5	0.0	1.5	0.0

Relative to the 2029 external nutrient load reduction targets set for the Lake Rotorua catchment in the Lakes Rotorua and Rotoiti Action Plan, loads discharged to the lake for the various treatment options range from 9% to 19% of the nitrogen load target (250 t N/y) and 9% to 63% of the phosphorus load target (10 t P/y).

Mass balance calculations undertaken in the context of guideline analyte concentrations in the National Policy Statement for Freshwater Management 2014 showed that proposed wastewater discharge to the stream would have either neutral (ammonium toxicity) or minor negative impacts (nitrate toxicity, dissolved oxygen). Semi–quantitative assessment based on summary statistics of projected wastewater composition (following membrane bioreactor treatment) showed that effects related to *E. coli* would be neutral or very minor (negative) depending on the level of treated wastewater disinfection achieved. This assessment was preliminary in that the projected concentrations related to effluent from membrane bioreactor treatment, and may not be applicable to all options; recommendations to guide further assessment during later design stages are presented. Absence of data precluded a quantitative assessment regarding periphyton, although qualitative assessment indicated that negative impacts on this aspect are unlikely based on consideration of factors that currently limit periphyton growth in the lower Puarenga Stream (suitable substrate is limited).

One–dimensional water quality modelling results showed that effects associated with each treatment option on lake trophic status would be neutral to very minor (negative). Effects were quantified on the basis of mean TLI for the eight-year modelling period, with options resulting in changes in TLI values of only 0 to 0.02 units. These changes are low; e.g., the range in measured TLI during the baseline period was ~0.7 units. Uncertainty in this result is low; however, model performance in predicting TLI was only moderate. While the model matched the measured eight-year mean TLI extremely closely (error < 0.01 unit), the model did not reproduce inter-annual differences in annual TLI values well. This was likely caused by the fact that the modelling period coincided with the period when aluminium sulphate (alum) was dosed to either one or two stream inflows to the lake to reduce dissolved reactive phosphorus concentrations. This action has led to a marked improvement in lake water quality; however, it was only possible to represent this action in the model in a static way, which did not account for the considerable variability in alum dosing rates that occurred during the period. The modelling indicated that alum dosing to the Puarenga and Utuhina Streams likely has a much greater impact on lake water quality than that predicted for any of the wastewater discharge options, and how the alum dosing plants are operated will have a significant impact on future water quality.

Consistent with the very minor effects associated with the wastewater treatment options on TLI, the 1–D model results indicated that the options would not cause a change in baseline Lake Ecosystem Health Attribute State values defined for total nitrogen, total phosphorus and chlorophyll *a* concentrations. In the context of the decision–making process, the lack of marked difference between the six wastewater treatment plant and land treatment options highlights the importance of carefully weighing up the cultural and economic considerations (not considered in this study) associated with each option. If large expenditure is required for relatively marginal improvements in wastewater treatment, then this may be more effectively invested elsewhere in the lake catchment to support lake water quality management, on the basis of $/t of nutrient load reduced. Similarly, cultural evaluation of various disposal methods might be prioritised over small differences in final treated wastewater concentrations and loads.

Three–dimensional hydrodynamic model simulations showed that treated wastewater concentrations in the lake (represented in the model using a conservative tracer) would generally be low (typically <1%) throughout the lake, including near–shore areas along Rotorua city lakefront. There is low uncertainty in this general result, which is consistent with the small volume of the wastewater discharge relative to the lake volume. Under certain conditions, concentrations may be higher close to the discharge sites. Results indicate that discharge to Sulphur Bay via the Puarenga Stream could lead to higher concentrations (up to ~20%) 100 m offshore of the stream mouth during certain periods. Modelled discharge at the lake shoreline at Site 5 (the more northerly of the two proposed shoreline sites) resulted in greater dispersion around the outfall location, with concentrations ~1–7% 100 m offshore of the outfall. Discharge to the lake bed (Site 6) resulted in the lowest surface water concentrations, although bottom water concentrations could be very high (70–95%) in the immediate vicinity of the discharge site during summer when the projected wastewater temperature was cooler than the lake. The small difference between maximum projected treated wastewater temperature (18 °C) and maximum lake water temperature (~21 °C) means that such accumulation in bottom waters would only occur during a ~2–3 month period, assuming that temperature exerts the dominant control on treated wastewater density.

The 3–D simulations highlighted the potential for wind–driven basin–scale circulation processes to greatly influence how treated wastewater mixes throughout the lake, depending on prior wind conditions and the location of the outfall. Specifically, southwest (SW) winds were predicted to cause partial accumulation of treated wastewater along the eastern shore of the lake for the scenarios of discharge to the Puarenga Stream or the lake shore site. North–east winds were predicted to cause more limited transport of treated wastewater towards Rotorua City lakefront, with this effect less pronounced for the scenario of lake shoreline discharge. Despite these effects, surface concentrations were still predicted to be low (<1%) in these near–shore areas. Offshore discharge to the lake bed was predicted to result in the lowest accumulation in near–shore areas due to increased advective transport and mixing of treated wastewater. Uncertainty in the predicted effects associated with basin–scale circulation processes is moderate, and model predictions have not been validated in the vicinity of the proposed discharge sites. Details of the studies that would be required to validate model predictions are presented.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF MAPS

Glossary

Bardenpho	A biological nutrient removal system that comprises a series of tanks with alternating anoxic/aerobic conditions to remove both N and P. Added to the Rotorua WWTP in 1991.
BoPRC	Bay of Plenty Regional Council
CAEDYM	Computational Aquatic Ecosystem Dynamics Model. An aquatic ecology and water quality model.
Chl *a*	Chlorophyll *a*
DON	Dissolved organic nitrogen
DRP	Dissolved reactive phosphorus
DYRESM	Dynamic Reservoir Simulation Model. A 1–D hydrodynamic model.
ELCOM	Estuary and Lake Computer Model. A 3–D hydrodynamics model.
LTS	Land Treatment System. Treated wastewater is currently spray–irrigated at the LTS, located to the south of Lake Rotorua.
MBR	Membrane bioreactor. A nutrient removal system that combines biological treatment and membrane separation. Added to the Rotorua WWTP in 2012.
N	Nitrogen
NH_4–N	Ammonium nitrogen
NOx–N	Nitrate plus nitrate nitrogen
P	Phosphorus
PO_4–P	Phosphate phosphorus
PON	Particulate organic nitrogen
PN	Particular nitrogen
PP	Particulate phosphorus
Q	Stream discharge
r	Pearson's correlation coefficient
RDC	Rotorua District Council

RMSE	Square root of the mean squared error
RPSC	Rotorua Project Steering Committee
TAG	Technical Advisory Group
TLI	Trophic Level Index. The metric is termed TLI_3 when it is calculated without Secchi depth data as values are based on three (rather than four) water quality variables.
TN	Total nitrogen
TP	Total phosphorus
WWTP	Waste water treatment plant

Introduction

Rotorua Lakes Council (RLC) is undertaking a decision–making process to resolve how treated municipal wastewater from the city of Rotorua (Bay of Plenty) should be discharged after 2019, when irrigation operations at the Land Treatment System (LTS) in the Whakarewarewa Forest are scheduled to cease. The process is being led by an appointed Rotorua Project Steering Committee (RPSC), which is assisted by an associated Technical Advisory Group (RPSC TAG) in providing advice on technical issues.

The Environmental Research Institute, University of Waikato, was commissioned to lead an environmental effects study that considered a range of proposed options for both treatment and discharge of municipal wastewater to Lake Rotorua. This assessment aims to inform the RPSC's decision–making process, an outcome of which will be a 'preferred disposal option', recommended to Rotorua Lakes Council by the Steering Committee. The preferred option will be subject to a separate Assessment of Environmental Effects following preliminary design (RLC 2014).

The study presented here includes mass balance calculations and environmental modelling to examine water quality effects associated with the proposed options. Disposal options include direct discharge to the lake via either sites on the shoreline or lake bed. Additionally, the options include indirect discharge to the lake following discharge to land or to the lower reach of the Puarenga Stream. As such, potential effects to both the Puarenga Stream and Lake Rotorua are considered.

Background and objectives

Lake Rotorua

Background
Lake Rotorua (Map 1) is nationally iconic and represents an important resource for Rotorua, supporting a range of recreational opportunities that attract tourists to the region. The lake is highly valued by Māori, and the lake is of particular cultural significance to Te Arawa who are the legal owners of the lake bed.

The lake is large (≈ 80 km^2) and volcanically–formed. As a consequence of its relatively shallow depth (mean depth ≈ 10 m), the lake is polymictic and only stratifies (density driven isolation of surface and bottom waters) continuously for periods of up to several weeks during calm conditions in summer months (November–March). Since the 1960s, Lake Rotorua has experienced water quality problems associated with eutrophication (Fish 1969; Rutherford 1984; Rutherford *et al*. 1989; Burns 2009). This is the process of increased productivity 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; depleted dissolved oxygen concentrations in bottom waters; unsightly blooms of cyanobacteria that may produce toxins; odours, and; extirpation of species that are adapted to less productive waters (Carpenter *et al*. 1998). The primary metric used by Bay of Plenty Regional Council (BoPRC) to monitor trophic status is the Trophic Level Index (TLI), which integrates annual mean measurements of Secchi depth and concentrations of total nitrogen, total phosphorus and chlorophyll *a* (Burns *et al*. 1999).

In response to public dissatisfaction with water quality, Lake Rotorua has been identified as a national priority for restoration (Parliamentary Commissioner for the Environment 2006). In 2008, the Ministry for the Environment committed NZ$72.1 million towards improving water quality in Lake Rotorua and three other priority lakes. This funding was subsequently matched by BoPRC and Rotorua Lakes Council (RLC). The Lakes Rotorua and Rotoiti Action Plan (BoPRC 2009) outlines actions to achieve the Lake Rotorua water quality objective of an annual TLI of 4.2, which corresponds to the lower end of the eutrophic range (4–5; Burns *et al*. 1999). A range of actions is underway and an improvement in water quality has occurred in recent years relative to the early– and mid–2000s (Figure 1), which were characterised by frequent blooms of cyanobacteria during summer and autumn (Abell *et al*. 2012; Hamilton *et al*., 2015). As a result, annual TLI since 2011 has either been achieved or been very close to the target (Figure 1). This improvement has occurred in association with operations to dose aluminium sulphate (alum) near the mouths of the Utuhina and Puarenga streams, two major stream inflows to the lake. Aluminium ions in alum chemically bind with phosphate, removing it from the water column and thereby reducing the amount of phosphorus that is available for primary production. Dosing has been undertaken on a near–daily basis since operations began in the Utuhina Stream in mid–2006, with dosing also undertaken in the Puarenga Stream since 2010. Recent modelling work has shown that the TLI target would have been exceeded in recent years without the application of alum (Hamilton *et al*. 2015). Furthermore, this work indicates that alum is not only reducing dissolved reactive phosphorus concentrations in the inflows, but is also further reducing phosphorus concentrations in lake water as excess alum is transported downstream of the dosing plant.

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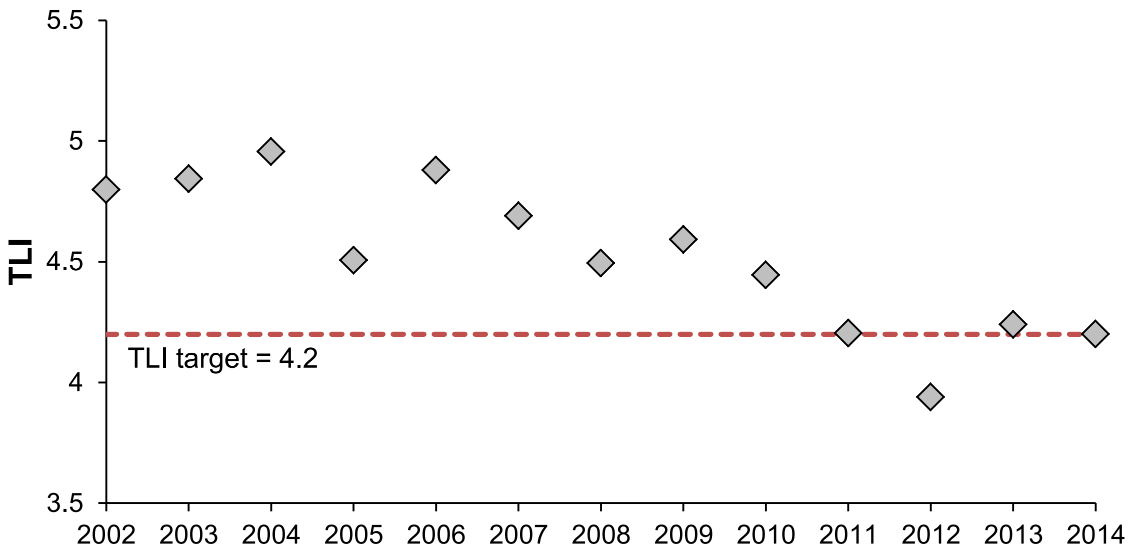

Figure 1 **Annual Trophic Level Index of Lake Rotorua.** Data for 2002–2012 are based on surface water samples only and thus values may differ slightly from by those used for BoPRC monitoring. Sources: 2002–2012 data (Abell *et al*. 2012); 2013 datum (Rotorua Te Arawa Lakes Programme 2014); 2014 datum (http://www.rotorualakes.co.nz/lake_rotorua_facts, accessed 29 May 2015).

Wastewater discharge in the lake catchment

Prior to 1991, municipal wastewater was discharged to the lake, contributing significant loads of nitrogen and phosphorus. Sewage–derived inputs were attributed to periods of water quality decline in the 1970s and 1980s (Rutherford 1984; Rutherford *et al*. 1989), with sewage inputs contributing to accumulation of nutrients (particularly phosphorus) in the bed sediments, in addition to inputs from other sources such as farmland. These accumulated nutrients contribute to internal loading as they are recycled within the water column, particularly during stratified periods in the summer when oxygen depletion in isolated bottom waters results in release of phosphate and ammonium from sediments (White *et al*., 1978; Burger *et al*., 2007). The magnitude of such internal loads of nitrogen and phosphorus during the 2000s was comparable to external loads from the lake catchment (Burger *et al*., 2007).

In 1991, discharge of treated municipal wastewater from Rotorua Wastewater Treatment Plant (WWTP) to the lake ceased, and spray–irrigation of treated wastewater commenced at the Land Treatment System (LTS), located in the Whakarewarewa Forest to the south of the lake (Map 1). The forest is in the Waipa Stream catchment, which is a tributary of the Puarenga Stream that inflows to Lake Rotorua. Rotorua Lakes Council currently has a Resource Consent to discharge 30 tonnes of nitrogen and three tonnes of phosphorus per annum via the LTS. Monitoring of the Waipa Stream shows that nitrogen loads frequently exceed the consent limit by a moderate amount, while phosphorus loads are typically well within the limit. Mean five-year loads for 2007–2011 were 35 t N/y and 1.7 t P/y (A. Lowe, pers. comm. 2013). Monitoring of the Puarenga Stream 2 km upstream of the lake since 1992 shows that dissolved inorganic nitrogen concentrations steadily increased over a period of approximately 10 years since operations began at the LTS, with current concentrations (~ 0.95 mg N/L) approximately 2.5–fold greater than those measured in 1992–1993. Compared with nitrogen, base flow phosphorus concentrations have remained relatively consistent in the Puarenga Stream, and have not exhibited a marked increase in response to the LTS operations. This indicates that in–stream removal processes such as adsorption attenuate the extent to which LTS phosphorus loads reach the lake.

Proposed options

The current Resource Consent for the LTS expires in 2021 and Rotorua Lakes Council is examining the use of an alternative wastewater disposal system. The proposed system involves various options of discharging treated wastewater directly to receiving waters (Mott MacDonald 2014). The options involve permutations of different:

1) enhancements to wastewater treatment;
2) wastewater discharge locations;
3) discharge arrangements.

Six main options for enhanced wastewater treatment are proposed, with several additional sub–options (Table 1). In total, these yield eleven permutations of final treated wastewater composition. The options involve varying grades of treatment to enhance the removal of nitrogen and phosphorus from the wastewater relative to current treatment performance.

Six potential treated wastewater discharge locations have been identified (Map 2):

1) three sites in the lower reach of the Puarenga Stream;
2) two sites along the shore of Lake Rotorua close to the mouth of the Puarenga Stream;
3) one site on the bed of the lake.

The potential discharge arrangements under consideration are:

1) direct discharge;
2) rock passage to direct discharge;
3) wetland;
4) rapid infiltration beds (RIB);
5) riparian/gabions;
6) monitoring pond.

Table 1 Proposed tertiary treatment options.

Option	Description	Sub-options	Details	Source
1	Base option	-	Upgrades to current tertiary treatment by addition of: flow balancing, P removal with chemical addition (alum) and UV disinfection.	Mott MacDonald (2014)
2	Base option + basic filtration	a. Disk filter b. Sand filter c. Membrane filter	Addition of filtration to remove solids, including particulate N and P.	Mott MacDonald (2014)
3	Base Option + filtration + denitrifying filter/bed	a. Denitrifying sand filter b. Sand filter + denitrifying carbon bed	Addition of filtration to remove solids, in addition to final denitrification step to convert dissolved inorganic N to atmospheric N gas.	Mott MacDonald (2014)
4	30 t N/y and 3 t P/y	-	Treatment processes configured to achieve maximum releases permitted under current Resource Consent conditions.	J. Bradley, pers. comm. 2015
5	30 t N/y and 1.5 t P/y	-	Treatment processes configured to achieve maximum N release and 50% of P release permitted under current Resource Consent conditions.	J. Bradley, pers. comm. 2015
6	Membrane bioreactor system rebuild	a b	No additional P treatment. + additional P treatment	K. Brian, pers. comm. 2015a

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Objectives of this study

The aim of this study is to assess the effects of the proposed wastewater discharge options on the water quality of Lake Rotorua and the lower reach of the Puarenga Stream. Specifically, the study examines:

- the potential instream ecological effects of discharging treated wastewater to the lower reaches of the Puarenga Stream;
- the potential effects of the proposed options on the trophic status of Lake Rotorua over multiple years;
- how mixing processes may affect how treated wastewater is diluted and dispersed throughout the lake, depending on the discharge location.

Methods

Overview

Three main techniques were used to inform the assessment:

1) *Mass balance calculations*.

   Dilution calculations were undertaken to quantify the proportion of the Puarenga Stream discharge that would comprise treated wastewater for a range of stream flows.

   Nutrient loads were estimated for the treatment options. These were compared with estimated background loads in the Puarenga Stream to quantify how loads in the stream are expected to change, and to inform assessment of potential in–stream effects of nutrient enrichment. Loads were also compared with external load reduction targets for the lake to provide catchment–scale context. Estimated loads were subsequently used as forcing data to 'drive' the water quality model introduced below.

2) *One–dimensional (1–D) lake modelling*.
   A numerical model was configured to simulate the water quality effects of discharging treated wastewater, relative to a baseline period that represents current conditions. The lake was conceptualised as a single vertical profile in the model, i.e., vertical differences in water quality were modelled but horizontal variations were not. This 1–D assumption permitted lake processes to be sufficiently simplified so that potential effects on lake trophic status over time scales of multiple years could be examined.
3) *Three–dimensional (3–D) lake modelling.*
   A 3–D hydrological model was configured to examine the mixing processes that control how simulated treated wastewater inputs are diluted and dispersed within the lake.

Mass balance calculations to estimate in–stream nutrient loads and concentrations

Dilution calculations

The proportion of the Puarenga Stream discharge that would comprise treated wastewater was calculated for a representative range of stream discharge conditions for a scenario involving discharge of treated wastewater at a constant rate to the stream. Calculations were undertaken using hourly discharge measurements in the stream for the period 2005 through 2015 (see Section 0 for further details). Proportions were calculated by dividing the projected wastewater discharge rate (0.2756 m^3/s) by the sum of this rate and the stream discharge. Proportions were then expressed as a percentage.

Treated wastewater nutrient loads

Information used to calculate the nutrient loads associated with each proposed treatment option is presented in Table 2. The predicted composition of the wastewater reflects upgrades/replacement of current tertiary treatment processes at the WWTP that will result in a range of improvements to the final wastewater quality. Predicted wastewater composition reflects post–treatment nutrient concentrations that result from the various options to upgrade and/or replace current tertiary treatment processes at the WWTP.

Options 1 to 3 represent treatment options that were identified in a feasibility study of alternatives to land disposal (Mott MacDonald 2014). Options 4 and 5 were configured to examine the effects of additional treatment options that were discussed at a Technical Advisory Group meeting on 28 May 2015 (J. Bradley, pers. comm. 2015). Specifically, these options comprise discharge of either: 30 t N/y and 3 t P/y (Option 4), or 30 t N/y and 1.5 t P/y (Option 5), and represent the implementation of an alternative LTS to the current site. These scenarios were configured by setting the dissolved inorganic nitrogen and phosphorus concentrations in treated wastewater equal to those of Options 1 and 2 (Table 2), and then varying the concentrations of the other fractions to achieve the desired loads. Option 6 represents predicted loads corresponding to conversion of the present Bardenpho system at the WWTP to a membrane bioreactor (MBR) system (i.e., all wastewater treated by MBR), with two alternative levels of phosphorus treatment (K. Brian, pers. comm. 2015a). This option was included in response to a request made at a Technical Advisory Group meeting on 16 June 2015 (S. Pauli, pers. comm 2015).

Details of any temporal variability in either wastewater discharge or composition were not provided, and therefore the assessment was based on the assumption that wastewater composition will remain constant.

Table 2 **Predicted final treated wastewater composition associated with each tertiary treatment option.**

Option	Sub-option	Discharge (ML/d)	Final effluent composition (mg/L)							
			TP	DRP	PP	TN	PON	DON	NO_3-N	NH_4-N
Option 1 (base option)		23.81	0.72	0.10	0.62	5.44	1.07	1.09	2.99	0.29
Option 2 (base + basic filtration)	a. Disc filter	23.81	0.37	0.10	0.27	4.86	0.49	1.09	2.99	0.29
	b. Sand filter	23.81	0.20	0.10	0.10	4.62	0.25	1.09	2.99	0.29
	c. Membrane filter	23.81	0.10	0.10	0	4.37	0	1.09	2.99	0.29
Option 3 (base + basic filtration + denitrifying filtration)	a. Denitrifying sand filter	23.81	0.20	0.10	0.10	2.63	0.25	1.09	1.00	0.29
	b. Sand filter + denitrifying carbon bed	23.81	0.20	0.10	0.10	3.63	0.25	1.09	2.00	0.29
Option 4 (improve existing plant to achive 30t N/y and 3 t P/y)		23.81	0.34	0.10	0.24	3.44	0.08	0.08	2.99	0.29
Option 5 (improve existing plant to achive 30t N/y and 1.5 t P/y)		23.81	0.17	0.10	0.07	3.44	0.08	0.08	2.99	0.29
Option 6 (full MBR)	a. P treated to 3.0 t P/y	23.81	0.35	0.35	0	3.53	0	0.94	2.10	0.50
	b. P treated to 1.5.0 t P/y	23.81	0.175	0.175	0	3.53	0	0.94	2.10	0.50

Puarenga Stream background nutrient loads

Nutrient loads in the Puarenga Stream were estimated for the baseline period of 2007 through 2014. Reasons for selection of this baseline period are discussed in Section 0 below.

Discharge

Discharge data for the Puarenga Stream were provided by BoPRC. Data for the period 2007 through 2010 were collected at the FRI gauge situated 2.1 km upstream of Lake Rotorua. Data for the period 2011 through 2014 were collected at the SH 30 gauge situated 0.8 km further downstream. There are no tributaries between the gauges and the data from the two sites were considered directly comparable. Discharge was recorded every 15 minutes (see BoPRC 2007 for quality assurance details). Measured data were available for 98.2% of the monitoring period (Table 3). All gaps in the record were filled using the following linear relationship (r^2 = 0.75, *RMSE* = 0.62 m^3/s):

$$Q_{Puarenga} = 1.01 \cdot Q_{Utuhina} + 1.1301$$

where $Q_{Puarenga}$ is mean hourly discharge (m^3/s) in the Puarenga Stream and $Q_{Utuhina}$ is mean hourly discharge (m^3/s) in the Utuhina Stream, measured at the Depot Street gauge. Data are shown in Figure 2.

Table 3 **Proportion of time (%) when discharge measurements are not available for the Puarenga Stream.**

Year	%	Gaps > 1 day
2007	2.3	~3 days (July), ~2 days (September)
2008	11.5	~27 days (July), ~4 days (September)
2009	0.1	
2010	0.0	
2011	0.0	
2012	5.5	~15 days (July/August)
2013	0.0	
2014	0.0	

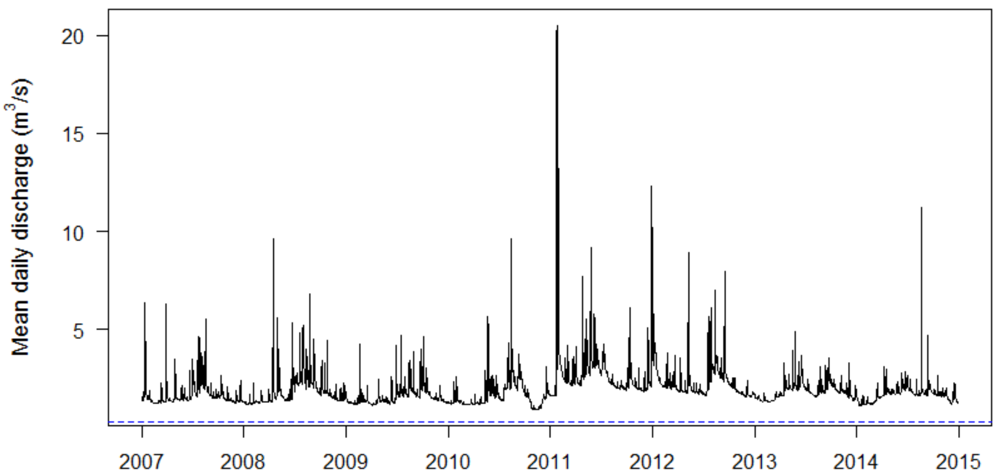

Figure 2 **Puarenga Stream mean daily discharge, 2007–2014. The dashed blue line denotes the mean discharge of treated wastewater for reference.**

Nutrient concentrations

Water quality data used to estimate baseline nutrient loads were primarily obtained from BoPRC. These data are based on monthly grab samples collected at the FRI gauge (now inactive) during 2007 through 2014. Additional data collected following storm events (Abell *et al*. 2013) were used to derive relationships between discharge and concentrations of nutrient fractions that are correlated with discharge.

Table 4 summarises the methods used to estimate baseline hourly mean nutrient concentrations. Linear interpolation of monthly measurements was used to estimate daily concentrations of nitrate, ammonium and dissolved reactive phosphorus. This was deemed suitable as concentrations of dissolved nutrient fractions are generally invariant with discharge in the Puarenga Stream. Concentrations of nitrate are a partial exception as they typically exhibit decreases during high discharge (dilution effect), although these are generally balanced by subsequent 'pulses' of elevated concentrations that are of approximate equal magnitude to the prior decreases.

For periods of hourly mean discharge > 3.0 m^3/s, concentrations of both particulate phosphorus and the non–dissolved inorganic nitrogen (DIN) fraction (i.e., TN-DIN) were estimated using linear (log_{10}–log_{10} space) relationships between concentration and discharge. Such relationships were weaker for discharge < 3.0 m^3/s, and thus linear interpolation was used to estimate concentrations of these analytes for these periods. The sum of total dissolved phosphorus minus dissolved reactive phosphorus was assumed to be zero (i.e., dissolved organic phosphorus was assumed to be negligible).

Table 4 **Summary of methods used to derive baseline hourly mean nutrient concentrations in the Puarenga Stream for the period 2007–2014.**

Analyte	Estimation method	Notes
PO_4-P	Linear interpolation of monthly measurements collected by BoPRC.	Missing measurements (*n* = 3) replaced with the mean of concentrations measured in that year.
PP	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q > 3 m^3/s: Derived from a linear relationship between $\log_{10}$Q and $\log_{10}$[PP] with correction for log-transformation bias (Ferguson 1986).	Measured PP was calculated as TP minus PO_4-P. Relationship was based on data presented in Abell *et al*. (2013), collected when dischage was 3.0 to 15.6 m^3/s (maximum PP = 0.44 mg/L). Maximum mean hourly discharge for 2007-2014 was 30.4 m^3/s; maximum modelled mean hourly [PP] was 0.51 mg/L.
TP	By calculation.	PO_4-P + PP
NO_x-N	Linear interpolation of monthly measurements collected by BoPRC.	Missing (n = 2) and anomalously low (*n* = 3) measurements replaced with the mean of concentrations measured in that year.
NH_4-N	Linear interpolation of monthly measurements collected by BoPRC.	Missing measurements (*n* = 4) replaced with the mean of concentrations measured in that year.
(TN-DIN)	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q > 3 m^3/s: Derived from a linear relationship between $\log_{10}$Q and $\log_{10}$[TN-DIN] with correction for log-transformation bias (Ferguson 1986).	This fraction includes dissolved (i.e., filterable) organic nitrogen (DON) and particulate nitrogen (PN).
DON	0.40 × (TN-DIN)	Based on the mean proportion of (TN-DIN) that comprised (TDN-DIN) in 80 samples collected during three storm events (Abell *et al*. 2013). There was no correlation between this proportion and Q.
PN	0.60 × (TN-DIN)	Based on the mean proportion of (TN-DIN) that comprised (TN-TDN) in 80 samples collected during three storm events (Abell *et al*. 2013). There was no correlation between this proportion and Q.
TN	By calculation.	NO_x-N + NH_4-N + DON + PN

Calculations to estimate in–stream loads and concentrations

Daily nutrient loads in the Puarenga Stream and the various proposed treated wastewater discharges were calculated as

$$L_x = K \cdot \sum_{i=1}^{24} \widehat{C_{x}}_i \cdot Q_i$$

where L_x is load (kg/d) of nutrient *x*, *K* is a unit conversion factor, $\widehat{C_{x}}_i$ is estimated mean concentration (mg/L) of nutrient *x* during hour *i*, and Q_i is mean discharge (m^3/s) for hour *i*. Daily loads were summed to calculate annual loads (t/y).

Loads for individual treatment options were compared with the nutrient reduction targets that have been set for Lake Rotorua (BoPRC 2009), in addition to the baseline loads in the Puarenga Stream to place the loads in context of downstream waters.

Daily mean nutrient concentrations that corresponded to combined Puarenga Stream and wastewater loads were estimated by dividing combined loads by the combined discharge. Thus, these estimated concentrations do not reflect any non–conservative processes such as uptake by plants or denitrification. The potential for such processes to influence nutrient concentrations in the Puarenga Stream downstream of the proposed stream discharge locations is limited given the very short length (and thus residence time) of this reach (Map 2).

Comparison of concentrations with values designated in the NPS 2014 to assess in–stream effects on Ecosystem Health

The National Policy Statement for Freshwater Management 2014 (New Zealand Government 2014) designates values for a range of attributes that correspond to different Ecosystem Health Attribute States. Attribute States range from *A* (high ecosystem health) to *D* (low ecosystem health). Values corresponding to the 'National Bottom Line' have also been defined, which correspond to the minimum acceptable state that has been set by the government (designated by the separation of *C* – *D* attribute states. Separate values have been defined for different aquatic ecosystem types. For rivers, values have been defined for the following attributes: nitrate (with respect to toxicity effects), ammonium (with respect to toxicity effects), dissolved oxygen, *E. coli* and periphyton.

Potential effects of the proposed options in relation to nitrate, ammonium and dissolved oxygen concentrations were assessed quantitatively by comparing baseline concentrations in the Puarenga Stream with estimated concentrations following addition of separate wastewater discharges corresponding to the six treatment options (Table 2). These differences were then considered in the context of Ecosystem Health Attribute State values for these analytes, which are reproduced in Table 5, Table 6 and Table 7. For nitrate and ammonium, these assessments were based on the time series of daily mean concentration data that were derived for each modelling scenario (see Section 0). For dissolved oxygen, the assessment was based on comparing monthly measurements collected by BoPRC in the lower Puarenga Stream with concentrations that were estimated for corresponding days for a scenario of anoxic wastewater discharge (i.e., a worst case scenario). Concentrations for this scenario were estimated using daily mean discharge data for the Puarenga Stream (Figure 2) and assuming conservation of mass. Degassing due to temperature effects was not considered. Concentrations were compared with the one–day minimum values that are specified in the NPS, although it is acknowledged that the spot measurements cannot be assumed to be minima.

Potential effects of the proposed options in relation to *E. coli* concentrations were assessed semi–quantitatively by determining the corresponding Ecosystem Health Attribute States (see Table 8) for each year in the baseline period using data collected by BoPRC, and considering these in the context of likely treated wastewater composition.

Potential effects of the proposed options in relation to periphyton were considered qualitatively, based on consideration of the potential for discharged wastewater to cause bottom–up effects on periphyton as a consequence of changes to nutrient concentrations.

Table 5 **Nitrate nitrogen concentrations (mg N/L) corresponding to River Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management in relation to nitrate toxicity (New Zealand Government 2014).**

Attribute state	Numeric attribute state		Narrative attribute state
	Annual median	Annual 95th percentile	
A	≤ 1.0	≤ 1.5	High conservation value system. Unlikely to be effects even on sensitive species.
B	> 1.0 and ≤ 2.4	>1.5 and ≤ 3.5	Some growth effect on up to 5% of species.
C	> 2.4 and ≤ 6.9	> 3.5 and ≤ 9.8	Growth effects on up to 20% of species (mainly sensitive species such as fish).
National bottom line	6.9	9.8	No acute effects.
D	> 6.9	> 9.8	Impacts on growth of multiple species, and starts approaching acute impact level (i.e. risk of death) for sensitive species at higher concentrations (> 20 mg N/L).

Table 6 **Ammoniacal nitrogen concentrations (mg N/L) corresponding to River Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management in relation to ammonia toxicity (New Zealand Government 2014).**

Attribute state	Numeric attribute state		Narrative attribute state
	Annual median	Annual maximum	
A	≤ 0.03	≤ 0.05	High conservation value system. Unlikely to be effects even on sensitive species.
B	> 0.03 and ≤ 0.24	>0.05 and ≤ 0.40	Some growth effect on up to 5% of species.
C	> 0.24 and ≤ 1.3	> 0.40 and ≤ 2.20	Growth effects on up to 20% of species (mainly sensitive species such as fish).
National bottom line	1.3	2.2	No acute effects.
D	> 1.30	> 2.20	Impacts on growth of multiple species, and starts approaching acute impact level (i.e. risk of death) for sensitive species at higher concentrations (> 20 mg N/L).

Table 7 **Dissolved oxygen concentrations corresponding to River Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management in relation to ammonia toxicity (New Zealand Government 2014).**

Attribute state	Numeric attribute state		Narrative attribute state
	7-day mean minimum (1 Nov to 30 April)	1-day minimum (1 Nov to 30 April)	
A	≥ 8.0	≥ 7.5	No stress caused by low dissolved oxygen on any aquatic organisms that are present at matched reference (near-pristine) sites.
B	≥ 7.0 and ≤ 8.0	≥ 5.0 and < 7.5	Occasional minor stress on sensitive organisms caused by short periods (a few hours each day) of lower dissolved oxygen. Risk of reduced abundance of sensitive fish and macroinvertebrate species.
C	≥ 5.0 and ≤ 7.0	≥ 4.0 and < 5.0	Moderate stress on a number of aquatic organisms caused by dissolved oxygen levels exceeding preference levels for periods of several hours each day. Risk of sensitive fish and macroinvertebrate species being lost.
National bottom line	5.0	4.0	
D	< 5.0	< 4.0	Significant, persistent stress on a range of aquatic organisms caused by dissolved oxygen exceeding tolerance levels. Likelihood of local extinctions of keystone species and loss of ecological integrity.

Table 8 ***E. coli* concentrations corresponding to River Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management in relation to ammonia toxicity (New Zealand Government 2014).**

Attribute state	Numeric attribute state 7-day mean minimum (1 Nov to 30 April; /100 mL)	Statistic	Narrative attribute state
A	≤ 260	Annual median	People are exposed to a very low risk of infection (less than 0.1% risk) from contact with water during activities with occasional immersion and some ingestion of water (such as wading and boating).
		95th percentile	People are exposed to a low risk of infection (up to 1% risk) when undertaking activities likely to involve full immersion.
B	> 260 ≤ 540	Annual median	People are exposed to a low risk of infection (less than 1% risk) from contact with water during activities with occasional immersion and some ingestion of water (such as wading and boating).
		95th percentile	People are exposed to a moderate risk of infection (less than 5% risk) when undertaking activities likely to involve full immersion. 540 / 100 mL is the minimum acceptable state for activities likely to involve full immersion.
C	> 540 ≤ 1000	Annual median	People are exposed to a moderate risk of infection (less than 5% risk) from contact with water during activities with occasional immersion and some ingestion of water (such as wading and boating). People are exposed to a high risk of infection (greater than 5% risk) from contact with water during activities likely to involve immersion
National Bottom Line	1000	Annual median	
D	> 1000	Annual median	People are exposed to a high risk of infection (greater than 5% risk) from contact with water during activities with occasional immersion and some ingestion of water (such as wading and boating).

One–dimensional lake modelling

Model selection

The 1–D model DYRESM–CAEDYM was selected. The model comprises a hydrodynamic model (DYRESM[2]) that is coupled to a water quality model (CAEDYM[3]). DYRESM predicts the vertical variations of temperature and density in lakes such as Lake Rotorua 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 *et al*. 2013; Imerito 2013).

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, and it has now been applied to Lake Rotorua for numerous years to understand in–lake processes and inform management decisions. Specifically, the model has previously been used to predict how Lake Rotorua water quality will respond: to reductions in external and internal loads (Burger *et al*. 2008); land use and climate changes (Hamilton *et al*. 2012), and; alum dosing (Hamilton *et al*. 2015). Thus selecting DYRESM–CAEDYM meant that this

[2] DYnamic REservoir Simulation Model
[3] Computational Aquatic Ecosystem Dynamics Model

study could benefit from the extensive body of previous work that has been undertaken to configure and calibrate the model to reflect the characteristics of Lake Rotorua.

Such process–based modelling enables the simulation of a wide range of variables at high temporal resolution to provide detailed understanding of major processes in the lake. The use of process–based models allows for greater certainty in the outcome of simulated 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 for model derivation. A constraint of this approach, however, is that such process–based models are "data hungry"; they require information for a large number of forcing variables such as those that relate to weather, morphometry and inflows, in addition to field measurements of simulated variables to assist model calibration. In this regard, Lake Rotorua is a suitable candidate as it has been relatively extensively monitored and there exists a large body of data to use for model configuration.

Model overview

DYRESM simulates multiple layers of variable thickness that change dynamically to accommodate changes in lake volume. DYRESM is primarily affected by surface exchanges of heat, mass and momentum, and resolves the vertical distributions of temperature, salinity, and density in lakes and reservoirs (Imerito 2013).

CAEDYM simulates fluxes that regulate biogeochemical variables such as nutrient concentrations and phytoplankton biomass (Hipsey *et al*. 2013). The model includes representations of cycling processes for carbon, nitrogen, phosphorus, dissolved oxygen and inorganic suspended sediments. The state variables that are simulated within CAEDYM can be adjusted depending on the study objectives and the availability of measured data for calibration. Accordingly, the following three generic groups of phytoplankton were represented in CAEDYM: 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[4]:

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

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 a Monod equation which required the user to assign nutrient half saturation constants to each phytoplankton group. Photo–inhibition was not represented. Simulated phytoplankton biomass can be dynamically converted to output estimates of chlorophyll *a* concentrations in the water column and summed for each of the three phytoplankton groups for different depths, at each model time step.

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

Conceptual diagrams of the representations of nitrogen and phosphorus cycling within CAEDYM are shown in Figure 3. Each process in the figure was explicitly represented in CAEDYM. Higher fauna and macrophytes were not considered.

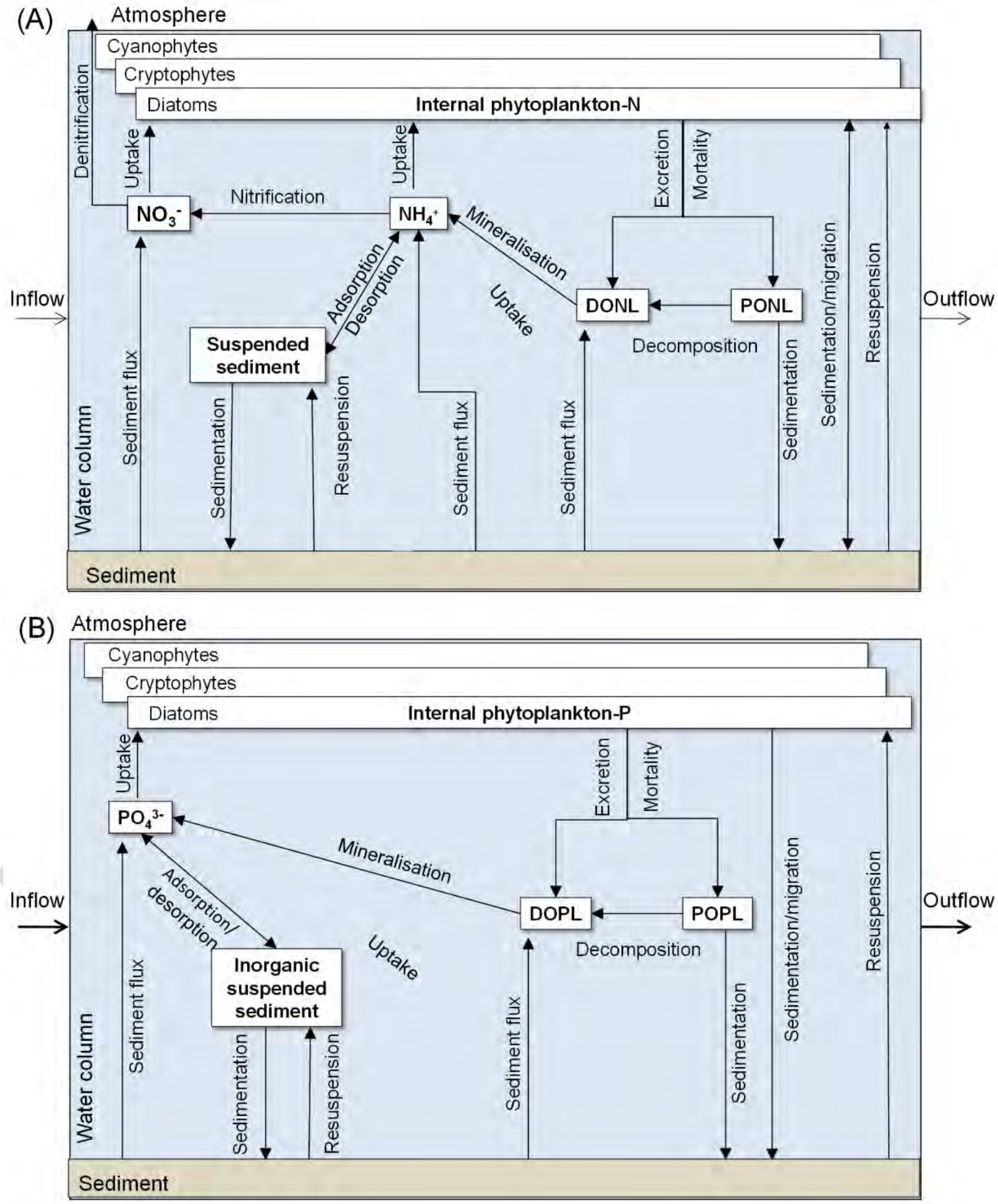

Figure 3 **Conceptual diagrams of the cycling of nitrogen (A) and phosphorus (B) within the water quality model (CAEDYM). DONL, labile dissolved organic nitrogen; PONL, labile particulate organic nitrogen; DOPL, labile dissolved organic phosphorus; POPL, labile particulate organic phosphorus.**

Model simulation, calibration and validation periods

An eight year baseline period of 2007–2014 was selected for the 1–D water quality modelling. This period encompasses the most recent period for which the necessary forcing data are available and it was deemed important to select a period that was as recent as possible to help to assess effects relative to current water quality. It was also desirable to select a baseline period that spanned multiple years so that it encompassed a range of forcing conditions (particularly weather) that were representative of current conditions. The first year was selected as 2007 because this corresponds to the first full year during which alum dosing was undertaken (see Section 0). Alum dosing has had a significant effect on lake water quality (Hamilton *et al*. 2015) and it was desirable to constrain the modelling period to include only the period when alum dosing was undertaken. This is because the effects of alum dosing are currently represented 'statically' in the DYRESM–CAEDYM configuration by adjusting parameters that control sediment nutrient release rates and particulate matter diameter to reflect nutrient adsorption and sediment flocculation caused by alum (discussed further in Section 0 below). Thus, the need to use a separate model configuration for periods with and without alum dosing currently inhibits the use of the model to simulate a single period that includes years both before and after 2007.

The calibration period was defined as 2007–2010 and the validation period was 2011–2014. These years are not based on calendar years; annual Trophic Level Index of the lake is calculated using data collected between 1 July and 30 June, and therefore the first year of the simulation spanned the period of 1 July 2006 to 30 June 2007.

Model performance for each period was quantified by comparing modelled and measured values of the following water quality parameters: temperature, dissolved oxygen, nutrients and chlorophyll *a* (Table 9). Comparisons were made with measured data collected at different depths by BoPRC as part of a monthly monitoring programme. For each sampling date, a mean of measurements collected at the two mid lake sites that are sampled by BoPRC ('Site 2' and 'Site 5') was calculated, and these mean values were used in all comparisons with model results.

The model was run with a time step of one day. Water quality parameters were initialized based on the most recent monitoring data that corresponded to the start date. A one year 'spin up' period was modelled prior to each simulation. This was configured by 'looping' forcing data for 2007, and model outputs from this period were not considered during analysis.

Table 9 Model performance statistics.

Abbreviation	Statistic	Details	Equation
r	Pearson product moment correlation	Measures the strength of the correlation between modelled and measured data, i.e. how 'in phase' the two signals are. Vales range from -1 (perfect negative correlation) to 1 (perfect positive correlation).	$\frac{\sum_{i=1}^{n}(o_i-\bar{o})\times(m_i-\bar{m})}{\sqrt{\sum_{i=1}^{n}(o_i-\bar{o})}\times\sqrt{\sum_{i=1}^{n}(m_i-\bar{m})}}$
RMSE	Root mean square error	A measure of the magnitude of the error between modelled and measured data which is disproportionately affected by large errors.	$\sqrt{\frac{\sum_{i=1}^{n}(m_i-o_i)^2}{n}}$
MAE	Mean absolute error	Measures the average error, irrespective of whether the model under– or over–predicts measurements.	$\frac{\sum_{i=1}^{n}\lvert(m_i-o_i)\rvert}{n}$

Model configuration

Bathymetry

Lake bathymetry was represented using a lake–area relationship provided by BoPRC. Maximum lake depth prescribed by this relationship was 25 m and therefore a small isolated hole present in the lake (depth ≈ 50 m) was ignored.

Meteorological input data

Meteorological data were obtained from records collected at the Rotorua Airport automatic weather station (AWS), located on the south–eastern shore of the lake (Map 1). Data collected prior to 2013 were obtained from the National Climate Database administered by NIWA (http://cliflo.niwa.co.nz/); data collected since January 2013 were provided by MetService. Mean daily data were collated for the following variables as inputs to the model:

- rainfall (m);
- wind speed (m/s);
- air temperature (° C);
- shortwave solar radiation (W/m^2);
- vapour pressure (hPa).

Daily cloud cover was estimated based on the difference between observed daily mean short–wave solar radiation and estimated theoretical minima and maxima (Luo *et al*. 2010).

Hydrologic input data

The model configuration included representations of daily mean discharge for nine major streams and nine minor streams (Table 10). Where available, stream discharge data were obtained from near–continuous records from hydrometric gauges that were operational throughout the modelling period. This was the case for the Ngongotaha Stream (operated by NIWA), and the Puarenga, Waingaehe and Utuhina streams (operated by BoPRC; see BoPRC 2007). For streams without a permanent gauge, mean discharge was estimated based on monthly measurements of discharge that were either collected by BoPRC, presented in other studies (Rutherford *et al*. 2008) or used in previous modelling applications

(Abell and Hamilton 2015). Daily fluctuations of discharge in such streams were then modelled based on fluctuations measured in comparable streams.

Outflow via the Ōhau Channel (the only outlet) was configured based on daily mean measured discharge provided by NIWA.

Ungauged inflows to the lake were estimated as the residual term in a water balance constructed for the lake. Thus

$$Ungauged\ = (Q_{\bar{O}hau} + E + \Delta S) - \left(Q_{inflow} + rainfall\right)$$

where *Ungauged* is mean daily ungauged inflow (m^3/s), $Q_{\bar{O}hau}$ is mean daily discharge of the only lake surface outflow (m^3/s), ΔS is mean daily rate of change in lake storage (m^3/s) due to water level change (provided by NIWA, measured at the Mission Bay monitoring station), Q_{inflow} is mean daily stream discharge (m^3/s) and *rainfall* is mean 15–day daily rainfall (m^3/s) based on measurements at Rotorua Airport applied across the lake.

This term therefore reflects error in the estimation of the other terms in the water balance, in addition to unmonitored inputs such as groundwater flow to the bed of the lake, overland flow and additional minor streams. This term was smoothed by calculating a running 15–day average to remove most negative values. A number of small negative values remained after this smoothing process; these were set to zero and a constant sum was added to the other values in the time series to account for this. Finally, it was necessary to increase this inflow by 18% (mean daily increase of 0.62 m^3/s) to maximise the goodness of fit between modelled and measured water levels (Table 10). It is uncertain why this increase was necessary; it may relate to minor differences in either evaporation rates, or the changes in storage calculated by the model and those estimated in the water balance. It is common to undertake such adjustments in lake modelling studies to correct minor discrepancies in modelled water levels.

Hourly mean evaporation rate (E; L/s) was calculated based on Fischer *et al*. (1979):

$$E = \frac{A\left(\frac{-0.622}{P} C_L\, \rho_a\, L_E U(e_a - e_s)(T_{surf})\right)\Big)}{L_v}$$

where A is the area of the lake (m^2), C_L is the latent heat transfer coefficient for wind speed (0.0013), ρ_a is air density (kg/m), L_E is the latent heat of evaporation of water (2,453,000 J/kg), U is measured wind speed (m/s), e_a is the vapour pressure of the air (Pa), e_s is the saturated vapour pressure of the air (Pa) corresponding to the lake water surface temperature (°C), P is the atmospheric pressure (Pa), L_v is the latent heat of vaporisation (2 260 000 J/kg) and T_{surf} is the surface water temperature (°C) estimated using a relationship established between day of the year and historic measurements. A value of 0 was substituted where E < 0 as the models do not simulate condensation effects.

e_s was calculated by the Magus–Tetens formula (Hodges and Dallimore 2011):

$$e_s(T_{0.5}) = 100 \exp\left[2.3026\left(\frac{7.5\,T_{0.5}}{T_{0.5} + 237.3}\right) + 0.758\right]$$

Table 10 Summary of how discharge was configured for the inflows and outflow.

Inflow type	Inflow	Mean discharge (m^3/s)	Details	Source
Major streams	Awahou Stream	1.69	The mean discharge was set to the mean of monthly instantaneous gaugings during 2005 through 2012 (*n* = 86). Temporal fluctuations were then imposed based on fluctuations measured in the Ngongotaha Stream.	BoPRC
	Hamurana Stream	2.57	This is a groundwater spring-dominated stream. Monthly (approximate) instantaneous gaugings were interpolated for the period 2007 through 2012 (*n* = 51). Discharge set to the mean of gaugings (2.558 m^3/s) during 2012 through 2014.	BoPRC
	Ngongotaha Stream	1.84	Based on measured data (99.9% of record) at SH 30 gauge. One gap of 89 h was filled with mean value of preceding and subsequent days.	NIWA
	Puarenga Stream	1.95	Based on measured data (97.2% of record) at FRI gauge (2007 to 2010) and SH30 gauge (2010 to 2014). Gaps were replaced with modelled data (2.8% of record) based on linear relationship (r^2 = 0.75) with measurements for Utuhina Stream.	BoPRC
	Utuhina Stream	1.81	Based on measured data (92.8% of record) at Depot Street gauge. Gaps were replaced with modelled data (7.2% of record) based on linear relationship (r^2 = 0.67) with measurements for Puarenga Stream.	BoPRC
	Waingaehe Stream	0.27	Based on measured data (99.5% of record) at SH30 gauge. Gaps were replaced with mean values of adjoining measurements (0.5% of record).	BoPRC
	Waiohewa Stream	0.38	As for the Awahou Stream. Mean discharge was estimated based on a sample of 70 measurements.	BoPRC
	Waiowhiro Stream	0.31	As for the Awahou Stream. Mean discharge was estimated based on a sample of 78 measurements.	BoPRC
	Waiteti Stream	1.23	As for the Awahou Stream. Mean discharge was estimated based on a sample of 76 measurements.	BoPRC
Minor Streams	Lynmore Stream	0.05	The long-term mean discharge was set to the mean of monthly instantaneous gaugings during 2005 through 2012 (*n* = 71). Temporal fluctuations were then imposed based on fluctuations measured in the Waingaehe Stream.	BoPRC
	Motutara (geothermal seep)	0.04	A constant discharge was assigned	
	Rotokawa 1 (geothermal seep)	0.02	A constant discharge was assigned	
	Rotokawa 2 (geothermal seep)	0.04	A constant discharge was assigned	
	Hauraki Stream	0.01	The long-term mean discharge was set to the mean discharge reported in Rutherford *et al*. (2008). Temporal fluctuations were then imposed based on fluctuations measured in the Waingaehe Stream.	
	Waitawa 1	0.06	The long-term mean discharge in these four streams was calculated from the mean discharge reported in Rutherford *et al*. (2008) for 'minor' catchments (0.4 m^3/s), minus the mean discharge for the other five minor streams. Temporal fluctuations were then imposed based on fluctuations measured in the Waingaehe Stream.	
	Waitawa 2	0.06		
	Waimehia Drain	0.06		
	Waiowhiro 2/ Waikuta	0.06		
Outflow	Ōhau Channel	18.50	Daily mean discharge was provided by NIWA.	NIWA
Ungauged	Ungauged	4.11	Based on the residual quantity in the water balance (mean = 3.49 m^3/s), plus 18% to maximise goodness of fit between modelled and measured water levels.	-

Inflow water quality

Temperature and dissolved oxygen

Hourly mean temperature (°C) of precipitation was set to lake surface water temperature, estimated using an empirical relationship between historical measurements and day of year.

Hourly mean temperatures (°C) of remaining surface inflows (Ts) were estimated using an empirical model described by Mohseni *et al*. (1998):

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

where T_a is the average daily air temperature measured at Rotorua Airport AWS (°C), α is the maximum historic measured stream temperature (°C) and both γ and β are dimensionless parameters. Parameters γ and β were determined by fitting the model to historic spot measurements of stream temperature provided by BoPRC (n = 65 – 96) and minimising root mean squared error. Measured data were not available for most minor streams and subsequently T_s for one stream (Lynmore) was assigned to five minor streams.

Dissolved oxygen (DO) concentrations of all inflows were assumed to be 100% saturated based on estimated water temperature. Accordingly, DO concentrations were estimated using the following equation derived by Mortimer (1981)

$$DO = \exp\left(7.71 - 1.31\ ln(T_s + 45.93)\right)$$

where DO is dissolved oxygen at saturation (mg/L).

Nutrient and suspended sediment concentrations

Major streams

Nutrient and inorganic suspended sediment (ISS) concentrations were assigned to stream inflows based on measured data. Data were primarily obtained from a dataset collected by BoPRC during routine monthly sampling. Additional data obtained from a study undertaken of two major stream inflows during 2010–2012 (Abell *et al*. 2013) were used to assign concentrations during storm flows.

The nine major stream inflows (Map 1) were represented separately in the model. Details of how nutrient and ISS concentrations were assigned to these streams are presented in Table 11. For completeness, the table repeats details for the Puarenga Stream that are described above in Section 0. Briefly, daily nutrient concentrations were typically assigned by linearly interpolating monthly measurements. Exceptions were concentrations of ISS, particulate phosphorus (PP) and the non–dissolved inorganic nitrogen (DIN) fraction of total nitrogen (TN) pool (i.e., TN-DIN) in the Ngongotaha, Puarenga and Utuhina streams. Concentrations of these analytes have been shown to positively correlate with discharge (Hoare 1982; Rutherford 2008), and failure to account for this effect results in marked underestimation of long–term loads to the lake (Abell *et al*. 2013). Such storm loads were quantified for the Ngongotaha, Puarenga and Utuhina streams as these have the greatest proportion of annual nitrogen and phosphorus loads transported in storm flow (Rutherford 2008). Storm loads were not quantified for other streams as storm fluxes are less dominant for these streams, due to relatively greater dominance of groundwater inputs. In addition, there were insufficient data to robustly define relationships between concentrations and discharge for these streams, and therefore the potential for increasing error by estimating such relationships was deemed to outweigh any error associated with underestimating storm fluxes.

Table 11 Methods to assign nutrient concentrations to major stream inflows. See glossary for definitions of abbreviations.

Analyte	Stream	Estimation method	Notes
PO_4-P	All	Linear interpolation of monthly measurements collected by BoPRC.	Missing/anomalous measurements replaced with the mean of concentrations measured in adjoining months.
PP	Puarenga	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q > 3 m^3/s: Derived from a linear relationship between $\log_{10}$Q and $\log_{10}$[PP] for the Puarenga Stream with correction for transformation bias (Ferguson 1986).	Relationship was based on data presented in Abell *et al*. (2013), collected from the Puarenga Stream when dischage was 3.0 to 15.6 m^3/s (maximum [PP] = 0.44 mg/L; *n* = 174; r^2 = 0.19). Maximum modelled mean daily [PP] was 0.38 mg/L.
	Ngongotaha and Utuhina	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q > 3 m^3/s: Derived from a linear relationship between $\log_{10}$Q and $\log_{10}$[PP] for the Ngongotaha Stream with correction for transformation bias (Ferguson 1986).	Relationship was based on data presented in Abell *et al*. (2013), collected when dischage was 3.0 to 22 m^3/s (maximum [PP] = 0.44 mg/L; *n* = 44; r^2=0.77). Maximum modelled mean daily [PP] was 0.53 mg/L and 0.44 mg/L.
	Awahou, Waiteti, Waingaehe, Waiowhiro, Waiohewa, Hamurana	Linear interpolation of monthly measurements collected by BoPRC.	Measured PP was calculated as TP minus PO_4-P.
TP	All	By calculation.	PO_4-P + PP
NO_x-N	All	Linear interpolation of monthly measurements collected by BoPRC.	Missing and anomalous (e.g., > TN) measurements replaced with the mean of concentrations measured in adjoining months.
NH_4-N	All	Linear interpolation of monthly measurements collected by BoPRC.	Missing measurements replaced with the mean of concentrations measured for adjoining months.
(TN-DIN)	Puarenga	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q > 3 m^3/s: Derived from a linear relationship between [(TN-DIN)] and $\log_{10}$Q for the Puarenga Stream with correction for log-transformation bias (Ferguson 1986).	This fraction includes dissolved (i.e., filterable) organic nitrogen (DON) and particulate nitrogen (PN). Relationship was based on data presented in Abell et al. (2013), collected from the Puarenga Stream when dischage was 3.0 to 15.6 m^3/s (maximum [(TN-DIN)] = 1.62 mg/L; n = 223; r^2 = 0.15). Maximum modelled mean daily [(TN-DIN)] was 1.60 mg/L.
	Ngongotaha and Utuhina	Q < 3 m^3/s: Linear interpolation of monthly measurements collected by BoPRC. Q > 3 m^3/s: Derived from a linear relationship between [(TN-DIN)] and $\log_{10}$Q for the Ngongotaha Stream with correction for log-transformation bias (Ferguson 1986).	Relationship was based on data presented in Abell *et al*. (2013), collected when dischage was 3.0 to 18 m^3/s (maximum [(TN-DIN)] = 1.63 mg/L; *n* = 38; r^2=0.85). Maximum modelled mean daily [(TN-DIN)] was 1.59 mg/L and 1.48 mg/L .
	Awahou, Waiteti, Waingaehe, Waiowhiro, Waiohewa, Hamurana	Linear interpolation of monthly measurements collected by BoPRC.	
DON	All	0.4 × (TN-DIN)	Based on the mean proportions of (TN-DIN) that comprised (TDN-DIN) in 80 samples collected during three storm events on the Puarenga Stream and 73 samples collected during three storm events on the Ngongotaha Stream (Abell *et al*. 2013). The mean proportions were the same for both streams and there was no correlation between the values for this proportion and Q.
PN	All	0.6 × (TN-DIN)	Based on the mean proportions of (TN-DIN) that comprised (TN-TDN) in 80 samples collected during three storm events on the Puarenga Stream and 73 samples collected during three storm events on the Ngongotaha Stream (Abell *et al*. 2013). The mean proportions were the same for both streams and there was no correlation between the values for this proportion and Q.
TN	All	By calculation.	NO_x-N + NH_4-N +DON + PN
ISS	Puarenga	Derived from a linear relationship between $\log_{10}$[(TSS] and $\log_{10}$Q for the Puarenga Stream with correction for log-transformation bias (Ferguson 1986).	Relationship was based on data presented in Abell et al. (2013), collected from the Puarenga Stream when dischage was 1.5 to 10.8 m^3/s (maximum [TSS] = 463 mg/L; n = 507; r^2 = 0.65). Maximum modelled mean daily [TSS] was 1422 mg/L. Assumed that [ISS] = 0.68 × [TSS], based on the mean value of [ISS]/[TSS] measured during storm sampling of Puarenga Stream (*n* = 234, σ = 0.12).
	Ngongotaha and Utuhina	Derived from a power function (negative exponent) between $\log_{10}$[(TSS] and $\log_{10}$Q for the Ngongotaha Stream with correction for log-transformation bias (Ferguson 1986).	Relationship was based on data presented in Abell et al. (2013), collected from the Puarenga Stream when dischage was 1.4 to 22 m^3/s (maximum [TSS] = 510 mg/L; n = 256; r^2 = 0.85). Maximum modelled mean daily [TSS] was 663 mg/L and 295 mg/L. Assumed that [ISS] = 0.57 × [TSS], based on the mean value of [ISS]/[TSS] measured during storm sampling of Ngongotaha Stream (*n* = 111, σ = 0.23).
	Awahou, Waiteti, Waingaehe, Waiowhiro, Waiohewa, Hamurana	Set equal to the mean TSS concentrations measured by BoPRC in each stream since 2000 (sampling undertaken in 2002 and 2003).	
DOCL	All	Calculated as 7.29 × [DIN]	Assumed that C:N is 7.29 (by mass), based on Sterner et al (2008)
POCL	All	Calculated as 7.29 × [PN]	Assumed that C:N is 7.29 (by mass), based on Sterner et al (2008)

Minor streams

Details of how nutrient and ISS concentrations were assigned to nine minor stream inflows are presented in Table 12. The minor streams were represented in the model by a single inflow for which discharge–weighted (i.e., volumetric) concentrations were specified based on estimated loads for individual streams.

Table 12 Methods to assign nutrient concentrations to minor stream inflows. See glossary for definitions of abbreviations.

Analyte	Stream	Estimation method	Notes
PO_4-P	Minor rural surface streams (Waitawa 1, Waitawa 2, Hauraki, Waimehia Drain, Waiowhiro)	Linear interpolation of monthly measurements collected by BoPRC from Waingaehe Stream (smallest of the major stream inflows, drains a predominantly pastoral catchment).	Missing/anomalous measurements replaced with the mean of concentrations measured in adjoining months.
	Lynmore Stream (minor urban surface stream)	Linear interpolation of monthly measurements collected by BoPRC from Lynmore Stream.	
	Groundwater seeps at the lake edge	Set to volumetric mean concentration of samples collected by BoPRC from eight lake–edge springs during 1992 and 1993 (0.176 mg/L; n = 134).	
PP	Minor rural surface streams	Linear interpolation of monthly measurements collected by BoPRC from Waingaehe Stream.	
	Lynmore Stream	Linear interpolation of monthly measurements collected by BoPRC from Lynmore Stream.	
	Groundwater seeps at the lake edge	Set to volumetric mean concentration of samples collected by BoPRC from eight lake–edge springs during 1992 and 1993 (0.074 mg/L; n = 134).	
TP	All	By calculation.	PO_4-P + PP
NO_x-N	Minor rural surface streams	Linear interpolation of monthly measurements collected by BoPRC from Waingaehe Stream.	Missing and anomalous (e.g., > TN) measurements replaced with the mean of concentrations measured in adjoining months.
	Lynmore Stream	Linear interpolation of monthly measurements collected by BoPRC from Lynmore Stream.	
	Groundwater seeps at the lake edge	Set to volumetric mean concentration of samples collected by BoPRC from eight lake–edge springs during 1992 and 1993 (0.036 mg/L; n = 134).	
NH_4-N	All	Linear interpolation of monthly measurements collected by BoPRC.	Missing measurements replaced with the mean of concentrations measured for adjoining months.
(TN-DIN)	Minor rural surface streams	Linear interpolation of monthly measurements collected by BoPRC from Waingaehe Stream.	
	Lynmore Stream	Linear interpolation of monthly measurements collected by BoPRC from Lynmore Stream.	
	Groundwater seeps at the lake edge	Set to volumetric mean concentration of samples collected by BoPRC from eight lake–edge springs during 1992 and 1993 (0.316 mg/L; n = 134).	
DON	All	0.4 × (TN-DIN)	As for major streams.
PN	All	0.6 × (TN-DIN)	
TN	All	By calculation.	NO_x-N + NH_4-N +DON + PN
ISS	Minor rural surface streams	Set equal to the mean TSS concentrations measured by BoPRC in Waingaehe Stream 2000 (sampling undertaken in 2002 and 2003).	
	Lynmore Stream	Set equal to the mean TSS concentrations measured by BoPRC in Lynmore Stream 2000 (sampling undertaken in 2002 and 2003).	
	Groundwater seeps at the lake edge	Assumed nil.	
DOCL	All	Calculated as 7.29 × [DIN]	As for major streams.
POCL	All	Calculated as 7.29 × [PN]	

Atmospheric deposition

Wet atmospheric deposition of nitrogen and phosphorus on the lake surface was represented by configuring precipitation as a surface inflow to the lake (rather than including this in the meteorological forcing file). Precipitation was assigned constant nitrogen concentrations of 0.285 mg/L (as NO_3–N) and phosphorus concentrations of 0.013 mg/L (as PO_4–P), based on values used in previous model applications (Hamilton *et al*. 2012), which were based on typical concentrations for the Taupo Volcanic Zone (Hamilton 2005). Concentrations of other nutrient fractions were not assigned to this input.

Ungauged (residual)

A final inflow was configured that was termed ‘ungauged’. This represented input associated with the residual term in the water balance (see Section 0) and therefore included groundwater inputs to lake bed of the lake, in addition to fluxes associated with overland flow additional minor streams and any under–estimation of hydraulic inputs in the other inflows. Daily nutrient and ISS concentrations in this inflow were assigned using discharge–weighted concentrations calculated using data for the nine major stream inflows.

Summary of assigned nutrient concentrations

A summary of nutrient concentrations assigned to each inflow is presented in Table 13.

Table 13 Summary of nutrient concentrations (g m^{-3}) assigned to inflows represented in the 1–D model, 2007–2014.

Analyte	Percentile	Inflow													
		Awahou	Hamarana	Puarenga	Puarenga (-LTS)	Puarenga (-alum)	Utunina	Utuhina (-alum)	Waingaehe	Waiohewa	Waiowhiro	Waiteti	Minor	Unguaged	Atmospheric deposition
PO_4-P	5	0.056	0.062	0.003	0.002	0.020	0.007	0.043	0.070	0.007	0.021	0.024	0.089	0.036	0.013
	25	0.063	0.075	0.006	0.005	0.028	0.016	0.048	0.089	0.013	0.028	0.030	0.101	0.044	0.013
	50	0.066	0.079	0.012	0.010	0.035	0.027	0.055	0.094	0.017	0.035	0.033	0.106	0.048	0.013
	75	0.070	0.082	0.042	0.033	0.045	0.035	0.060	0.098	0.021	0.039	0.037	0.109	0.051	0.013
	95	0.078	0.087	0.065	0.051	0.065	0.046	0.065	0.105	0.029	0.044	0.045	0.115	0.058	0.013
TP	5	0.07	0.08	0.05	0.04	0.07	0.05	0.06	0.10	0.07	0.06	0.04	0.13	0.07	0.013
	25	0.07	0.08	0.07	0.06	0.09	0.06	0.08	0.11	0.08	0.08	0.05	0.14	0.08	0.013
	50	0.07	0.09	0.09	0.07	0.10	0.07	0.10	0.12	0.10	0.09	0.06	0.14	0.09	0.013
	75	0.08	0.09	0.11	0.09	0.12	0.08	0.11	0.13	0.11	0.10	0.07	0.15	0.09	0.013
	95	0.10	0.11	0.14	0.11	0.14	0.10	0.14	0.15	0.15	0.11	0.09	0.17	0.11	0.013
NO_3-N	5	1.087	0.646	0.677	0.295	0.677	0.509	0.509	1.270	0.983	0.759	1.137	1.066	0.820	0.285
	25	1.240	0.695	0.774	0.295	0.774	0.591	0.591	1.379	1.162	0.854	1.300	1.167	0.888	0.285
	50	1.320	0.726	0.844	0.295	0.844	0.652	0.652	1.454	1.330	0.911	1.368	1.244	0.949	0.285
	75	1.448	0.775	0.948	0.295	0.948	0.708	0.708	1.528	1.457	0.968	1.441	1.318	0.984	0.285
	95	1.519	0.807	1.114	0.295	1.114	0.822	0.822	1.631	1.680	1.082	1.581	1.402	1.062	0.285
NH_4-N	5	0.001	0.003	0.034	0.063	0.034	0.024	0.024	0.003	0.373	0.007	0.008	0.099	0.033	0.00
	25	0.004	0.005	0.060	0.063	0.060	0.030	0.030	0.005	0.897	0.013	0.012	0.107	0.048	0.00
	50	0.005	0.006	0.069	0.063	0.069	0.035	0.035	0.007	1.221	0.019	0.015	0.113	0.061	0.00
	75	0.009	0.008	0.079	0.063	0.079	0.042	0.042	0.011	1.505	0.028	0.018	0.119	0.075	0.00
	95	0.020	0.014	0.104	0.063	0.104	0.056	0.056	0.015	1.938	0.046	0.028	0.126	0.088	0.00
TN	5	1.16	0.74	0.80	0.38	0.80	0.63	0.63	1.37	1.99	0.84	1.27	1.36	0.99	0.285
	25	1.32	0.79	0.95	0.42	0.95	0.71	0.71	1.46	2.35	0.95	1.41	1.44	1.05	0.285
	50	1.41	0.83	1.05	0.47	1.05	0.77	0.77	1.56	2.61	1.00	1.47	1.52	1.08	0.285
	75	1.50	0.86	1.20	0.54	1.20	0.86	0.86	1.64	2.89	1.08	1.55	1.58	1.16	0.285
	95	1.64	0.88	1.46	0.75	1.46	1.07	1.07	1.78	3.34	1.17	1.70	1.66	1.29	0.285

Alum dosing of Puarenga and Utuhina streams

Alum was added to the Utuhina and Puarenga streams during the baseline period, resulting in reduced dissolved reactive phosphorus concentrations in the streams and the lake (see Section 0). It was therefore necessary to represent this action in the model configuration for the baseline period.

The water quality monitoring site at the Utuhina Stream is downstream of the alum dosing plant and therefore the measured water quality data for this stream reflected the in–stream effects of alum (i.e., reduced dissolved reactive phosphorus concentrations). The water quality monitoring site at the Puarenga Stream was upstream of the alum dosing plant and therefore it was necessary to reduce dissolved reactive phosphorus concentrations in the inflow data for this stream inflow to reflect alum effects. Concentrations were reduced in proportion to the mean load of aluminum that was applied during a particular month (data provided by BoPRC). This was calculated using a linear–$\log_{10}$ relationship derived by Hamilton *et al*. (2015) between the concentration reduction factor and aluminium load, based on data collected by BoPRC at sites upstream and downstream of the alum dosing plant (Figure 4).

In addition, two changes were made to the configuration of the water quality parameters in CAEDYM to reflect the in–lake effects of alum. Firstly, internal loading associated with hypoxia was suppressed by reducing the maximum potential PO_4–P release rate from bed sediments to 0.02 g/m^2/d, which is lower than the rate assigned in previous model applications that simulated periods prior to alum dosing. Secondly, elevated in–lake flocculation of organic material caused by alum was represented by assigning a high particulate organic material diameter of 0.018 mm. Further details about the rationale for the methods used to represent in–lake alum effects are provided in Hamilton *et al*. (2015).

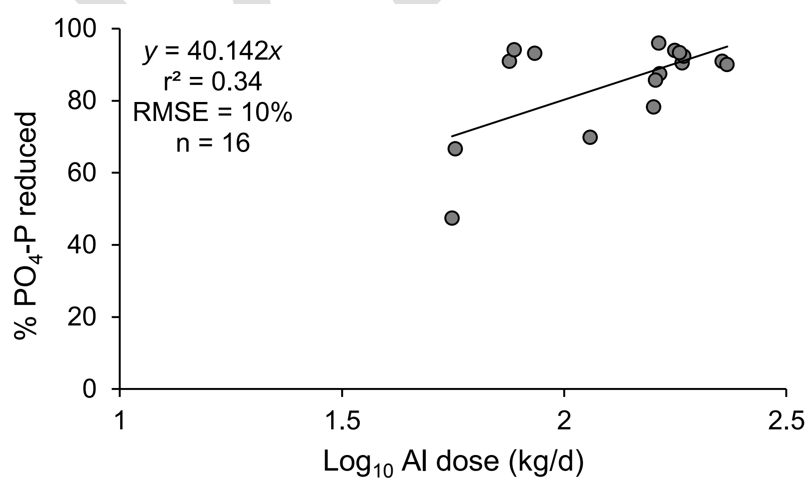

Figure 4 **Relationship between percentage reductions to dissolved reactive phosphorus (PO_4–P) concentrations and mean monthly aluminium dose in the Puarenga Stream. Data provided by BoPRC.**

Model scenarios

Baseline and wastewater discharge

Scenarios simulated with the 1–D model are listed in Table 14. The baseline (1D_0) scenario involved no discharge of treated wastewater and therefore provides a benchmark representative of current conditions against which the effects of the various scenarios can be compared. Separate scenarios were simulated to represent discharge of treated wastewater to surface waters following treatment using each of the six treatment options (Table 2). These scenarios were configured by adding the treated wastewater as a separate inflow that enters the lake surface. These scenarios therefore represent discharge to either the Puarenga Stream or the lake shore sites (Map 2).

Two further scenarios were configured to examine the effects of lake bed discharge. The treatment options selected for these scenarios were 2c and 3a because they provide some contrast; relative to the other options, these respectively have low phosphorus concentrations and moderate nitrogen concentrations, or low nitrogen concentrations and moderate phosphorus concentrations.

Discharge rates and nutrient concentrations were assigned to the treated wastewater using the information presented in Mott MacDonald (2014; see Table 2). Table 15 presents the mean annual nitrogen and phosphorus loads in the Puarenga Stream that correspond to the 1–D scenarios.

The treated wastewater temperature was assumed to follow an annual sinusoidal trend with a maximum of 18 °C and a minimum of 16 °C (K. Brian, pers. comm. 2015b; Figure 5). Precise specifications of dissolved oxygen concentrations were unavailable so treated wastewater was generally assumed to be 100% saturated in the scenarios (see Section 0), although two additional scenarios were included to simulate discharge of anoxic treated wastewater (Options 2c and 3a) to isolate the effects of varying this parameter.

No distinctions were made between the various discharge arrangements, such as gabions or rapid infiltration beds (Section 0). The purpose of these options is to convey treated wastewater, rather than to provide treatment (Mott MacDonald 2014; RPSC 2014). Consequently, no specific discharge arrangement has been specified for the scenarios.

The lake outflow volume was increased (+ 0.276 m^3/s) to balance the additional inflow for all scenarios involving treated wastewater discharge.

Table 14 Scenarios simulated using the 1–D model.

#	Code	Scenario	Details
1	1D_0	Baseline with no wastewater discharge simulated.	Eight year period (2007-2014). Alum dosing effects represented.
2	1D_1_Surface	Treatment option 1, discharge to surface waters	
3	1D_2a_Surface	Treatment option 2a, discharge to surface waters	
4	1D_2b_Surface	Treatment option 2b, discharge to surface waters	
5	1D_2c_Surface	Treatment option 2c, discharge to surface waters	
6	1D_3a_Surface	Treatment option 3a, discharge to surface waters	
7	1D_3b_Surface	Treatment option 3b, discharge to surface waters	
8	1D_4_Surface	Treatment option 4, discharge to surface waters	
9	1D_5_Surface	Treatment option 5, discharge to surface waters	
10	1D_6a_Surface	Treatment option 6a, discharge to surface waters	
11	1D_6b_Surface	Treatment option 6b, discharge to surface waters	
12	1D_2c_Surface - DO	Treatment option 2c, discharge to surface, no dissolved oxygen in wastewater	Option 2c has the 'best' P treatment (TP = 0.10 mg/L) and 'moderate' N treatment (TN = 4.37 mg/L)
13	1D_3a_Surface - DO	Treatment option 3a, discharge to surface, no dissolved oxygen in wastewater	Option 3a has the 'best' N treatment (TN = 2.63 mg/L) and 'moderate' P treatment (TP = 0.20 mg/L)
14	1D_2c_Bed	Treatment option 2c, discharge to lake bed	
15	1D_3a_Bed	Treatment option 3a, discharge to lake bed	
16	1D_0 - LTS	Baseline, Land Treatment System loads removed from the Puarenga Stream	
17	1D_2c_Surface - LTS	Treatment option 2c, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	
18	1D_3a_Surface - LTS	Treatment option 3a, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	
19	1D_4_Surface - LTS	Treatment option 4, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	
20	1D_5_Surface - LTS	Treatment option 5, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	
21	1D_6a_Surface - LTS	Treatment option 6a, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	
22	1D_6b_Surface - LTS	Treatment option 6b, discharge to surface, Land Treatment System loads removed from the Puarenga Stream	
23	1D_0 - Alum	Baseline, alum effects (in-lake and in-stream) not simulated	
24	1D_2c_Surface - Alum	Treatment option 2c, discharge to surface, alum effects (in-lake and in-stream) not simulated	
25	1D_3a_Surface - Alum	Treatment option 3a, discharge to surface, alum effects (in-lake and in-stream) not simulated	
26	1D_0 - LTS - Alum	Baseline, Land Treatment System loads removed from the Puarenga Stream, alum effects (in-lake and in-stream) not simulated	
27	1D_2c_Surface - LTS - Alum	Treatment option 2c, discharge to surface, Land Treatment System loads removed from the Puarenga Stream, alum effects (in-lake and in-stream) not simulated	
28	1D_3a_Surface - LTS - Alum	Treatment option 3a, discharge to surface, Land Treatment System loads removed from the Puarenga Stream, alum effects (in-lake and in-stream) not simulated	
29	1D_0 + 'pure' wastewater	Baseline with discharge of wastewater to surface waters that contains no nutrients	Not proposed but simulated to quanity potential flushing effects

Table 15 **Summary of mean annual Puarenga Stream nutrient loads for the 1–D model scenarios. For modelled scenarios, treated wastewater loads were added to one of the three baseline load options, as required for each configuration (see Table 14).**

Scenario/Option	Description	TN (t/y)		DIN (t N/y)		TP (t/y)		PO_4-P (t P/y)	
		Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.
1D_0_Stream	Baseline Puarenga Stream loads (PO_4-P attenuated by alum)	70.1	16.4	58.1	11.7	6.0	1.6	1.4	1.1
1D_0-LTS	Baseline Puarenga Stream loads with LTS loads removed	34.0	8.4	22.0	3.5	4.8	1.3	1.1	0.9
1D_0 - Alum	Baseline Puarenga Stream loads with no alum dosing	70.1	16.4	58.1	11.7	6.9	1.9	2.3	0.5
Option 1	Loads in treated wastewater	47.3	0.1	28.5	0.0	6.3	0.0	0.9	0.0
Option 2a		42.3	0.1	28.5	0.0	3.2	0.0	0.9	0.0
Option 2b		40.2	0.0	28.5	0.0	1.7	0.0	0.9	0.0
Option 2c		38.0	0.0	28.5	0.0	0.9	0.0	0.9	0.0
Option 3a		22.9	0.0	11.2	0.0	1.7	0.0	0.9	0.0
Option 3b		31.6	0.0	19.9	0.0	1.7	0.0	0.9	0.0
Option 4		30.0	0.0	28.5	0.0	3.0	0.0	0.9	0.0
Option 5		30.0	0.0	28.5	0.0	1.5	0.0	0.9	0.0
Option 6a		30.7	0.0	22.6	0.0	3.0	0.0	3.0	0.0
Option 6b		30.7	0.0	22.6	0.0	1.5	0.0	1.5	0.0

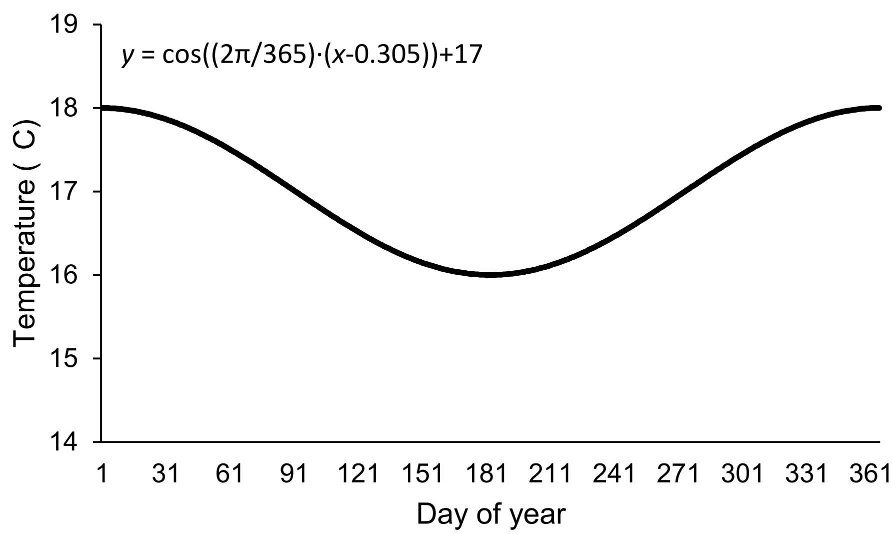

Figure 5 **Water temperatures assigned to treated wastewater.**

Removal of LTS loads and alum dosing

A further 13 scenarios (#16–28 in Table 14) were simulated to examine permutations of the following two conditions: 1) removal of nutrient loads from the Puarenga Stream associated with the LTS; 2) cessation of alum dosing.

Removal of LTS loads was simulated to reflect the decline in background nutrient loads in the Puarenga Stream that is anticipated to occur over the medium to long term following LTS closure. The rate of this decline is uncertain and background nutrient loads are expected to be higher than those in the '-LTS' scenario for several years after the LTS is closed while residual loads are 'flushed' through the catchment (see Discussion for further consideration of lag times). Scenarios comprising no LTS loads were configured by reducing concentrations of nitrogen and phosphorus fractions in the Puarenga Stream. Discharge was not reduced and it was assumed that there would be negligible decline in water yield following LTS closure as the majority of irrigated water is presumed to be lost from the catchment by evapotranspiration. Dissolved inorganic nitrogen concentrations were set to the mean of concentrations measured in the Puarenga Stream in 1992 and 1993, immediately following the initiation of the LTS in 1991. No measurements before 1991 were available and the 1992–1993 data were assumed representative of conditions prior to the marked increase in nitrogen concentrations that occurred through the mid to late 1990s (Tomer *et al*. 2000; Burns *et al*. 2009). Thus, nitrogen concentrations in the Puarenga Stream under the '-LTS' scenarios were approximately 2.5– to 3–fold less than contemporary concentrations[5]. Unlike nitrogen, phosphorus concentrations measured by BoPRC did not exhibit a marked rise in the years following LTS initiation, and contemporary concentrations are comparable with those in the early 1990s, with data indication only a slight increase in total phosphorus and no increase in dissolved reactive phosphorus[6]. Phosphorus concentrations are typically more variable than nitrogen concentrations as the particulate fraction is strongly correlated with discharge. Phosphorus concentrations in the '-LTS' scenarios were therefore configured by adjusting concentrations of all phosphorus fractions by a constant factor (0.81) to reduce the phosphorus load in the Puarenga Stream during the baseline period by an average of 1.7 t P/y, which is the 5–year 'sewage–derived' load estimated from LTS consent monitoring during 2007–2012 (A. Lowe, pers. comm. 2013)[7].

Scenarios were simulated to examine the effects of removing alum dosing to examine how discontinuing this action will influence the predicted effects of discharging treated wastewater. Configuring these scenarios involved: 1) increasing dissolved reactive phosphorus concentrations in the Utuhina and Puarenga streams to 'non alum' levels; 2) adjusting the CAEDYM parameters that were specifically modified to represent the in–lake effects of alum dosing.

Dissolved reactive phosphorus concentrations in the Utuhina Stream (monitored downstream of the alum dosing plant) were amended by setting them equal to the product of the mean ratio of dissolved

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[5] Assigned concentrations were: NH_4–N = 0.064 mg/L; NO_3–N = 0.295 mg/L.

[6] E.g., mean 1992–1993 data: TP = 0.060 mg/L, PO4–P = 0.042 mg/L; mean 2013–2014 data: TP = 0.090 mg/L, PO4–P = 0.036 mg/L.

[7] Thus the baseline phosphorus load was reduced by 13.6 t (8 × 1.7) over the eight years. Note that this calculation method meant that the load in each year was not reduced by exactly 1.7 t, and therefore the difference in mean annual phosphorus load between the scenarios with and without LTS loads is ~1.4 t P/y (Table 15).

reactive phosphorus to total phosphorus during 2001-2005 (pre–alum dosing; 0.804) and assigned total phosphorus, with the maximum value set to 0.065 mg/L (90th percentile of 2001-2005 monitoring data) to avoid anomalously high values during storms, when total phosphorus was estimated using a relationship with discharge (Table 11). Dissolved reactive phosphorus concentrations in the Puarenga Stream (monitored upstream of the alum dosing plant) were set to the concentrations determined before modifications to represent alum effects (see Section 0).

Removal of in–lake alum effects was represented in CAEDYM by adjusting the maximum PO_4–P release rate and particulate organic material diameter to 0.04 g/m^2/d and 0.09 mm respectively. These values correspond to calibrated values that were used in a version of the model configured for the period prior to alum dosing commencing (Hamilton *et al*. 2015).

Additional scenarios

Two additional configurations of treated wastewater discharge were simulated to examine the effects of improvements to current treatment performance. These two scenarios involved discharge of either: 1) 30 t N/y and 3 t P/y, or; 2) 30 t N/y and 1.5 t P/y. The first of these configurations was simulated both with and without LTS loads. The second of these scenarios was simulated without inclusion of LTS loads. These scenarios were configured by setting the dissolved inorganic nitrogen and phosphorus concentrations in treated wastewater equal to those of Options 1 and 2 (Table 2), and then varying the concentrations of the other fractions to achieve the desired loads.

A final scenario (1D_0 + 'pure' wastewater; Table 14) was configured that involved addition of wastewater that contained no nutrients. The objective of this was to isolate any potential effects that that are predicted to occur following wastewater discharge solely due to a slight reduction in residence time in receiving waters, rather than enhanced productivity due to nutrient addition.

Comparison of scenarios

Annual TLI_3 values were compared between the model scenarios to provide an assessment of the predicted effects of each scenario on lake trophic status in the context of water quality objectives for Lake Rotorua (BoPRC 2009). The TLI_3 integrates concentrations of total nitrogen, total phosphorus and chlorophyll *a*, based on the equations presented in Burns *et al*. (1999). TLI_3 values were calculated using surface water data.

The TLI_3 value is comparable with the TLI (see Section 0) although Secchi depth is omitted from the calculation, which is not calculated explicitly in CAEDYM. This omission means that TLI_3 and TLI are not identical, and the TLI target for Lake Rotorua of 4.20 (BoPRC 2009) is equivalent to 4.32 in TLI_3 units (Hamilton *et al*. 2015).

In addition, modelled concentrations of chlorophyll *a*, total nitrogen and total phosphorus for each scenario were compared with Ecosystem Health attribute values prescribed for lakes in the current National Policy Statement for Freshwater Management (New Zealand Government 2014). These values are reproduced in Table 16, Table 17 and Table 18.

Table 16 Chlorophyll *a* concentrations (µg/L) corresponding to Lake Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management (New Zealand Government 2014).

Attribute state	Numeric attribute state		Narrative attribute state
	Annual median	Annual maximum	
A	≤ 2	≤ 10	Lake ecological communities are healthy and resilient, similar to natural reference conditions.
B	> 2 and ≤ 5	>10 and ≤ 25	Lake ecological communities are slightly impacted by additional algal and plant growth arising from nutrients levels that are elevated above natural reference conditions.
C	> 5 and ≤ 12	> 25 and ≤ 60	Lake ecological communities are moderately impacted by additional algal and plant growth arising from nutrients levels that are elevated well above natural reference conditions.
National bottom line	12	60	
D	> 12	> 60	Lake ecological communities have undergone or are at high risk of a regime shift to a persistent, degraded state, due to impacts of elevated nutrients leading to excessive algal and/or plant growth, as well as from losing oxygen in bottom waters of deep lakes.

Table 17 Total nitrogen concentrations (µg/L) corresponding to Lake Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management (New Zealand Government 2014).

Attribute state	Numeric attribute state	Narrative attribute state
	Annual median (polymictic)	
A	≤ 300	Lake ecological communities are healthy and resilient, similar to natural reference conditions.
B	> 300 and ≤ 500	Lake ecological communities are slightly impacted by additional algal and plant growth arising from nutrients levels that are elevated above natural reference conditions.
C	> 500 and ≤ 800	Lake ecological communities are moderately impacted by additional algal and plant growth arising from nutrients levels that are elevated well above natural reference conditions.
National bottom line	800	
D	> 800	Lake ecological communities have undergone or are at high risk of a regime shift to a persistent, degraded state, due to impacts of elevated nutrients leading to excessive algal and/or plant growth, as well as from losing oxygen in bottom waters of deep lakes.

Table 18 Total phosphorus concentrations (µg/L) corresponding to Lake Ecosystem Health Attribute States designated in the National Policy Statement for Freshwater Management (New Zealand Government 2014).

Attribute state	Numeric attribute state	Narrative attribute state
	Annual median	
A	≤ 10	Lake ecological communities are healthy and resilient, similar to natural reference conditions.
B	> 10 and ≤ 20	Lake ecological communities are slightly impacted by additional algal and plant growth arising from nutrients levels that are elevated above natural reference conditions.
C	> 20 and ≤ 50	Lake ecological communities are moderately impacted by additional algal and plant growth arising from nutrients levels that are elevated well above natural reference conditions.
National bottom line	50	
D	> 50	Lake ecological communities have undergone or are at high risk of a regime shift to a persistent, degraded state, due to impacts of elevated nutrients leading to excessive algal and/or plant growth, as well as from losing oxygen in bottom waters of deep lakes.

Three–dimensional lake modelling

Model selection
ELCOM[8] (v. 2.2) was selected for the 3–D modelling. ELCOM is a 3–D hydrodynamics, thermodynamics and transport model that was developed at the Centre for Water Research, University of Western Australia. The model has been used extensively worldwide, and it has recently been used to study mixing processes in Lake Rotorua over periods of weeks to a month (Abell and Hamilton 2015; Gibbs *et al., in prep*.). Elsewhere in New Zealand, ELCOM has been used, either on its own or in combination with CAEDYM, to study systems that include Tauranga Harbour (Tay *et al*. 2013), Lake Benmore (Norton *et al*. 2009, Trolle et al. 2014), Lake Rotoiti (Von Westernhagen 2010) and Lake Rotoehu (Allan 2014).

Model overview
ELCOM simulates velocity, salinity and temperature distributions in water bodies. The model solves the unsteady Reynolds–averaged Navier–Stokes and scalar transport equations, with modules for heat and momentum transfer across the water surface due to wind and atmospheric thermodynamics (Hodges and Dallimore 2011).

ELCOM was used in this study to investigate how mixing processes in the lake may affect the transport of treated wastewater that is discharged at the proposed locations (Map 2). This required configuring the model to include an inflow that represented treated wastewater discharge. The propagation of the inflow was then examined by observing the path of a conservative tracer included in the inflow.

Model simulation periods and validation
Two separate periods were simulated to examine mixing under contrasting conditions; these were: summer 2013/2014 and winter 2014. The model was typically run for a two–month period, although some simulations designed to examine model sensitivity to wind forcing (see below) were run for only one month. Each simulation was preceded by a two–week 'spin up' period that was not considered in analysis. The performance of the model with regard to simulating the temperature structure of the lake was validated by comparing simulated temperatures with high frequency temperature measurements collected at the monitoring buoy operated by the University of Waikato. Modelled temperatures were compared with measurements at three depths: 0.5 m (epilimnion when stratified), 12.5 m (approximate depth of metalimnion when stratified) and 20.5 m (hypolimnion when stratified and deepest point monitored).

Further validation of mixing processes was not undertaken; the implications of this for model uncertainty are considered in the Discussion.

Model configuration
Model application required simplifying lake morphology by discretizing the water column into 3–D cells with dimensions: x = 50 m, y = 50 m and z = 0.5 – 2 m. Mean elevation of each cell was determined by interpolation using a bathymetry map with 5–m horizontal resolution that was provided by BoPRC.

[8] Estuary, Lake and Coastal Ocean Model

'Flow' boundary conditions were specified at the lake–bottom and sidewalls. ELCOM was run without CAEDM and thus heat flux and storage associated with particulate material (e.g., phytoplankton cells) were constant. Hourly discharge, temperature and dissolved oxygen concentrations were assigned to 18 separate inflows using the methods described for 1–D model configuration (Section 0).

Meteorological forcing data for the following variables were obtained from the Rotorua Airport AWS (Map 1): wind speed, wind direction, air temperature, solar radiation, atmospheric pressure and rainfall. Cloud cover was estimated from short–wave solar radiation (see Section 0). Meteorological data for the two modelling periods are presented in Figure 6 and Figure 7.

The summer period was typically characterized by having moderate wind speeds (5 to 8 m/s) in the afternoon, frequently from a north–west to north–east direction, indicative of sea breezes from the Bay of Plenty (Figure 6). There was, however, a period of approximately two weeks in early January when the wind was predominantly from a south–west to westerly direction. This was approximately three weeks into the simulation period.

Wind speeds were generally higher during the winter period (Figure 7). Approximately one week after the start of the simulation period, there was a period of several days (~10–13 July) with high rainfall (~ 50 mm) and north–east winds of moderate to high speed (~5 to 13 m/s). Later, there were multiple periods of several days with consistent south–westerly winds, which are typical of winter in Rotorua.

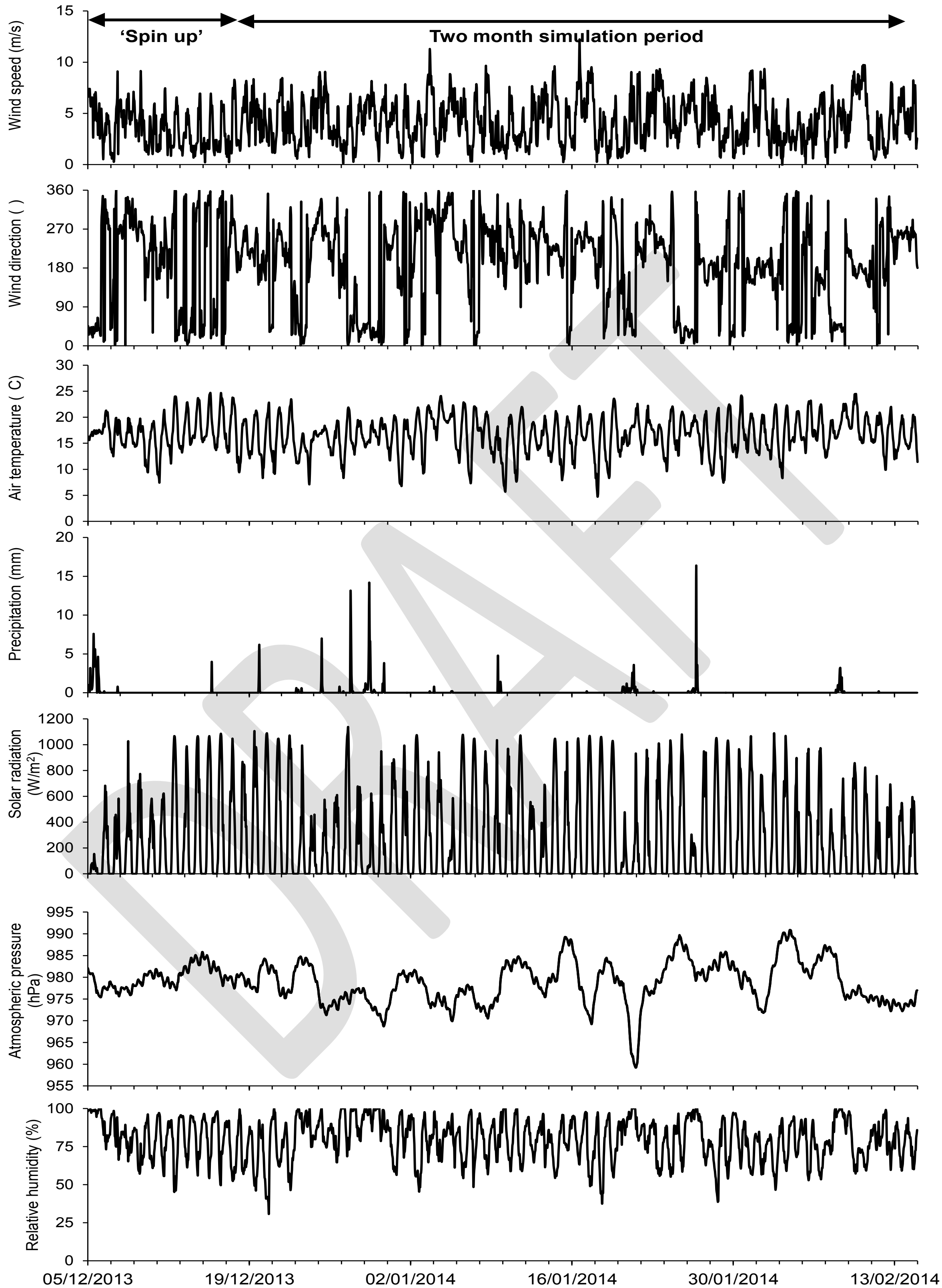

Figure 6 **Hourly mean meteorological data for the summer 2013/14 modelling period.**

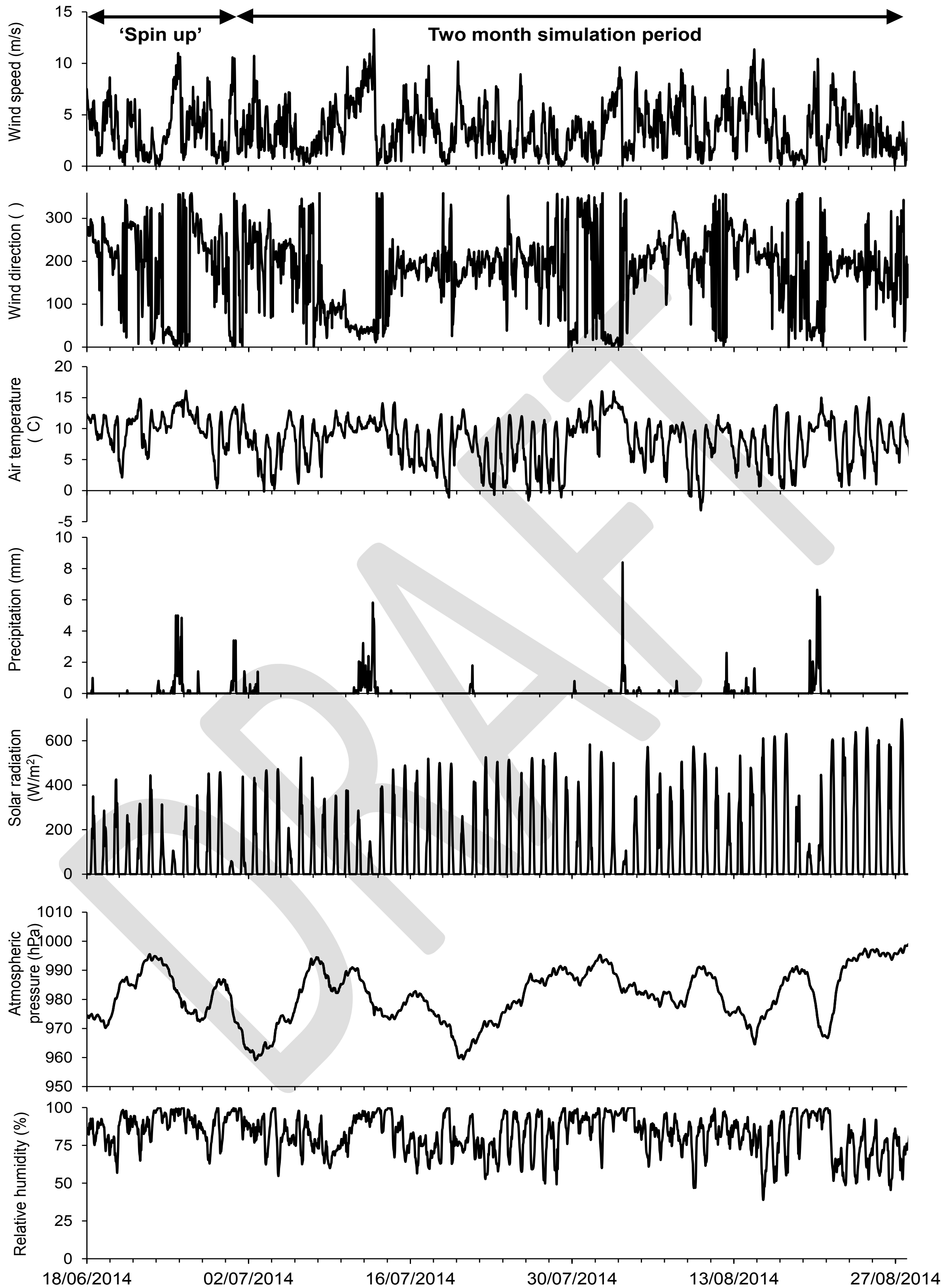

Figure 7 **Hourly mean meteorological data for the winter 2014 modelling period.**

Model scenarios

The 3–D modelling scenarios are listed in Table 19. Scenarios involved simulating discharge of treated wastewater to the lake at a constant rate (0.2756 m^3/s; Mott MacDonald 2014). Discharge was simulated to either the Puarenga Stream (representing discharge locations 1 to 3; Map 2), Lake Rotorua shoreline (at discharge location 5; Map 2) or the lake bed 2 km to the north of the Puarenga Stream mouth, at a depth of ~22 m (representing discharge location 6; Map 2). These scenarios were simulated for both the summer and winter periods. In addition, the summer scenarios were simulated using each of the following configurations of wind forcing data: 1) constant moderate winds (4 m/s) from the north–east; 2) constant moderate winds (4 m/s) from the south–west. These two artificial wind configurations were included because they represent the dominant wind directions in Rotorua (Figure 8), and previous work has indicated that these wind conditions establish alternate circulation patterns that have the potential to exert major and differing effects on how treated wastewater moves throughout the lake (Gibbs *et al*. 2011; Abell and Hamilton 2015; Gibbs *et al*., *in prep*.).

Table 19 Scenarios simulated with the 3–D model.

#	Code	Scenario
1	3D_S_Stream_NE	Summer, wastewater discharge to the Puarenga Stream, NE wind forcing
2	3D_S_Stream_SW	Summer, wastewater discharge to the Puarenga Stream, SW wind forcing
3	3D_S_Shore_NE	Summer, wastewater discharge to the lake shoreline (Site 5), NE wind forcii
4	3D_S_Shore_SW	Summer, wastewater discharge to the lake shoreline (Site 5), SW wind forcii
5	3D_S_Bed_NE	Summer, wastewater discharge to the lake bed (Site 6), NE wind forcing
6	3D_S_Bed_SW	Summer, wastewater discharge to the lake bed (Site 6), SW wind forcing
7	3D_S_Stream	Summer, wastewater discharge to the Puarenga Stream
8	3D_S_Shore	Summer, wastewater discharge to the lake shoreline (Site 5)
9	3D_S_Bed	Summer, wastewater discharge to the lake bed (Site 6)
10	3D_W_Stream	Winter, wastewater discharge to the Puarenga Stream
11	3D_W_Shore	Winter, wastewater discharge to the lake shoreline (Site 5)
12	3D_W_Bed	Winter, wastewater discharge to the lake bed (Site 6)

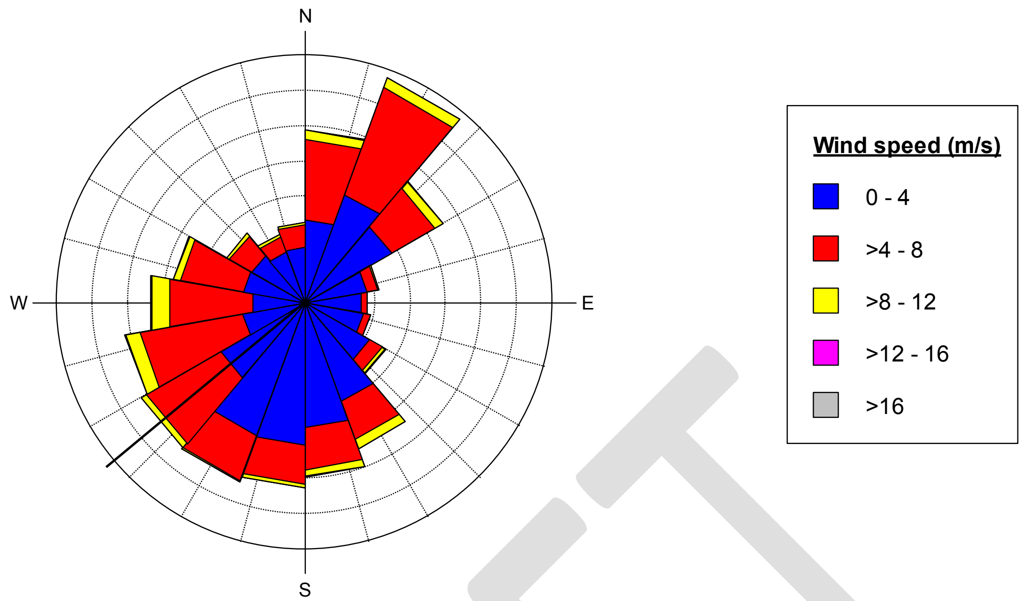

Figure 8 **Summary of hourly wind measurements at Rotorua Airport Automatic Weather Station, 2007–2014.**

1.1.1. Comparison of scenarios

Simulated tracer concentrations at various depths and locations in the lake were visualised for individual scenarios using ARMSLite v. 2.1.2, developed at the Centre for Water Research, University of Western Australia (Dallimore 2011).

Results

Mass balance calculations to estimate in–stream nutrient loads and concentrations

Dilution calculations

Figure 9 presents a cumulative frequency curve that shows the range in the proportions of the Puarenga Stream that would comprise treated wastewater under a scenario of constant wastewater discharge to the stream, based on 2005–2015 stream discharge data. Thus, results indicate that treated wastewater is expected to comprise 0.9% to 24.4% of the stream flow, with treated wastewater comprising < 13.6% of the stream flow for 50% of the time, and < 17.8% of the stream flow for 90% of the time.

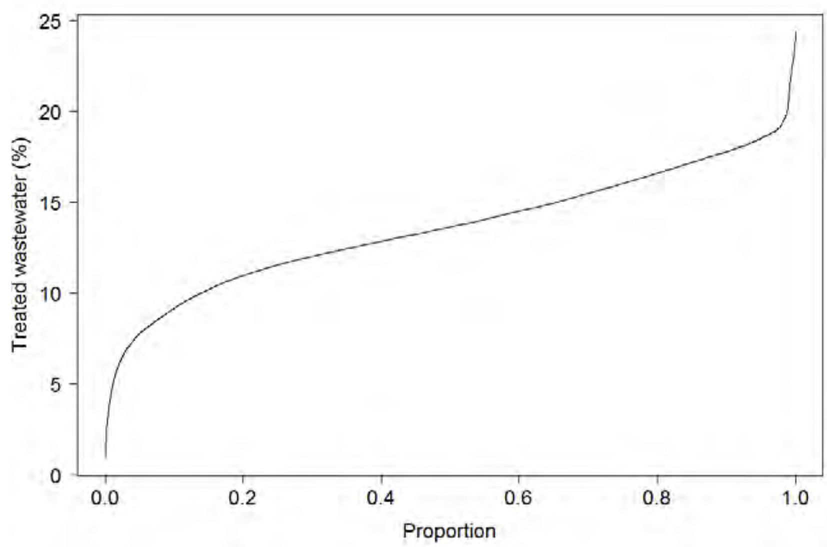

Figure 9 **Cumulative frequency curve showing the range in the percentage of Puarenga Stream water that would comprises treated wastewater under a scenario of constant wastewater discharge to the stream. Based on stream discharge data from 2005–2015.**

Data presented in Figure 9 are summarised by month in Table 20. There is only weak seasonality in the data, reflecting the lack of a strong seasonal pattern in Puarenga Stream discharge (Figure 2). The median predicted proportion of the Puarenga Stream flow that comprises treated wastewater is 12% to 13% during May through October, and 15% to 17% during November through April (Table 20).

Table 20 **Percentiles of the percentage of Puarenga Stream flow that is predicted to comprise treated wastewater by month. Based on stream discharge data from 2005–2015.**

Month	Percentile				
	10th	25th	50th	75th	90th
January	8%	13%	16%	18%	19%
February	10%	12%	17%	18%	19%
March	11%	13%	16%	18%	18%
April	10%	13%	15%	17%	18%
May	8%	11%	13%	16%	17%
June	8%	11%	13%	15%	17%
July	9%	11%	13%	14%	15%
August	8%	9%	12%	13%	14%
September	9%	11%	12%	14%	15%
October	10%	11%	13%	14%	16%
November	12%	13%	15%	16%	19%
December	11%	13%	15%	17%	18%

Treated wastewater nutrient loads to the Puarenga Stream

Nutrient loads for the different treatment options are sumarised in Table 15 in Section 0 above. Loads for each option are presented graphically in Figure 10 and Figure 11 below, which show annual (TLI years) nutrient loads for each of the 1–D modelling scenarios for the duration of the modelling period.

There was considerable between–year variability in estimated nutrient loads conveyed by the Puarenga Stream during the baseline period. These differences primarily reflect differences in rainfall that in turn affect discharge (Figure 2). Variability in discharge influences nutrient loads directly by affecting hydraulic loads, and indirectly by affecting particulate nutrient concentrations during storm flow periods (Table 11). In particular, 2011 was a notably wet year (Figure 2) with mean annual discharge 29% greater than the mean for the period.

Annual total nitrogen loads in the Puarenga Stream under the baseline scenario range from 48.4 to 105.6 t N/y (mean = 70.1 t N/y). Additional total nitrogen loads for the different treatment options vary from 23 t N/y (Option 3a) to 47 t N/y (Option 1), corresponding to 43% to 67% of the mean baseline load (Table 15; Figure 10).

Annual total phosphorus loads in the Puarenga Stream under the baseline scenario range from 3.1 to 8.9 t P/y (mean = 6.0 t P/y). Note that phosphorus loads under the baseline scenario reflect reduced phosphorus concentrations to represent the effects of alum dosing (see Section 0); annual total phosphorus loads prior to these corrections (i.e., for the '1D_0-Alum' scenario) range from 4.5 to 10.6 t P/y (mean = 6.6 t P/y). Additional total phosphorus loads for the different treatment options vary from

0.9 t P/y (Option 2c) to 6.3 t P/y (Option 1), corresponding to 14% to 104% of the mean annual baseline load for the stream (Table 15; Figure 10).

Relative to the 2029 external nutrient load reduction targets set for Lake Rotorua catchment (BoPRC 2009), the loads for the treatment options alone correspond to 9% to 19% of the nitrogen load target (250 t N/y) and 9% to 63% of the phosphorus load target (10 t P/y).

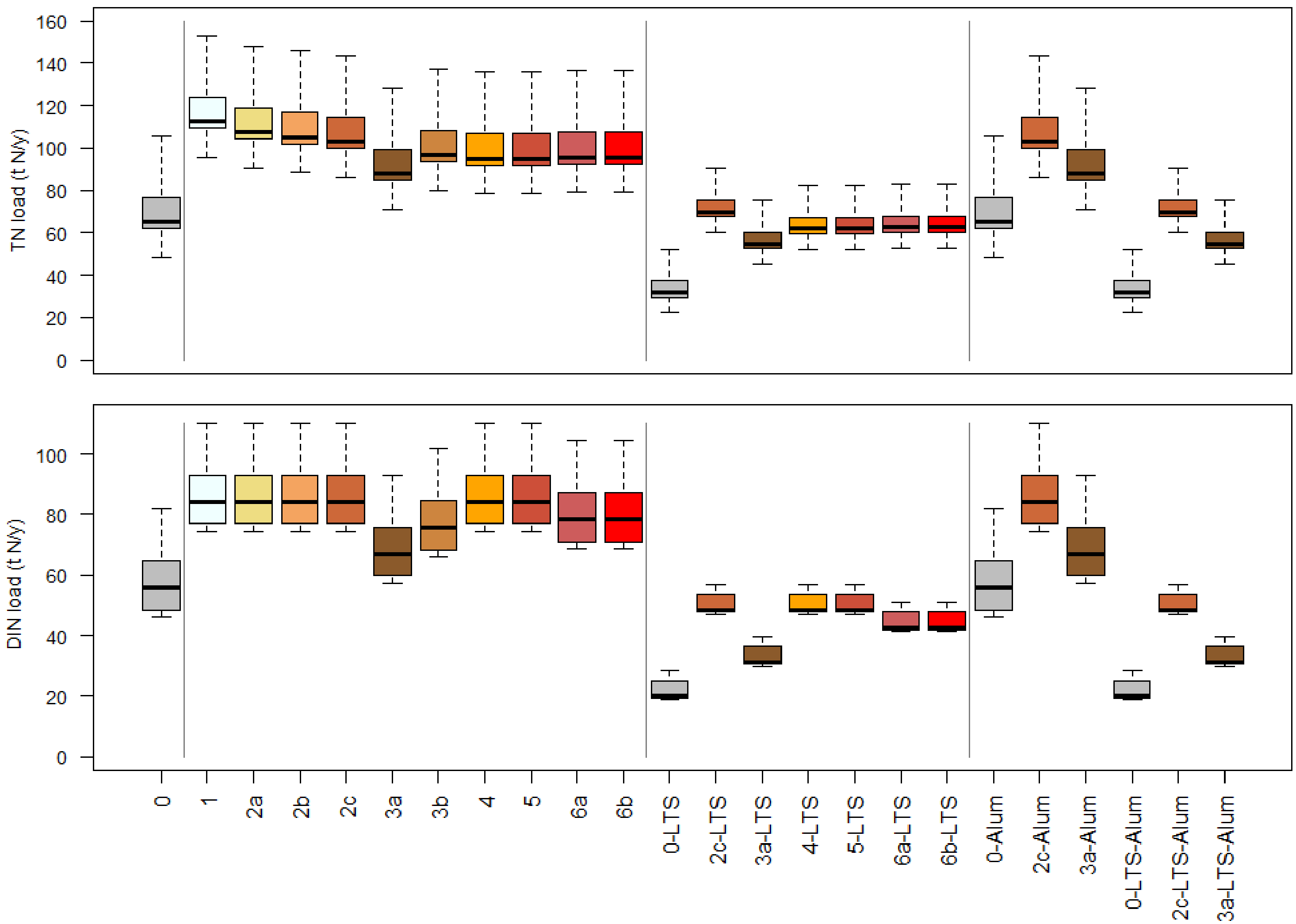

Figure 10 **Annual (2007–2014) total nitrogen (TN) and dissolved inorganic nitrogen (DIN) loads in the Puarenga Stream that correspond to 1–D modelling scenarios (see Table 14 for scenario descriptions). Horizontal lines denote median values; boxes denote interquartile range; whiskers denote range.**

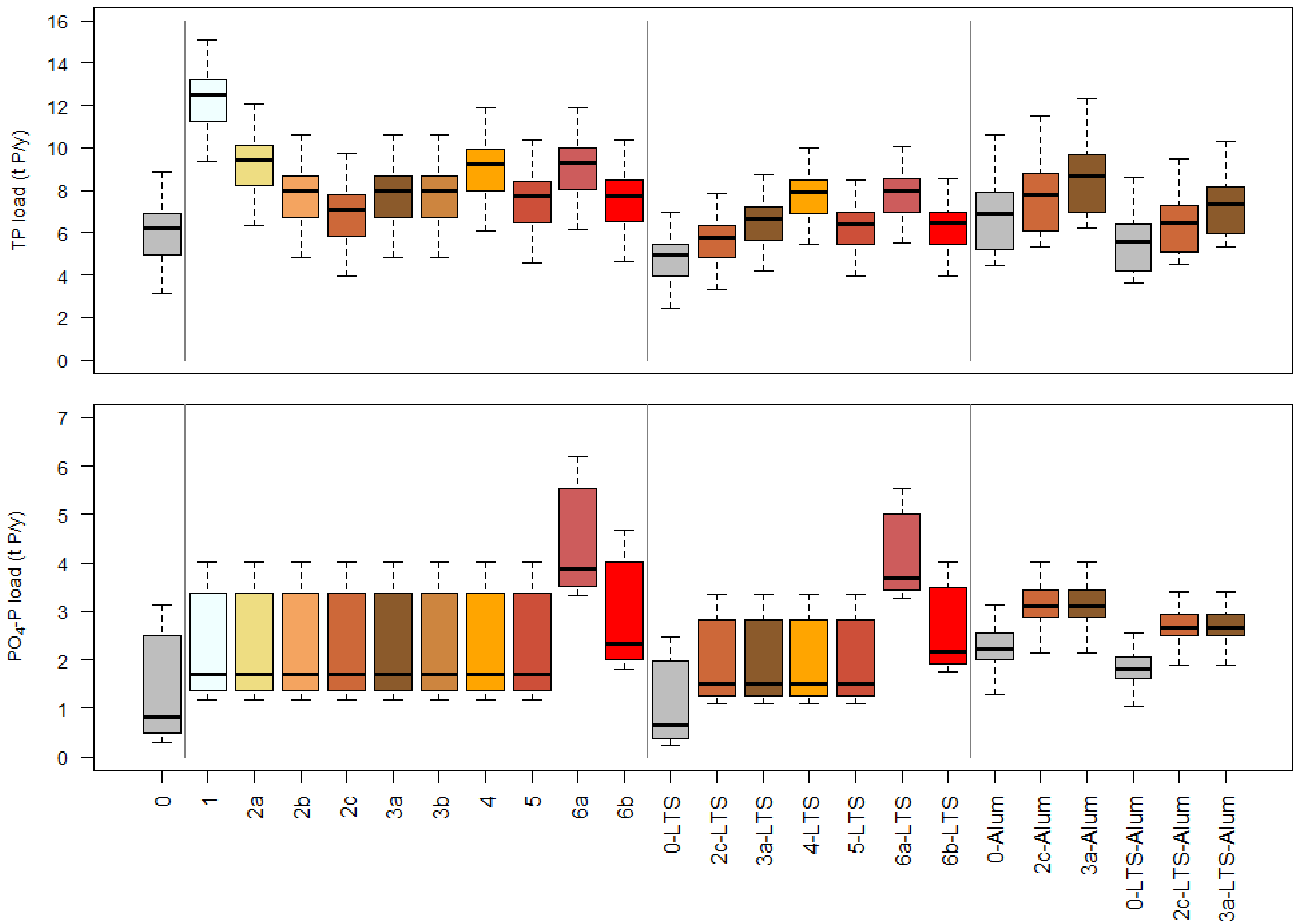

Figure 11 **Annual (2007–2014) total phosphorus (TP) and phosphate–phosphorus (PO4–P) loads in the Puarenga Stream that correspond to 1–D modelling scenarios (see Table 14 for scenario descriptions). Horizontal lines denote median values; boxes denote interquartile range; whiskers denote range.**

Comparison of concentrations with values designated in the NPS 2014 to assess in–stream effects on Ecosystem Health

Nitrate nitrogen (toxicity)

Attribute States relating to nitrate toxicity are defined on the basis of annual 95th percentile and annual median concentrations (Table 5). For both of these statistics, background nitrate concentrations in the Puarenga Stream correspond to the ranges that are designated for Attribute State A, with median concentrations corresponding to the upper (i.e., more impacted) end of the defined range (Figure 12; Table 21; Table 22). This State corresponds to high conservation value systems (Table 5).

Mass balance calculations using data for the study period (2007–2014) indicate that in–stream discharge of wastewater following treatment with the prescribed options will not cause this Attribute State to change, based on estimated annual 95th percentile concentrations (Table 21). On the basis of annual median nitrate concentrations, Options 1, 2, 4 and 5 are predicted to result in concentrations that correspond to the lower end of the range for Attribute State B during each of the eight years (Table 22). This State corresponds to the range at which some growth effect on up to 5% of species may occur (Table 5), although note that the 95th percentile value for these options still corresponds to Attribute State A (Table 21). Results for Option 3a show that median concentrations correspond to Attribute State A for all years. Results for Options 3b and 6 show that median concentrations correspond to either Attribute State A, or the lower end of the range (i.e. lower concentrations) for Attribute State B (Table 22).

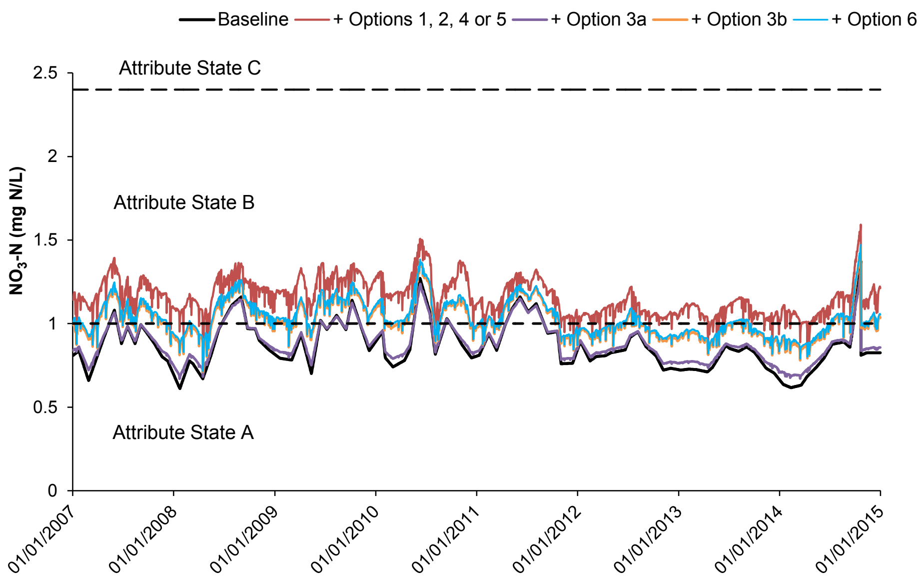

Figure 12 **Estimated mean daily nitrate–nitrogen concentrations in the Puarenga Stream for baseline conditions and following addition of treated wastewater. Dashed lines denote annual median values that correspond to Attribute States defined in the National Policy Statement for Freshwater Management 2014.**

Table 21 **Annual 95th percentile nitrate–nitrogen concentrations (2007–2014) based on: monthly water quality monitoring in the Puarenga Stream; estimated concentrations in the Puarenga Stream following addition of treated wastewater, and; Attribute States defined in the National Policy Statement for Freshwater Management 2014. Letters in parentheses denote Attribute States.**

		Annual 95th percentile (mg N/L)							
		2007	2008	2009	2010	2011	2012	2013	2014
Puarenga Stream monthly measurements		1.0 (A)	1.1 (A)	1.1 (A)	1.2 (A)	1.1 (A)	0.9 (A)	0.9 (A)	0.9 (A)
Scenarios	Baseline	1.0 (A)	1.1 (A)	1.1 (A)	1.2 (A)	1.1 (A)	0.9 (A)	0.9 (A)	1.1 (A)
	Baseline + Options 1, 2, 4, or 5	1.3 (A)	1.3 (A)	1.3 (A)	1.4 (A)	1.3 (A)	1.2 (A)	1.1 (A)	1.4 (A)
	Baseline + Option 3a	1.1 (A)	1.1 (A)	1.1 (A)	1.2 (A)	1.1 (A)	0.9 (A)	0.9 (A)	1.1 (A)
	Baseline + Option 3b	1.2 (A)	1.2 (A)	1.2 (A)	1.3 (A)	1.2 (A)	1.0 (A)	1.0 (A)	1.2 (A)
	Baseline + Option 6	1.2 (A)	1.2 (A)	1.2 (A)	1.3 (A)	1.2 (A)	1.1 (A)	1.0 (A)	1.3 (A)
NPS 2014 (annual values)	Attribute State A	≤ 1.5							
	Attribute State B	>1.5 and ≤ 3.5							
	Attribute State C	> 3.5 and ≤ 9.8							
	National Bottom Line	9.8							
	Attribute State D	> 9.8							

Table 22 **Annual median nitrate–nitrogen concentrations (2007–2014) based on: monthly water quality monitoring in the Puarenga Stream; estimated concentrations in the Puarenga Stream following addition of treated wastewater, and; Attribute States defined in the National Policy Statement for Freshwater Management 2014. Letters in parentheses denote Attribute States.**

		Median (mg N/L)							
		2007	2008	2009	2010	2011	2012	2013	2014
Puarenga Stream monthly measurements		0.9 (A)	0.9 (A)	0.9 (A)	0.9 (A)	0.9 (A)	0.8 (A)	0.7 (A)	0.8 (A)
Scenarios	Baseline	0.9 (A)	0.9 (A)	0.9 (A)	0.9 (A)	0.9 (A)	0.8 (A)	0.7 (A)	0.8 (A)
	Baseline + Options 1, 2, 4, or 5	1.1 (B)	1.2 (B)	1.2 (B)	1.2 (B)	1.2 (B)	1.1 (B)	1.1 (B)	1.1 (B)
	Baseline + Option 3a	0.9 (A)	0.9 (A)	0.9 (A)	0.9 (A)	1.0 (A)	0.8 (A)	0.8 (A)	0.8 (A)
	Baseline + Option 3b	1.0 (A)	1.0 (A)	1.1 (B)	1.1 (B)	1.1 (B)	1.0 (A)	0.9 (A)	1.0 (A)
	Baseline + Option 6	1.0 (A)	1.1 (B)	1.1 (B)	1.1 (B)	1.1 (B)	1.0 (A)	1.0 (A)	1.0 (A)
NPS 2014 (annual values)	Attribute State A	≤ 1.0							
	Attribute State B	> 1.0 and ≤ 2.4							
	Attribute State C	> 2.4 and ≤ 6.9							
	National Bottom Line	6.9							
	Attribute State D	> 6.9							

Ammonium nitrogen (toxicity)

Attribute States relating to ammonium toxicity are defined on the basis of annual maximum and annual median concentrations, based on pH of 8 and temperature of 20 °C (Table 6). For the purpose of the assessment, no pH or temperature corrections were applied to the Attribute State classifications. Ammonium toxicity increases with increasing temperature and pH (Wetzel 2001) and, since typical ambient pH and temperature in the Puarenga Stream are less than pH 8 and 20 °C respectively[9], the decision to use uncorrected Attribute State classifications represents a precautionary approach.

On the basis of both annual maximum and median concentrations, background ammonium concentrations in the Puarenga Stream correspond to Attribute State B (Figure 13; Table 23; Table 24). This State corresponds to the range at which some growth effects on up to 5% of species may occur (Table 6). Discharge of wastewater treated using any of the proposed options is not predicted to cause a change of Attribute State (Table 23; Table 24).

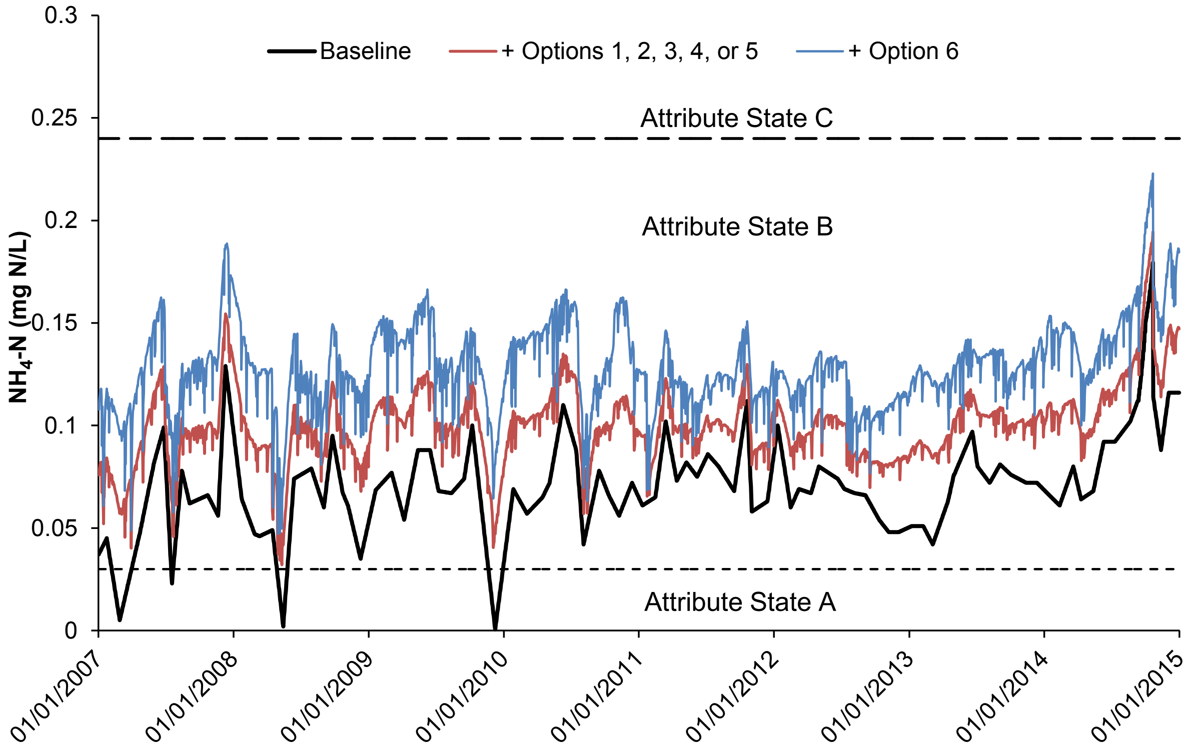

Figure 13 **Estimated mean daily ammonium nitrogen concentrations in the Puarenga Stream for baseline conditions and following addition of treated wastewater. Dashed lines denote annual median values that correspond to Attribute States defined in the National Policy Statement for Freshwater Management 2014.**

[9] Monitoring data collected by BoPRC during the study period indicates that pH in the Puarenga Stream is consistently < 8 (median = 6.6, range = 6.2 to pH 7.1) and temperature is less than 20 °C, with the exception of very occasional periods during summer afternoons (median = 11.4 °C, range = 11.4 to 20.7 °C)

Table 23 **Annual maximum ammonium–nitrogen concentrations (2007–2014) based on: monthly water quality monitoring in the Puarenga Stream; estimated concentrations in the Puarenga Stream following addition of treated wastewater, and; Attribute States defined in the National Policy Statement for Freshwater Management 2014. Letters in parentheses denote Attribute States.**

		Annual maximum (mg N/L)							
		2007	2008	2009	2010	2011	2012	2013	2014
Puarenga Stream monthly measurements		0.13 (B)	0.10 (B)	0.10 (B)	0.11 (B)	0.11 (B)	0.10 (B)	0.10 (B)	0.23 (B)
Scenarios	Baseline	0.13 (B)	0.10 (B)	0.10 (B)	0.11 (B)	0.11 (B)	0.10 (B)	0.10 (B)	0.18 (B)
	Baseline + Options 1, 2, 3, 4 or 5	0.15 (B)	0.13 (B)	0.13 (B)	0.13 (B)	0.13 (B)	0.11 (B)	0.12 (B)	0.19 (B)
	Baseline + Option 6	0.19 (B)	0.17 (B)	0.17 (B)	0.17 (B)	0.15 (B)	0.14 (B)	0.14 (B)	0.22 (B)
NPS 2014 (annual values)	Attribute State A	≤ 0.03							
	Attribute State B	> 0.03 and ≤ 0.24							
	Attribute State C	> 0.24 and ≤ 1.3							
	National Bottom Line	1.30							
	Attribute State D	> 1.30							

Table 24 **Annual median ammonium–nitrogen concentrations (2007–2014) based on: monthly water quality monitoring in the Puarenga Stream; estimated concentrations in the Puarenga Stream following addition of treated wastewater, and; Attribute States defined in the National Policy Statement for Freshwater Management 2014. Letters in parentheses denote Attribute States.**

		Median (mg N/L)							
		2007	2008	2009	2010	2011	2012	2013	2014
Puarenga Stream monthly measurements		0.06 (B)	0.06 (B)	0.07 (B)	0.07 (B)	0.07 (B)	0.07 (B)	0.07 (B)	0.09 (B)
Scenarios	Baseline	0.06 (B)	0.06 (B)	0.07 (B)	0.07 (B)	0.08 (B)	0.07 (B)	0.07 (B)	0.09 (B)
	Baseline + Options 1, 2, 3, 4 or 5	0.09 (B)	0.09 (B)	0.10 (B)	0.10 (B)	0.10 (B)	0.09 (B)	0.10 (B)	0.12 (B)
	Baseline + Option 6	0.12 (B)	0.12 (B)	0.14 (B)	0.14 (B)	0.12 (B)	0.11 (B)	0.13 (B)	0.15 (B)
NPS 2014	Attribute State A	≤ 0.03							
	Attribute State B	> 0.03 and ≤ 0.24							
	Attribute State C	> 0.24 and ≤ 1.3							
	National Bottom Line	1.30							
	Attribute State D	> 1.30							

Dissolved oxygen

Background dissolved concentrations measured in the Puarenga Stream by BoPRC generally corresponded to Attribute State A, with only 2 of the 91 measurements (2%) slightly less than the value that defines the boundary of States A and B (Figure 14). Attribute State A corresponds to a condition of "*no stress caused by low dissolved oxygen on any aquatic organisms that are present at matched reference (near-pristine) sites*" (Table 7). Calculations showed that, relative to this baseline, a worst case scenario involving addition of anoxic treated wastewater would cause higher frequency of measurements that correspond to Attribute State B, with the majority (73%) of measurements still corresponding to Attribute State A. Attribute State B corresponds to a state of "*occasional minor stress on sensitive organisms caused by short periods (a few hours each day) of lower dissolved oxygen* [causing] *risk of reduced abundance of sensitive fish and macroinvertebrate species"* (Table 7).

It is important to note that the above comparisons were made based on monthly spot measurements of dissolved oxygen concentrations. The concentrations specified in the National Policy Statement are defined as minimum values, which require near–continuous measurements to calculate (Table 7; New Zealand Government 2014). Such data were unavailable, and the analysis may therefore underestimate impacts in cases where spot measurements are significantly higher than daily minima.

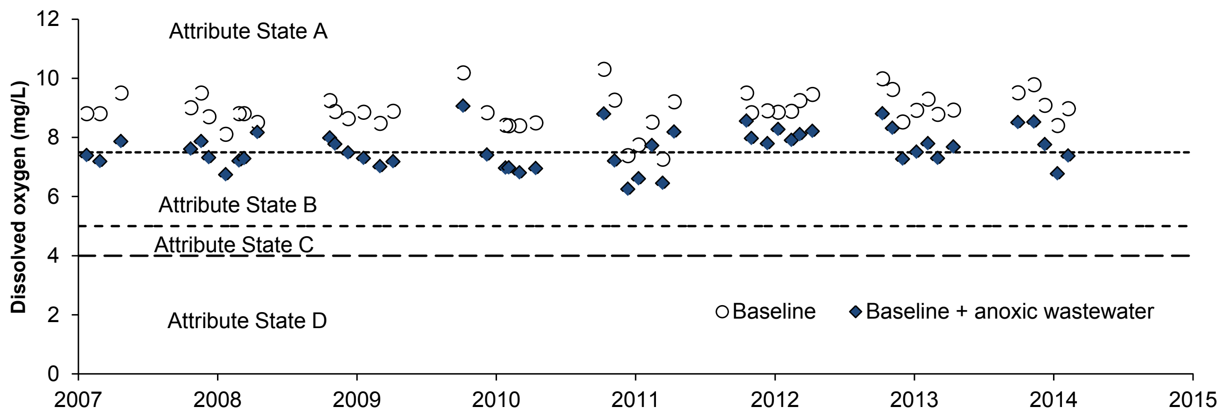

Figure 14 **Monthly measurements of dissolved oxygen concentration in the lower Puarenga Stream collected during November–April by BoPRC (circles), compared with estimated concentrations following addition of anoxic wastewater (diamonds). Dashed lines denote 1–day minimum values that correspond to Attribute States defined in the National Policy Statement for Freshwater Management 2014.**

E. coli

Historical *E. coli* concentrations measured by BoPRC show moderately–high temporal variability (Figure 15). Consequently, Attribute States were determined for individual years to characterise the baseline conditions in the Puarenga Stream with respect to this analyte (Table 25). The *E. coli* Attribute State was B for six of the eight years in the baseline period. Attribute State B corresponds to a low (<1%) risk of infection to water users (Table 8). Concentrations corresponded to either Attribute States A or D during a single year (see Table 8 for details).

No specific data were available for projected *E. coli* concentrations for each of the treatment options. Data were provided, however, of *E. coli* concentrations measured following membrane bioreactor treatment at the current WWTP (K. Brian, pers. comm. 2015a; Table 26). The median count is zero and these concentrations are very low compared with the concentrations measured in the Puarenga Stream, which have an annual median count of 29/100 mL to 185/100 mL[10]. If these concentrations are representative of those corresponding to the proposed options, then there is predicted to be a neutral to very minor negative effect on the current risk to human health related to *E. coli* in the lower Puarenga Stream.

A qualifier to this is that the standard deviation is much greater than the mean, which indicates that one or more values in the dataset are much higher than the mean. The raw data used to derive these statistics were, however, unavailable to inform this assessment. Recommendations to address the uncertainty in this aspect of the assessment are outlined in the Discussion.

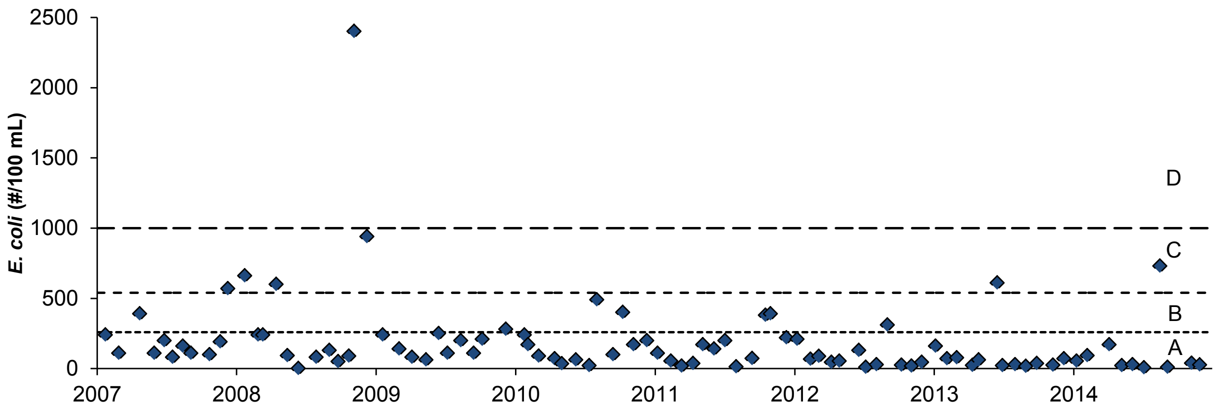

Figure 15 **Monthly measurements of *E. coli* concentration in the lower Puarenga Stream collected by BoPRC. Dashed lines denote values (defined as both annual median and annual 95th percentile) that correspond to Attribute States defined in the National Policy Statement for Freshwater Management 2014.**

Table 25 ***E. coli* concentrations in the lower Puarenga Stream measured by BoPRC and associated Attribute States, as defined in the National Policy Statement for Freshwater Management 2014.**

Year	Annual median (#/100 mL)	95th percentile (#/100 mL)	Attribute State
2007	160	480	B
2008	185	1597	D
2009	170	266.5	B
2010	135	458.5	B
2011	125	384.5	B
2012	49.5	255	A
2013	50	362.5	B
2014	29	450	B

Table 26 **Summary of E. coli concentrations following treatment with the current membrane bioreactor (K. Brian, pers. comm. 2015a).**

Statistic	E.coli (#/ 100 mL)
n	277
95th percentile	6.2
Median	0
Mean	5.6
Std. Dev.	61

Periphyton

No baseline periphyton data were available for the lower Puarenga Stream to inform this assessment and the baseline Attribute State for this parameter is currently undetermined.

Bottom up control by nutrients typically exerts strong control on periphyton biomass accumulation in New Zealand rivers, particularly during summer (Biggs and Kilroy 2000). The proposed options will result in minor increases to background dissolved nutrients in a short (< 1.5 km) reach of the Puarenga Stream if wastewater is discharged to either of sites 1, 2 or 3 (Map 2). The potential for this discharge to contribute to periphyton growth will depend on the suitability of the substrate and the relative importance of other controls on periphyton growth in the stream, notably light (influenced by stream depth and optical transmissivity) and scouring (influenced by peak stream velocity during storm flow periods). Based on author observations, the substrate characteristics of the lower reach of the Puarenga Stream are deemed to provide limited potential for periphyton proliferation; the bed predominantly comprises fine–textured sediments and the extent of natural hard surfaces such as exposed boulders, cobbles or bedrock limits periphyton growth. Thus, impacts associated with periphyton are predicted to be neutral.

One–dimensional lake water quality modelling

Calibration and validation

Overview

Satisfactory model performance was achieved for the 2007–2014 study period with DYRESM–CAEDYM parameter values set to those assigned in a recent study by Hamilton *et al*. (2015), who calibrated the model for the period 2004–2007. The only exception was that it was necessary to reduce the maximum sediment release rate of ammonium nitrogen[11] to a value that was used in an earlier model application (Hamilton *et al*. 2012) to improve the model fit with measured total nitrogen concentrations. Details of the other parameter values are tabulated in Hamilton *et al*. (2015).

[11] Value reduced from 0.5 $g/m^2/day$ to 0.2 $g/m^2/day$.

Overall, model performance was comparable with other model applications to Lake Rotorua (Burger *et al*. 2008; Hamilton *et al*. 2012; Hamilton *et al*. 2015), and with that of water quality model applications more generally (Arhonditsis and Brett 2003).

Water level

There was a very good match between modelled and measured water levels (Figure 16).

1.1.1.1. Temperature and dissolved oxygen

Similarly, there was a very good match between modelled and measured water temperatures and a good match between modelled and measured dissolved oxygen concentrations (Figure 17, Figure 18). In particular, the model typically reproduced summer deoxygenation events in the bottom waters with high accuracy (Figure 18).

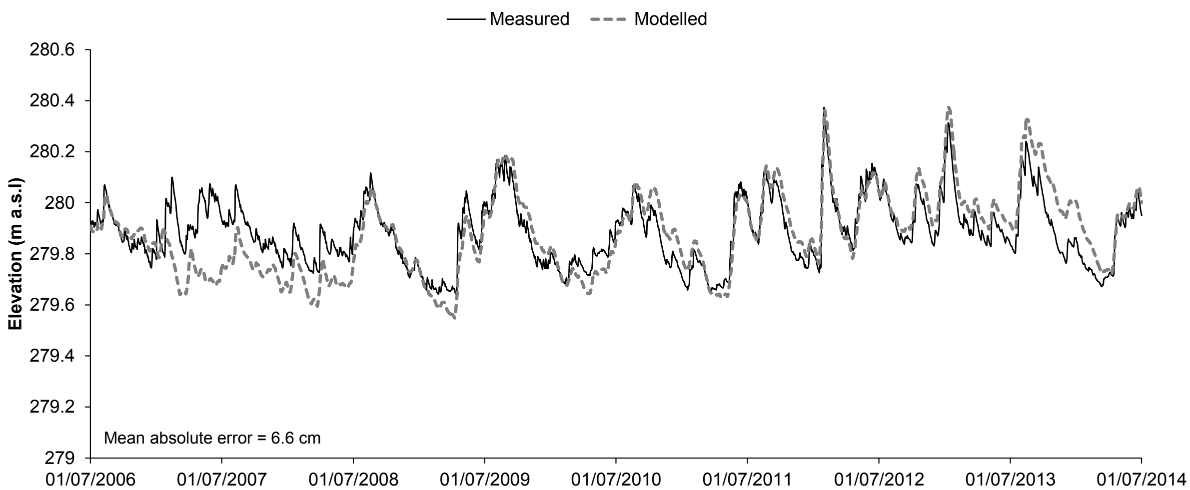

Figure 16 **Modelled and measured water levels during the 1–D modelling period'.**

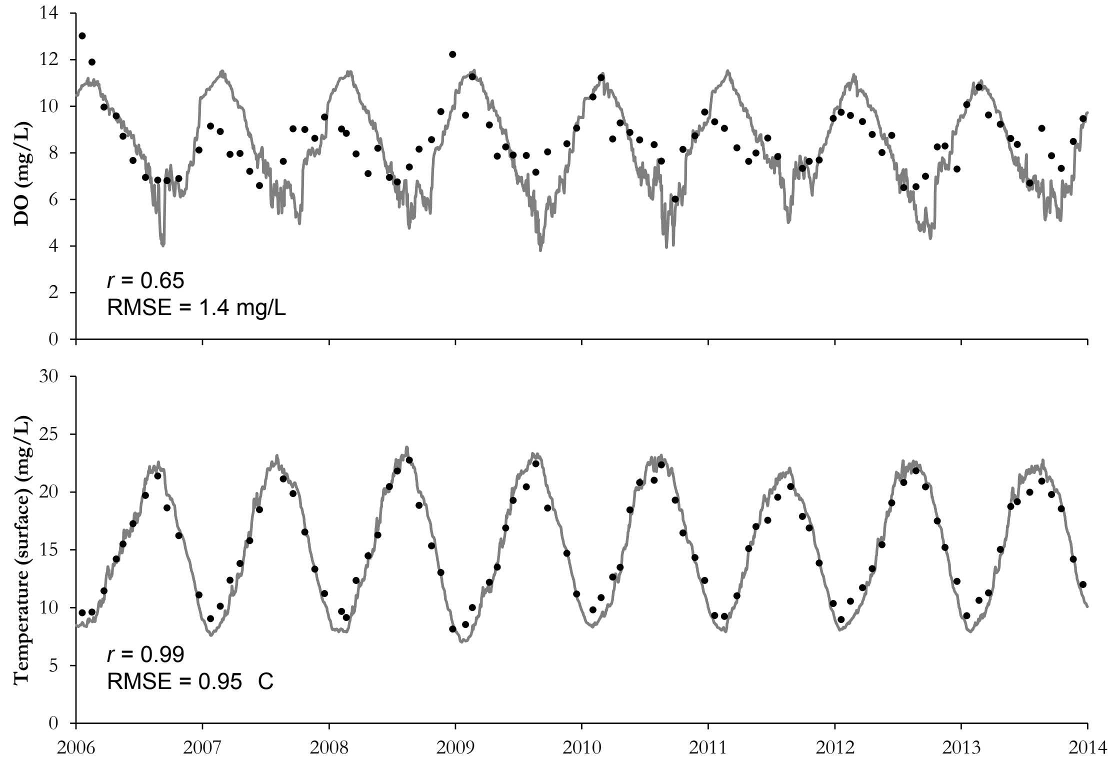

Figure 17 **Comparisons of measured (circles) and modelled (line) surface concentrations of dissolved oxygen (DO) and temperature.**

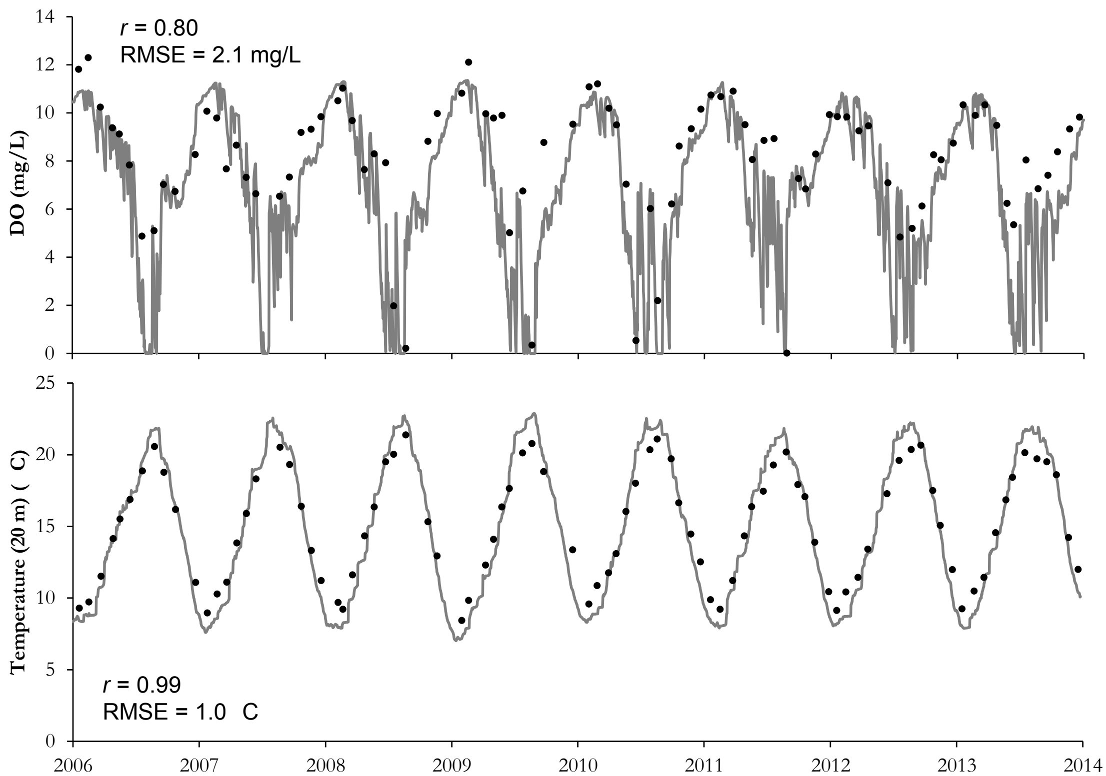

Figure 18 **Comparisons of measured (circles) and modelled (line) concentrations of dissolved oxygen (DO) and temperature in bottom waters (depth = 20 m).**

Chlorophyll* a *and nutrients

The model reproduced the magnitude of the chlorophyll *a* measurements reasonably well (Figure 19; Table 27) although inter–annual differences were not well–produced, most notably for the validation period (r = -0.06; Table 27). Both trends and magnitude were reproduced satisfactorily for most nutrient fractions (Figure 19; Figure 20; Table 27). Relatively high concentrations of dissolved inorganic nitrogen fractions were observed in the surface water measurements after 2011; these were typically not reproduced (Figure 20).

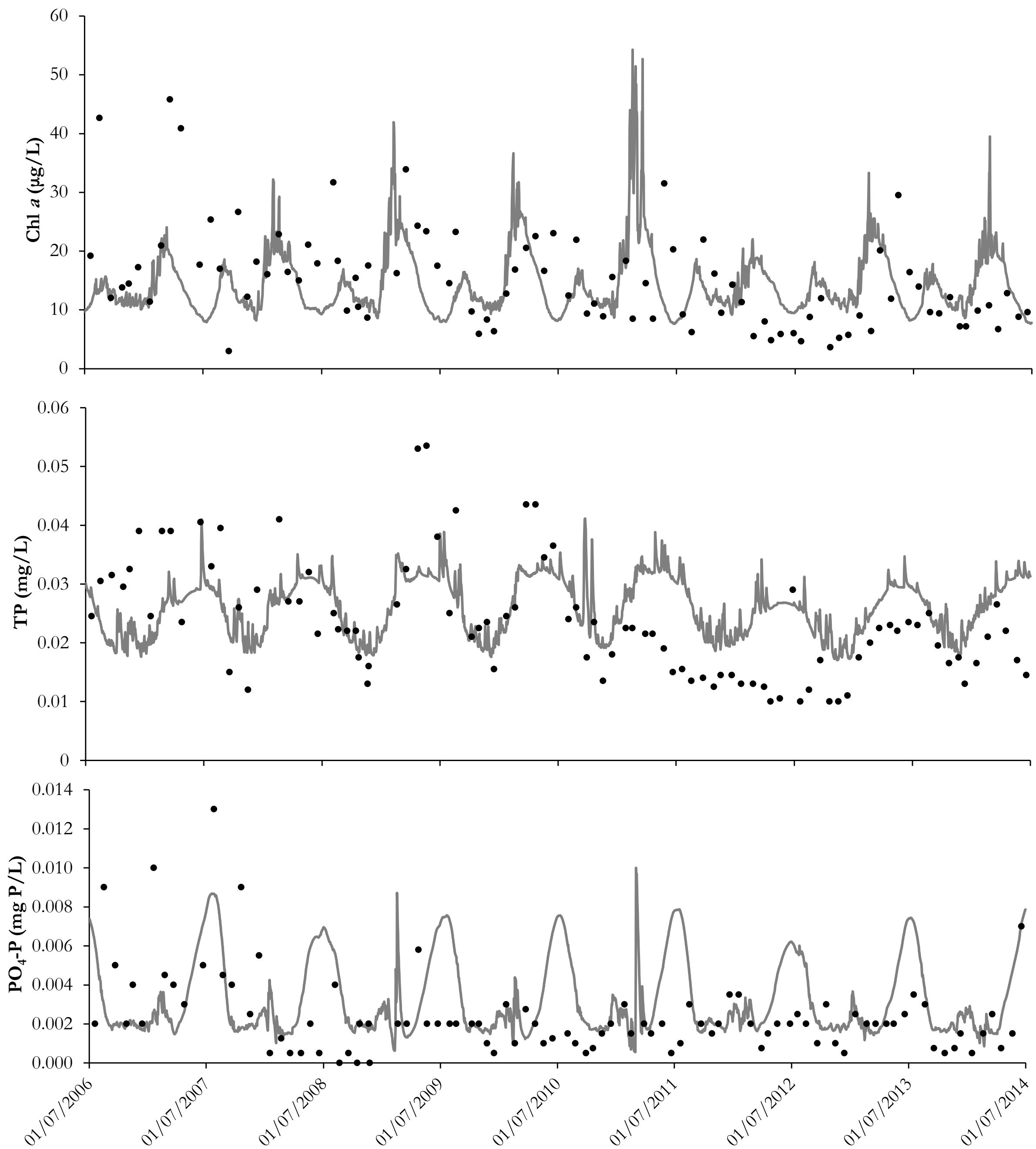

Figure 19 **Comparisons of measured (circles) and modelled (line) surface concentrations of chlorophyll *a* and phosphorus fractions. The detection limit for PO_4–P was 0.008 mg P/L prior to October 2009, and 0.001 mg P/L thereafter.**

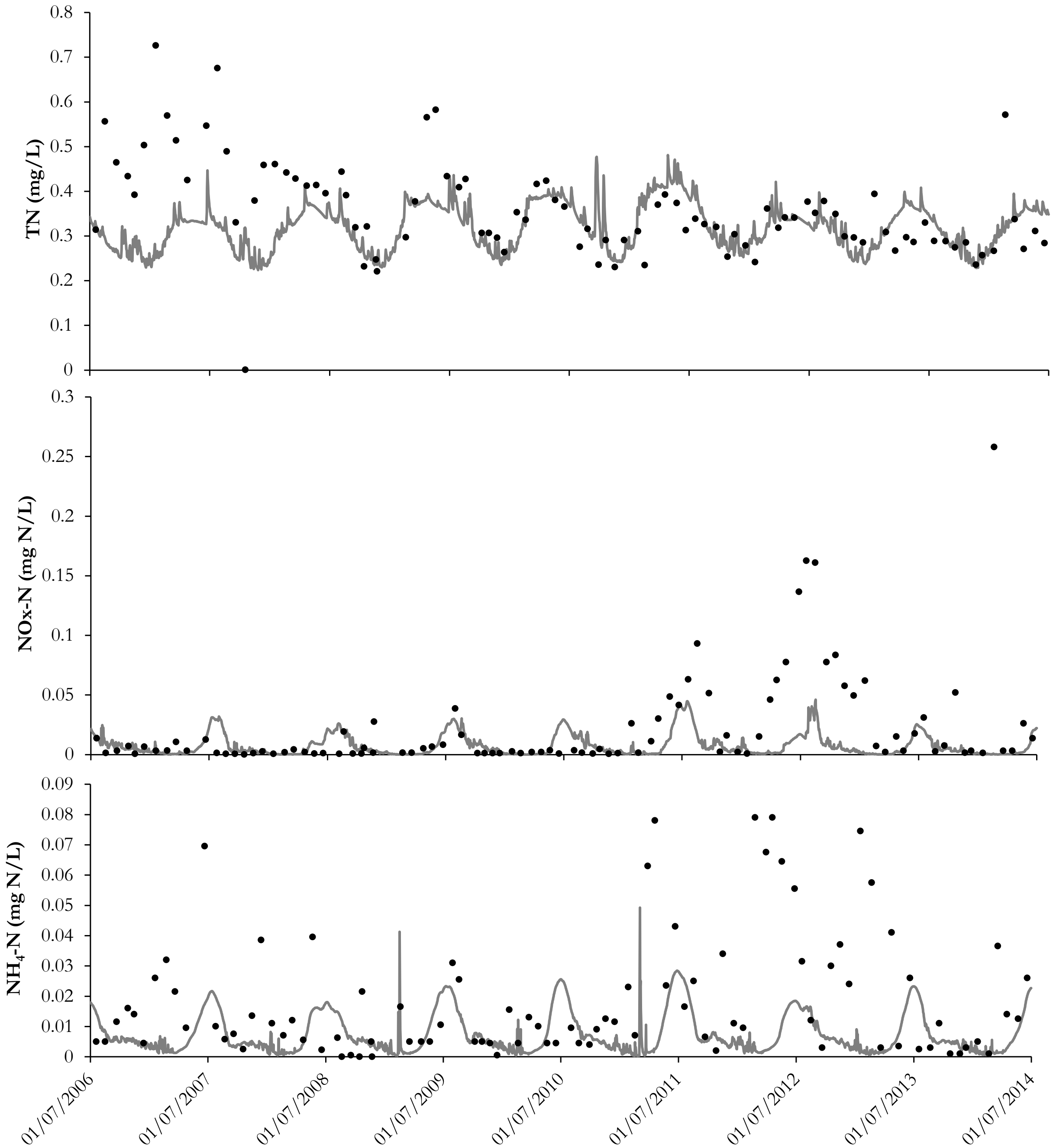

Figure 20 **Comparisons of measured (circles) and modelled (line) surface water nitrogen concentrations.**

Table 27 Model performance statistics for calibration (2007–2010) and validation (2011–2014) periods (chlorophyll *a* and nutrients).

		SURFACE		20 m	
		2007-2010	2011-2014	2007-2010	2011-2014
Chl *a*	*r*	0.22	-0.06		
(µg/L)	RMSE	10.31	9.01		
	Mean error	-4.71	2.82		
TP	*r*	0.23	0.48	0.38	0.39
(mg/L)	RMSE	0.03	0.01	0.02	0.01
	Mean error	-0.01	0.01	-0.01	0.01
PO_4-P	*r*	0.12	0.34	0.63	0.34
(mg P/L)	RMSE	0.003	0.002	0.010	0.007
	Mean error	<0.001	0.001	-0.001	0.003
TN	*r*	0.20	0.31	0.24	0.37
(mg/L)	RMSE	0.15	0.06	0.17	0.07
	Mean error	-0.10	0.01	-0.11	0.01
NO_3-N	*r*	0.37	0.32	0.29	0.47
(mg N/L)	RMSE	0.009	0.058	0.009	0.051
	Mean error	0.003	-0.030	0.001	-0.034
NH_4-N	*r*	0.20	-0.05	0.54	0.41
(mg N/L)	RMSE	0.014	0.031	0.092	0.092
	Mean error	-0.005	-0.017	-0.028	-0.037

TLI_3

Modelled annual TLI_3 values approximated measurements (Figure 21; Table 28), reflecting the satisfactory performance of the model with regard to simulating the three constituent parameters (Table 27). Generally, the model did not simulate inter–annual trends in the measured TLI_3 well. In particular, error was high in 2007 when TLI_3 was underestimated by 0.54 units, and in 2012 when TLI_3 was overestimated by 0.34 units. These errors likely reflect variability in alum dosing; this is discussed further in the Discussion where implications for model application are outlined.

A measure of good model performance has previously been identified as an ability to model the measured TLI_3 value with an error of $\leqslant$ 0.1 units (Hamilton *et al*. 2012). This was only achieved for one year (2010). However, the eight–year average measured TLI_3 for the period was just 0.01 units greater than the modelled value.

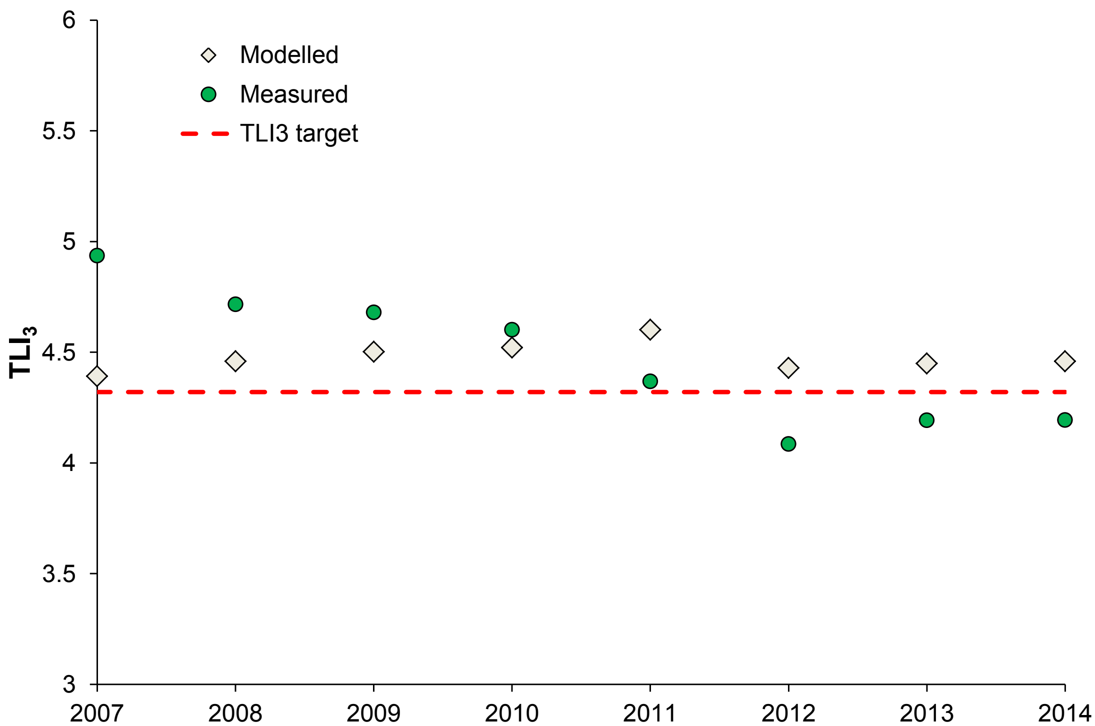

Figure 21 **Comparison of modelled and measured annual TLI_3. The dashed red line denotes the TLI_3–adjusted target for Lake Rotorua.**

Table 28 **Summary of model performance for simulation of annual TLI_3.**

	Calibration (2007-2010)	Validation (2011-2014)	Eight year period (2007-2014)
Measured TLI_3 (mean)	4.73	4.21	4.47
Modelled TLI_3 (mean)	4.47	4.48	4.48
r	-0.53	0.75	-0.10
RMSE	0.32	0.28	0.30
Mean error	-0.27	0.27	0.00

Modelled external loads

External nutrient loads that were represented in the baseline model scenario are presented in Figure 22 and Figure 23, alongside loads for individual treatment options. In broad terms, the figures show that the nutrient loads associated with each option are comparable with those of a major stream inflow.

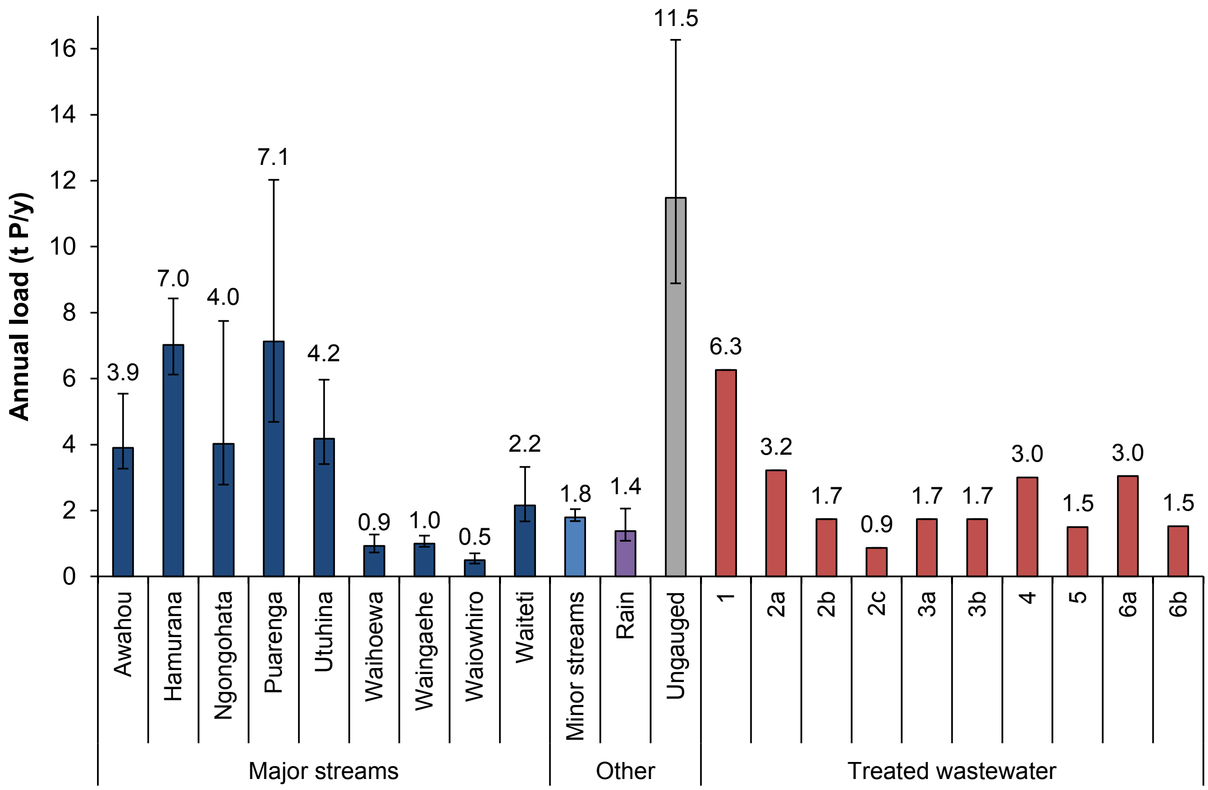

Figure 22 **Summary of mean external phosphorus loads used as forcing data in baseline model simulations. Puarenga Stream loads do not reflect attenuation by alum. Vertical lines denote between–year variations.**

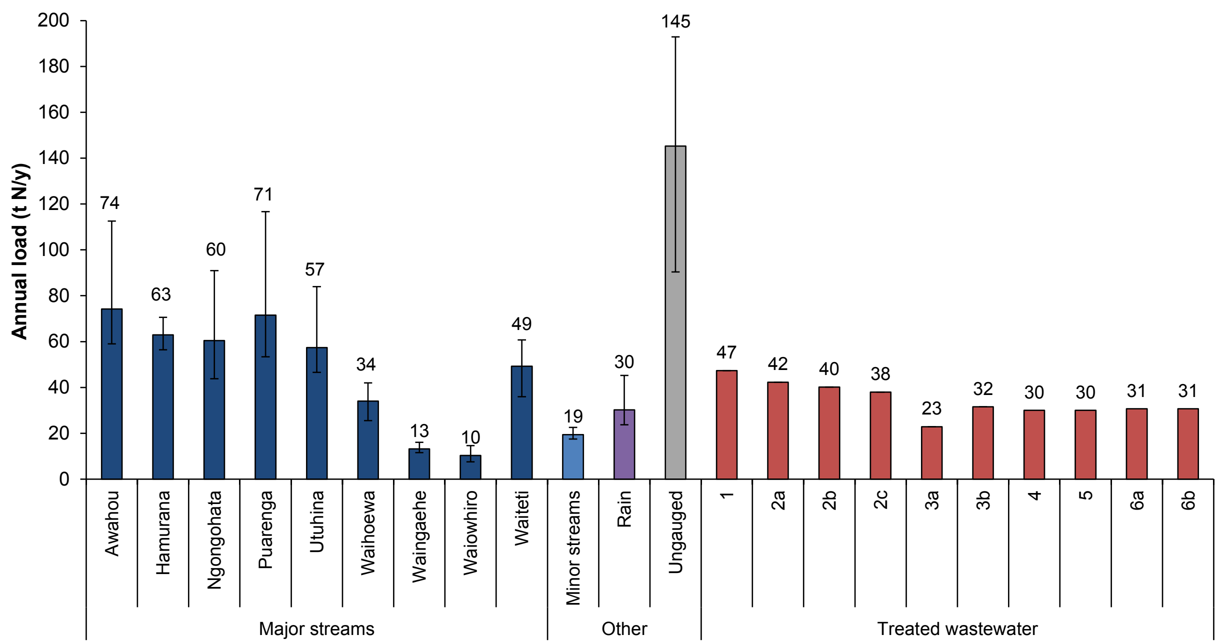

Figure 23 **Summary of mean external nitrogen loads used as forcing data in baseline model simulations. Vertical lines denote between–year variations.**

Simulated TLI_3 for scenarios

Simulated eight–year mean TLI_3 values for each scenario are presented in Table 29. Magnitudes of departure from the baseline scenario are presented in Figure 24.

These results indicate that all proposed scenarios of treated wastewater have a very minor effect on the TLI_3, relative to the baseline scenario. The eight–year mean results in Table 29 show that the treatment options result in an increase of between 0.01 and 0.02 TLI_3 units relative to the baseline scenario. Table 30 provides a summary for individual years of differences between model predictions relative to the baseline scenario. These data highlight differences between the individual treatment options in finer detail than the eight year–mean values presented in Table 29; however, the differences between the options are still very small, especially when compared with the magnitude of model error (Figure 21). The scenario involving addition of 'pure' water (#29; Table 29) highlights the occurrence of very minor water quality improvements associated with flushing effects. Results for this scenario provide insight into why some scenarios actually exhibit extremely minor improvements in TLI_3 for a small number of years (notably 2011) compared with the baseline period.

Neither of the scenarios involving either discharge of anoxic treated wastewater, or discharge to the lake bed, had an appreciable effect on modelled TLI_3. The scenarios involving removal of LTS loads highlight a very small effect due to this action; TLI_3 is 0.02 less for the baseline scenario when LTS loads are removed. By contrast, the scenarios involving cessation of alum dosing to streams had a much more substantial effect, with all of these scenarios resulting in an increase of ~0.5 TLI_3 units.

Table 29 **Summary of predicted TLI_3 values. Each value is the mean of eight annual TLI_3 values for 2007–2014.**

#	Scenario	Treatment upgrade	Discharge depth (m)	Legacy LTS loads?	Alum simulated?	Mean annual TLI_3
1	1D_0 (Baseline)	None	n/a	✓	✓	4.48
2	1D_1_Surface	1	0	✓	✓	4.49
3	1D_2a_Surface	2a	0	✓	✓	4.48
4	1D_2b_Surface	2b	0	✓	✓	4.49
5	1D_2c_Surface	2c	0	✓	✓	4.48
6	1D_3a_Surface	3a	0	✓	✓	4.49
7	1D_3b_Surface	3b	0	✓	✓	4.49
8	1D_4_Surface	4	0	✓	✓	4.47
9	1D_5_Surface	5	0	✓	✓	4.48
10	1D_6a_Surface	6a	0	✓	✓	4.50
11	1D_6b_Surface	6b	0	✓	✓	4.49
12	1D_2c_Surface - DO	2c	0	✓	✓	4.48
13	1D_3a_Surface - DO	3a	0	✓	✓	4.49
14	1D_2c_Bed	2c	10	✓	✓	4.49
15	1D_3a_Bed	2c	10	✓	✓	4.49
16	1D_0 - LTS	None	n/a	x	✓	4.46
17	1D_2c_Surface - LTS	2c	0	x	✓	4.47
18	1D_3a_Surface - LTS	3a	0	x	✓	4.48
19	1D_4_Surface - LTS	4	0	x	✓	4.47
20	1D_5_Surface - LTS	5	0	x	✓	4.47
21	1D_6a_Surface - LTS	6a	0	x	✓	4.49
22	1D_6b_Surface - LTS	6b	0	x	✓	4.46
23	1D_0 - Alum	None	n/a	✓	x	5.03
24	1D_2c_Surface - Alum	2c	0	✓	x	5.05
25	1D_3a_Surface - Alum	3a	0	✓	x	5.05
26	1D_0 - LTS - Alum	None	n/a	x	x	5.02
27	1D_2c_Surface - LTS - Alum	2c	0	x	x	5.03
28	1D_3a_Surface - LTS - Alum	3a	0	x	x	5.05
29	1D_0 + 'pure' wastewater	None	n/a	✓	✓	4.47
-	Measured		Mean of annual TLI3, 2007-2014			4.47

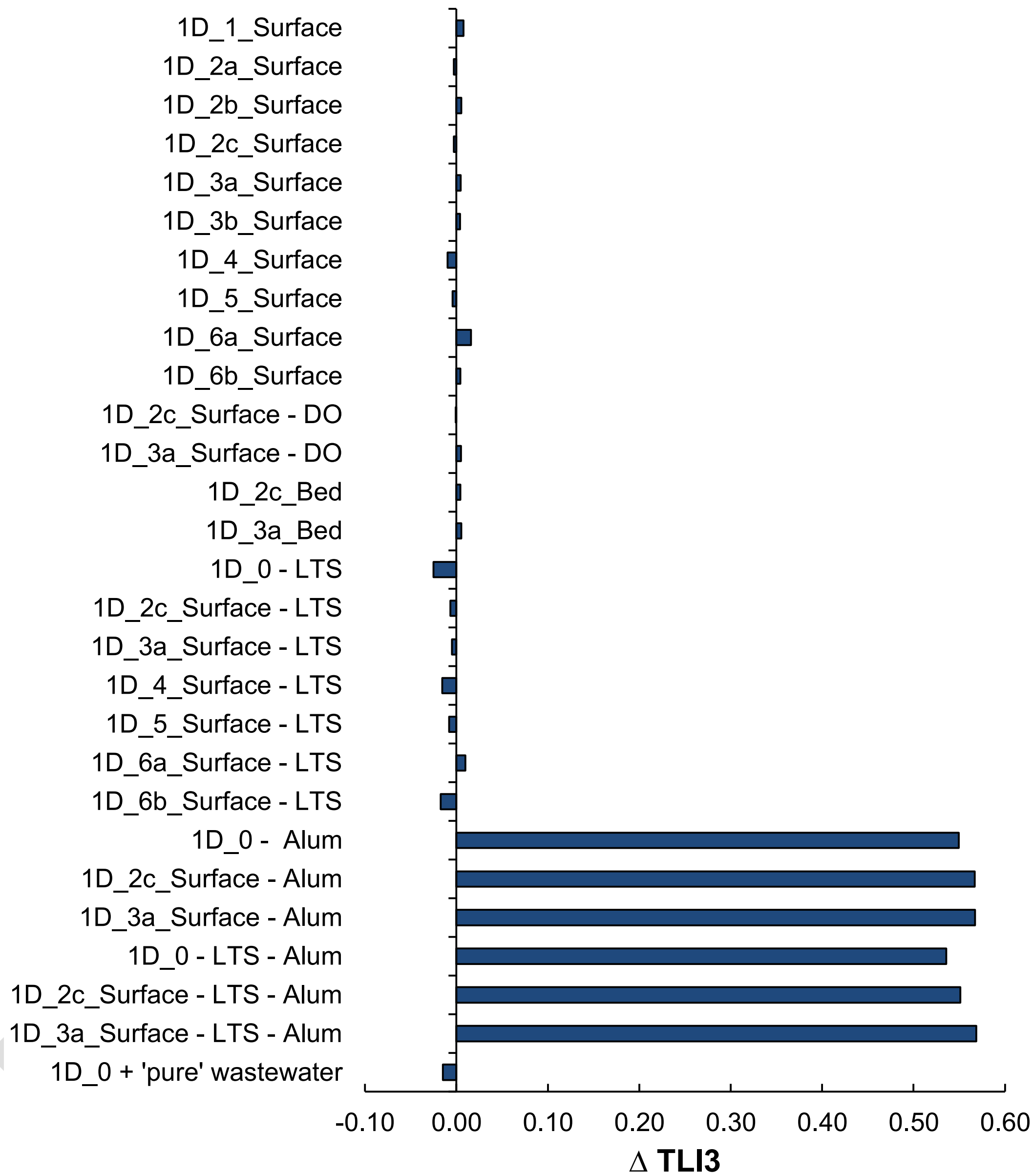

Figure 24 **Change in eight–year mean annual TLI_3 for each 1–D scenario (Table 19) relative to the baseline simulation (no wastewater added).**

Table 30 **Percentage change in annual TLI_3 for each 1–D scenario (Table 19) relative to the baseline simulation (no wastewater added) for individual years. Shading is proportional to relative differences.**

Year	1D_1_Surface	1D_2a_Surface	1D_2b_Surface	1D_2c_Surface	1D_3a_Surface	1D_3b_Surface	1D_4_Surface	1D_5_Surface	1D_6a_Surface	1D_6b_Surface	1D_2c_Surface - DO	1D_3a_Surface - DO	1D_2c_Bed
2007	0.13	0.05	-0.47	0.05	-0.05	-0.39	-0.55	-0.24	-0.06	0.14	-0.22	-0.17	0.11
2008	0.56	-0.02	0.13	-0.02	0.19	-0.23	-0.23	-0.25	0.27	-0.32	0.49	-0.15	0.26
2009	-0.24	-0.45	-0.96	-0.45	-0.29	-0.16	-0.80	-0.82	0.04	-0.54	-0.53	-0.15	-0.51
2010	-0.43	-0.92	0.66	-0.92	-0.16	0.26	-0.92	0.39	0.26	0.33	-0.45	0.27	-0.34
2011	0.13	-0.15	0.48	-0.15	0.27	0.47	0.44	-0.14	0.90	0.54	-0.09	0.39	0.33
2012	0.49	0.45	0.44	0.45	0.28	0.21	0.48	0.22	0.88	0.51	0.33	0.47	0.71
2013	0.46	0.50	0.37	0.50	0.51	0.51	0.03	0.10	0.61	0.18	0.30	0.38	0.18
2014	0.31	0.08	0.31	0.08	0.13	0.04	-0.20	-0.03	-0.05	-0.06	0.05	-0.12	0.03
Mean	**0.18**	**-0.06**	**0.12**	**-0.06**	**0.11**	**0.09**	**-0.22**	**-0.09**	**0.36**	**0.10**	**-0.01**	**0.11**	**0.10**

Table 30 continued.

Year	1D_3a_ Bed	1D_0 - LTS	1D_2c_Surface - LTS	1D_3a_Surface - LTS	1D_4_Surface - LTS	1D_5_Surface - LTS	1D_6a_Surface - LTS	1D_6b_Surface - LTS	1D_0 - Alum	1D_2c_Surface - Alum	1D_3a_Surface - Alum	1D_0 - LTS - Alum	1D_2c_Surface - LTS - Alum	1D_3a_Surface - LTS - Alum	1D_0 + 'pure' wastewater
2007	-0.12	-0.67	-0.56	-0.05	-0.34	-0.31	0.20	-0.55	12.30	12.57	11.66	11.98	11.95	11.92	-0.25
2008	0.74	-0.39	0.02	-0.07	-0.22	-0.50	0.07	-0.12	11.43	12.29	12.55	10.66	11.58	12.06	-0.27
2009	-0.35	-1.08	-0.61	-0.58	-0.93	-0.34	-0.01	-0.86	10.76	10.96	11.82	10.13	11.21	11.15	-1.28
2010	-0.42	-0.91	-0.21	-0.36	-1.21	-0.65	0.55	-1.32	12.07	12.97	12.76	12.49	12.31	13.21	0.40
2011	0.03	-0.19	0.18	-0.57	-0.33	0.10	0.14	-0.29	12.48	12.84	12.66	11.96	12.51	13.22	-0.16
2012	0.44	-0.27	0.26	0.45	0.40	0.43	0.59	-0.18	14.58	14.54	14.39	13.78	14.32	14.81	-0.07
2013	0.48	-0.34	0.08	0.45	0.08	0.14	0.11	0.28	12.82	12.97	13.17	13.01	13.00	12.88	-0.26
2014	0.20	-0.62	-0.36	-0.13	-0.20	-0.28	0.14	-0.04	11.68	12.08	12.24	11.65	11.52	12.29	-0.72
Mean	**0.12**	**-0.56**	**-0.15**	**-0.11**	**-0.34**	**-0.18**	**0.22**	**-0.38**	**12.26**	**12.65**	**12.66**	**11.96**	**12.30**	**12.69**	**-0.33**

Predicted nutrient limitation status of phytoplankton

The values for the simulated nitrogen and phosphorus limitation functions that partly control phytoplankton growth (ƒ(N) and ƒ(P) respectively; see Section 0) were examined to gain insight into the relative importance of each of these nutrients in influencing phytoplankton biomass accumulation (Figure 25). Under the baseline scenario, the functions indicate that phosphorus limitation was slightly more dominant (the values were lower) for the majority of the period, although the values were frequently very similar during late summer to autumn. When the representation of alum dosing was removed, the values showed that nitrogen limitation was generally the most dominant.

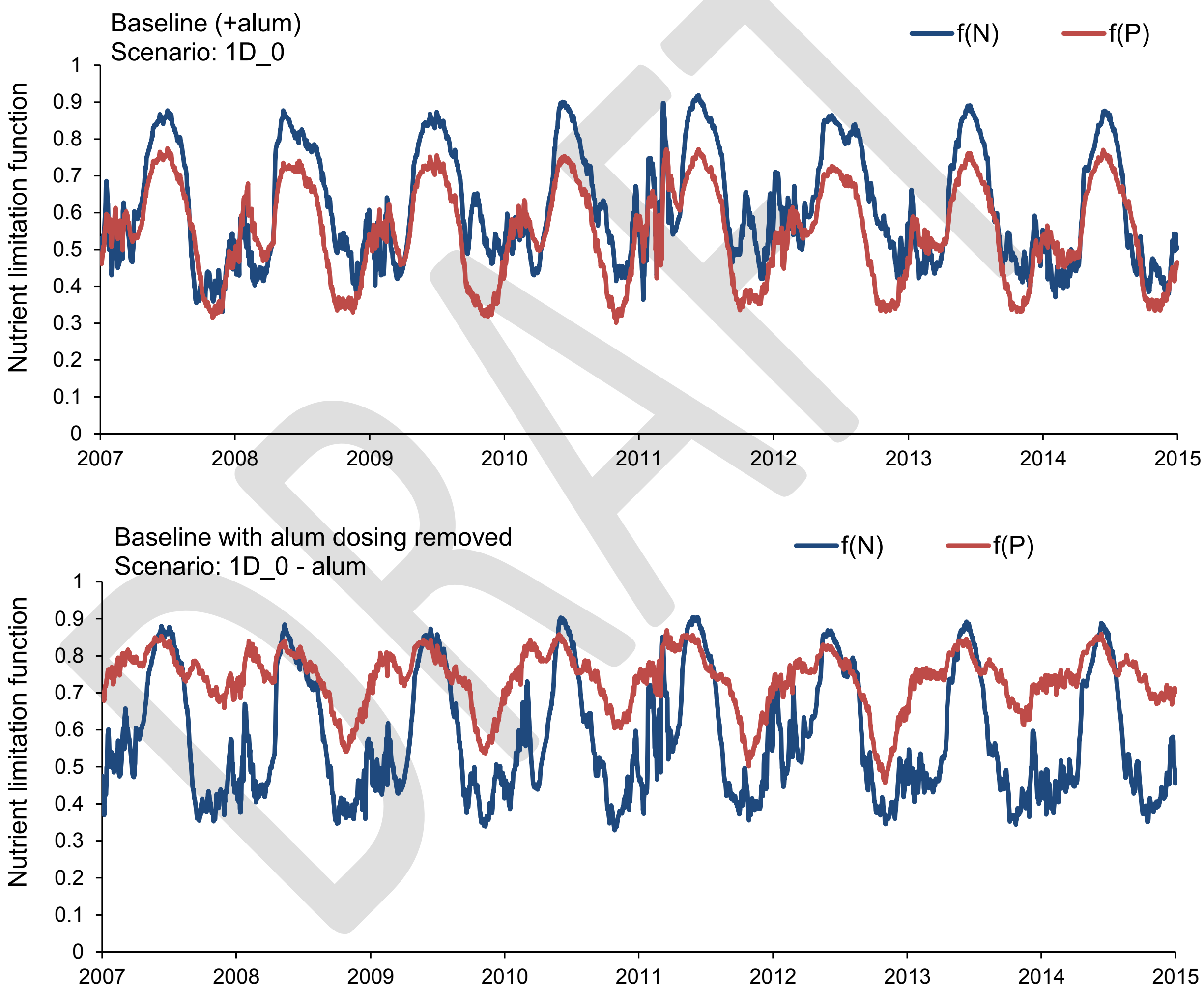

Figure 25 **Nitrogen and phosphorus limitation functions corresponding to the baseline scenario with (1D_0) and without (1D_0 - alum) alum dosing effects simulated.**

Comparison of concentrations with values designated in the NPS 2014 to assess in–lake effects on Ecosystem Health

Table 31 presents comparisons of model output with values designated for Attribute States for the three parameters that were assessed.

Consistent with the very minor effects on TLI_3 that were observed (Section 0), no changes were predicted to occur to the modelled baseline Attribute States for each of the scenarios that involved addition of treated wastewater to the baseline scenario. Note that median concentrations of chlorophyll *a* were above (albeit often slightly) the designated 'national bottom line' (the cutoff between *C* and *D*) of 12.0 µg/L for all scenarios (Table 16).

Table 31 **Median surface water concentrations of chlorophyll *a*, total phosphorus and total nitrogen for each 1–D scenario (Table 19) for the period 2007–2014, with corresponding Attribute States based on the National Policy Statement for Freshwater Management 2014.**

	Chlorophyll *a* (µg/L)		Total nitrogen (mg/L)		Total phosphorus (mg/L)	
Scenario	Median	Attribute State	Median	Attribute State	Median	Attribute State
1D_0	12.62	D	0.32	B	0.026	C
1D_1_Surface	12.61	D	0.33	B	0.026	C
1D_2a_Surface	12.47	D	0.33	B	0.026	C
1D_2b_Surface	12.53	D	0.33	B	0.026	C
1D_2c_Surface	12.47	D	0.33	B	0.026	C
1D_3a_Surface	12.56	D	0.33	B	0.026	C
1D_3b_Surface	12.58	D	0.33	B	0.026	C
1D_4_Surface	12.59	D	0.32	B	0.026	C
1D_5_Surface	12.55	D	0.32	B	0.026	C
1D_6a_Surface	12.80	D	0.33	B	0.027	C
1D_6b_Surface	12.57	D	0.33	B	0.026	C
1D_2c_Surface - DO	12.61	D	0.33	B	0.026	C
1D_3a_Surface - DO	12.56	D	0.33	B	0.026	C
1D_2c_Bed	12.55	D	0.33	B	0.026	C
1D_3a_Bed	12.66	D	0.33	B	0.026	C
1D_0 - LTS	12.52	D	0.31	B	0.026	C
1D_2c_Surface - LTS	12.49	D	0.32	B	0.026	C
1D_3a_Surface - LTS	12.52	D	0.32	B	0.027	C
1D_4_Surface - LTS	12.45	D	0.32	B	0.026	C
1D_5_Surface - LTS	12.51	D	0.32	B	0.026	C
1D_6a_Surface - LTS	12.80	D	0.32	B	0.027	C
1D_6b_Surface - LTS	12.49	D	0.32	B	0.026	C
1D_0 - Alum	15.85	D	0.43	B	0.060	D
1D_2c_Surface - Alum	16.10	D	0.45	B	0.059	D
1D_3a_Surface - Alum	15.91	D	0.45	B	0.059	D
1D_0 - LTS - Alum	15.57	D	0.43	B	0.059	D
1D_2c_Surface - LTS - Alum	15.77	D	0.44	B	0.059	D
1D_3a_Surface - LTS - Alum	15.79	D	0.44	B	0.060	D
1D_0 + 'pure' wastewater	12.50	D	0.32	B	0.026	C
Measured	14.95	D	0.34	B	0.022	C

Three–dimensional hydrodynamic modelling

Validation of simulated temperature at monitoring buoy

Summer

Simulated water temperatures for the summer period show periods of stratification lasting for durations of up to about one week, punctuated by intermittent mixing events (Figure 26). Such patterns are typical in Lake Rotorua during the summer months.

Near continuous temperature measurements were collected at the Lake Rotorua monitoring buoy at depths of 0.5 m and 12.5 m during the summer period. Measurements were also collected intermittently at deeper depths; comparisons were made between modelled and measured temperatures at 20.5 m, which is the deepest point measured.

Comparisons between modelled and measured temperatures show that the model reproduced the observed temperature structure of the lake very well during the summer period (Figure 27).

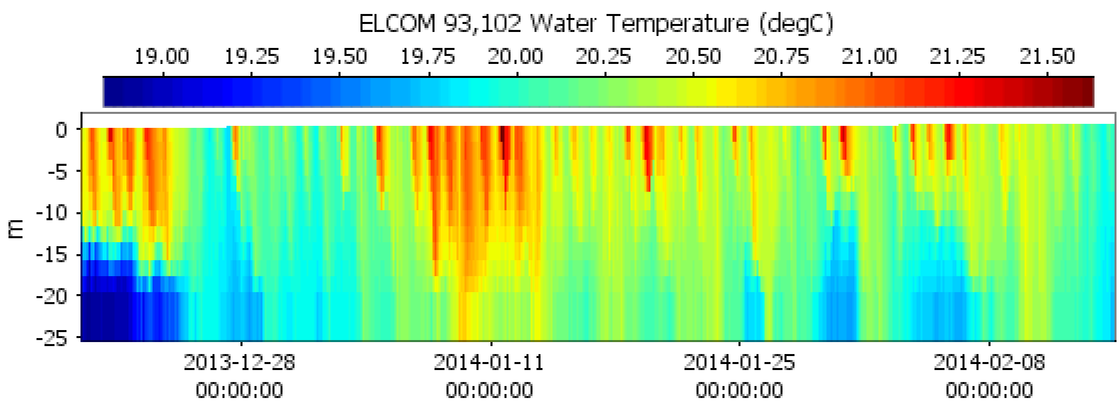

Figure 26 **Simulated water temperatures during the summer (2013/2014) modelling period.**

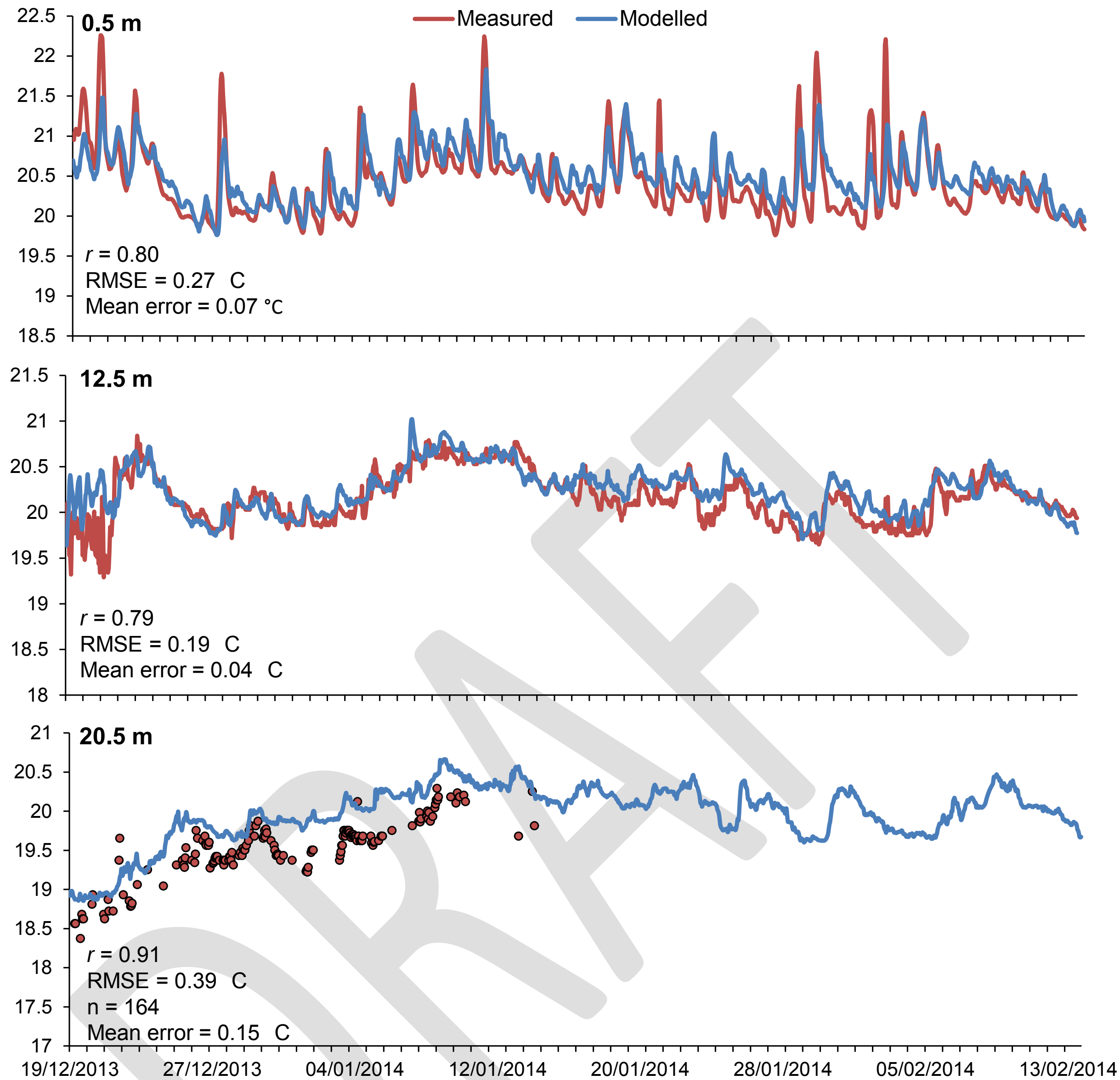

Figure 27 **Comparisons between modelled (3–D model) and measured temperatures for three depths at a central lake site during summer 2013/2014.**

Winter

Simulated water temperatures for the winter period show isothermal conditions, with a consistent decline in temperatures during the period as the winter season proceeds (Figure 28).

Near continuous temperature measurements were collected at the Lake Rotorua monitoring buoy at a range of depths during the winter period, although there is a gap of approximately one week in the data during mid–July. Comparisons between modelled and measured temperatures at three depths show that the model reproduced the gradual declining trend in observed temperatures very well (r = 0.91 to

0.95); however, the model consistently under–estimated the measured temperatures by an average of 1.5 °C to 1.7 °C (Figure 29). The reason for this discrepancy is uncertain, although it may reflect underestimation of heat retained by particles such as phytoplankton (note that ELCOM was run independently of the water quality model, CAEDYM). The implications of this for simulating basin–scale mixing processes (which are primarily driven by wind; Gibbs *et al., in prep*.) are considered to be minor.

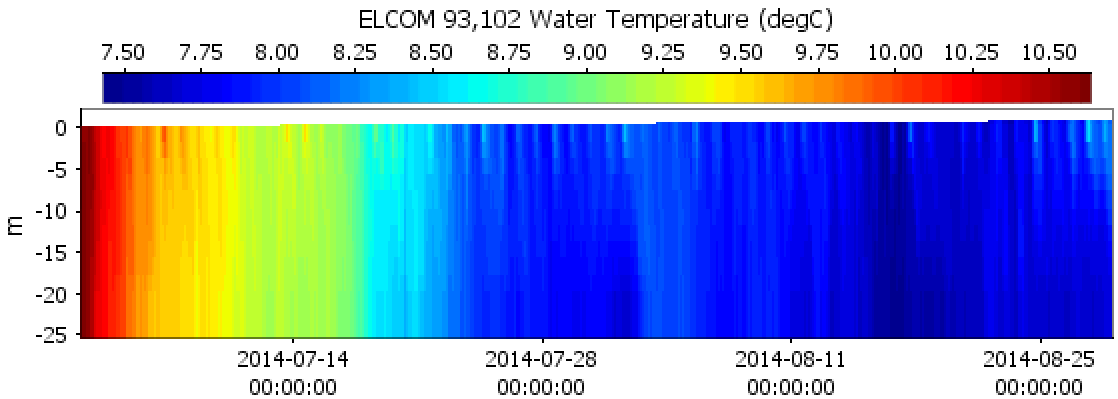

Figure 28 Simulated water temperatures during the winter (2014) modelling period.

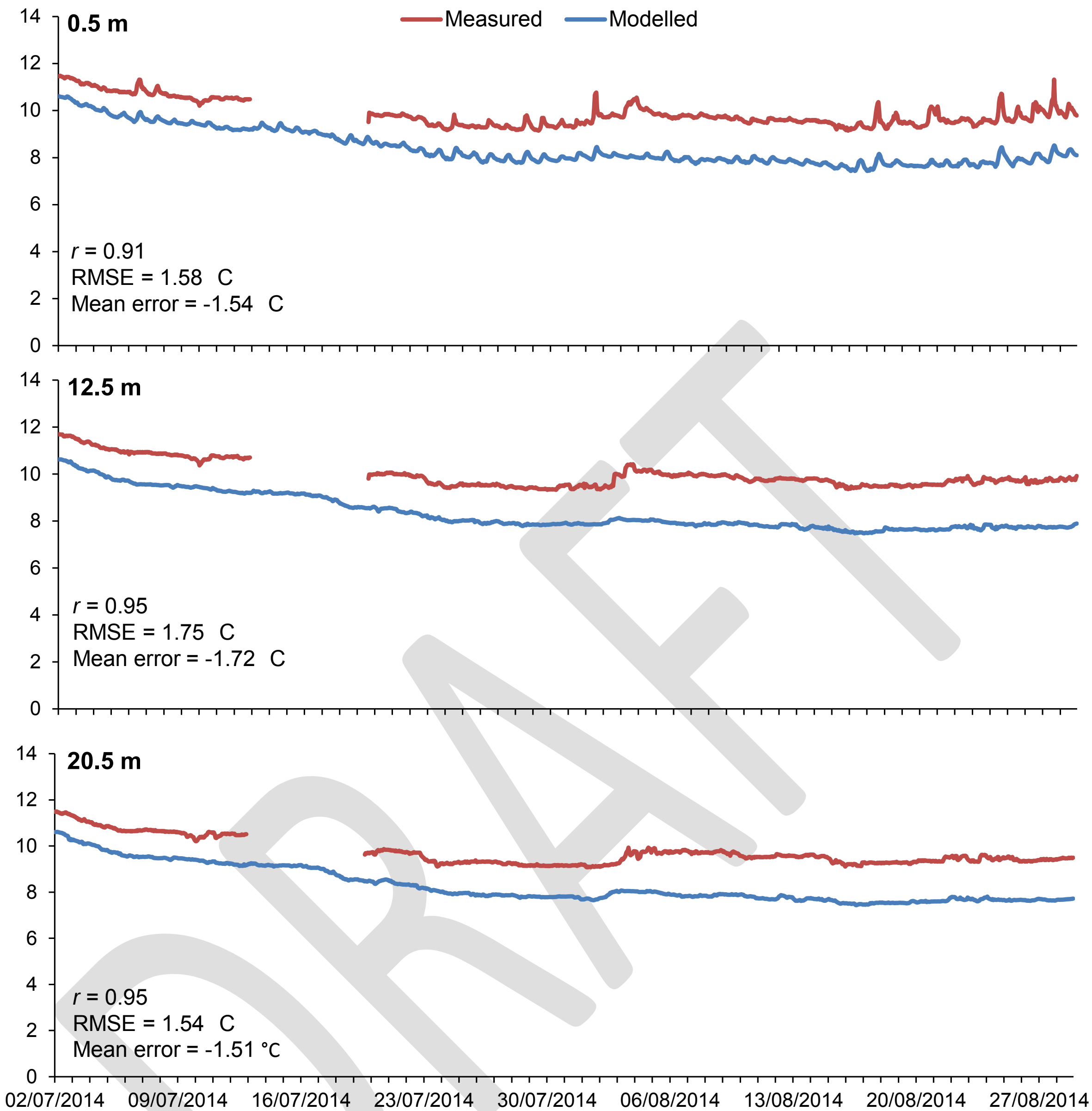

Figure 29 **Comparisons between modelled (3–D model) and measured temperatures for three depths at a central lake site during winter 2014.**

Simulated tracer concentrations

Effects of wind forcing

Simulations showed that basin–scale circulation processes can dominate mixing processes in the lake under certain wind forcing conditions. These circulation processes exerted a strong influence on transport (advection) of the simulated tracer.

Figure 30 illustrates the alternate circulation processes that become dominant following periods of consistent wind forcing from either the SW or the NE. Simulations show that such continuous winds set up a double gyre feature within the lake. Following SW winds, currents to the north of Sulphur Bay flow to the east, and then follow the shoreline northwards towards the Ōhau Channel (Figure 30a). This flow is reversed following SW winds, with currents following the shoreline southwards from the Ōhau Channel (Figure 30b). These currents then converge with a second gyre in the western basin of the lake, with subsequent northwards distribution of water to the north of Sulphur Bay.

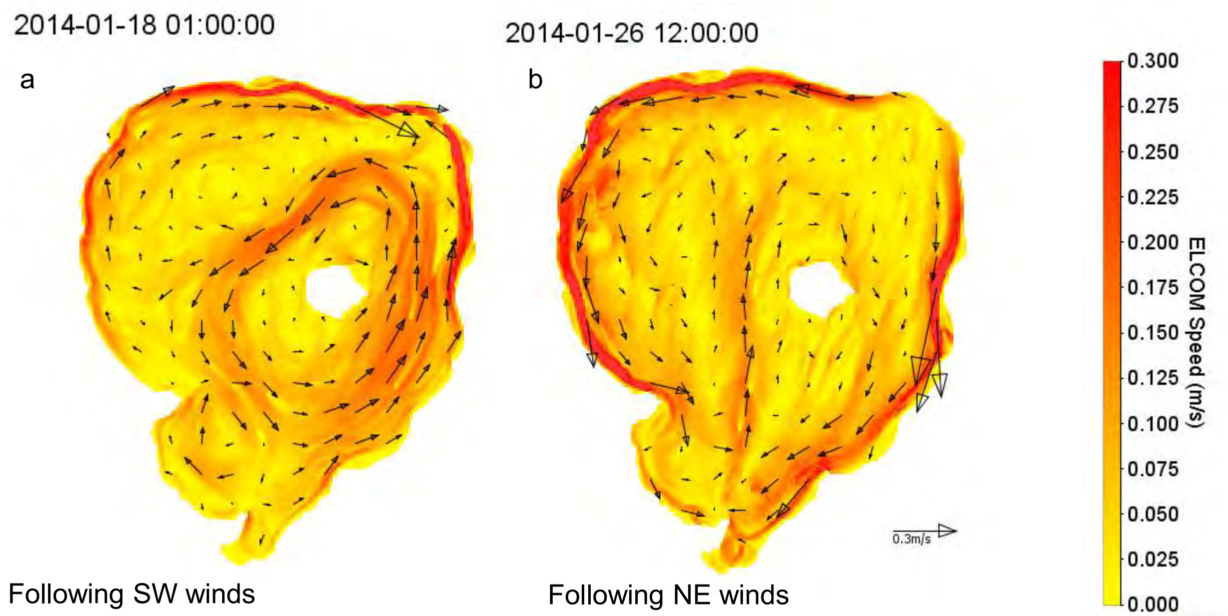

Figure 30 **Simulated water column average water speed and velocity vectors for two dates in summer 2014. *a*. 18 January, following a 72–hour period of continuous SW winds, with a mean hourly speed of 6.2 m/s (maximum = 12.2 m/s). *b*. 26 January, following a 48–hour period of continuous NE winds, with a mean hourly speed of 5.6 m/s (maximum = 8.3 m/s; Figure 6).**

The potential effect of these two circulation processes on treated wastewater dilution was investigated by simulating continuous wind forcing (4 m/s) from SW and NE directions. Simulated surface water concentrations of a conservative tracer added to the wastewater discharge were examined to understand how the inflow is dispersed throughout the lake. When examining simulated concentration data, it is important to consider that computational constraints meant that the lengths of the simulation period (~2 months) were considerably less than the mean hydraulic residence time of the lake (~1.5 years). This means that tracer concentrations are not at long–term equilibrium, and the mean concentration across the lake would therefore increase if the simulations were to run for longer. In addition, the conservative nature of the simulated tracer means that the concentrations are not reflective of the effects of attenuation process such as biological uptake and settling that may exert important controls on the distributions of analytes such as dissolved nutrients or microbes.

Simulated tracer concentrations that correspond to discharge to the Puarenga Stream, the lakeshore (Site 5) and the lake bed (Site 6) under scenarios of continuous NE or SW wind during summer are shown in Figure 31 (surface water concentrations) and Figure 32 (water column average concentrations). Plots show concentrations six weeks after the simulation started, and they are representative of the broad spatial patterns displayed throughout the simulations. Under a scenario of stream discharge with continuous NE wind, the simulated treated wastewater is predominantly transported westwards towards Rotorua city lakefront, and then dispersed northwards, following the northwards flow created at the gyre convergence shown in Figure 30b. Surface concentrations in the vicinity of Rotorua city lakefront under this scenario reach a maximum of ~0.1%, with water column average concentrations ~0.25%. Under a scenario of stream discharge with continuous SW wind, the simulated treated wastewater is predominantly transported along the eastern shoreline towards the Ōhau Channel, where surface concentrations are 0.15% to 0.20%. Surface and water column average concentrations in the vicinity of Rotorua city lakefront are very low (<0.1%) under this scenario.

The results for the scenario of discharge to the lake shoreline (Site 5) are very similar to those for discharge to the Puarenga Stream, reflecting the proximity of the two sites. Note, however, that there is no accumulation of tracer in Sulphur Bay for the SW wind forcing scenario, unlike the stream discharge simulation.

Under a scenario of lake bed discharge with continuous NE wind, simulated treated wastewater is initially transported northwards from the discharge location. It then becomes relatively well–dispersed throughout the lake, with low concentrations (~ ⩽0.1%) observed in near–shore areas. Under a scenario of lake bed discharge with continuous SW wind, simulated treated wastewater is predominantly transported southwards from the discharge location, towards Sulphur Bay and Rotorua City lakefront. As for the stream discharge scenario, treated wastewater is transported along the eastern shoreline towards the Ōhau Channel; however, there is generally greater dispersion and near–shore concentrations are approximately half those for the stream discharge scenario. The water column average concentrations are markedly higher than the surface water concentrations, reflecting the negative buoyancy of the treated wastewater (~18 °C; Figure 5) relative to ambient lake water (~20 °C; Figure 26). Maximum surface water concentrations for the lake bed discharge scenarios in Figure 31 are ~0.2%, compared with water column average maxima of ~30% in Figure 32. Bottom water concentrations (layer thickness = 2 m) are shown in Figure 33, which shows very high tracer concentrations (>70%) in the area immediately surrounding the discharge site. Under the NE wind scenario, maximum concentrations (~75%) are confined to an area of ~ 250 m × 250 m over the discharge site, with concentrations greatly diluted (< 3%) outside of this area (Figure 33). Under the SW wind scenario, maximum concentrations are higher (70–95%) but also confined to an area of ~ 250 m × 250 m. Concentrations of 10–20% extend approximately 1 km to both the north and south (Figure 33).

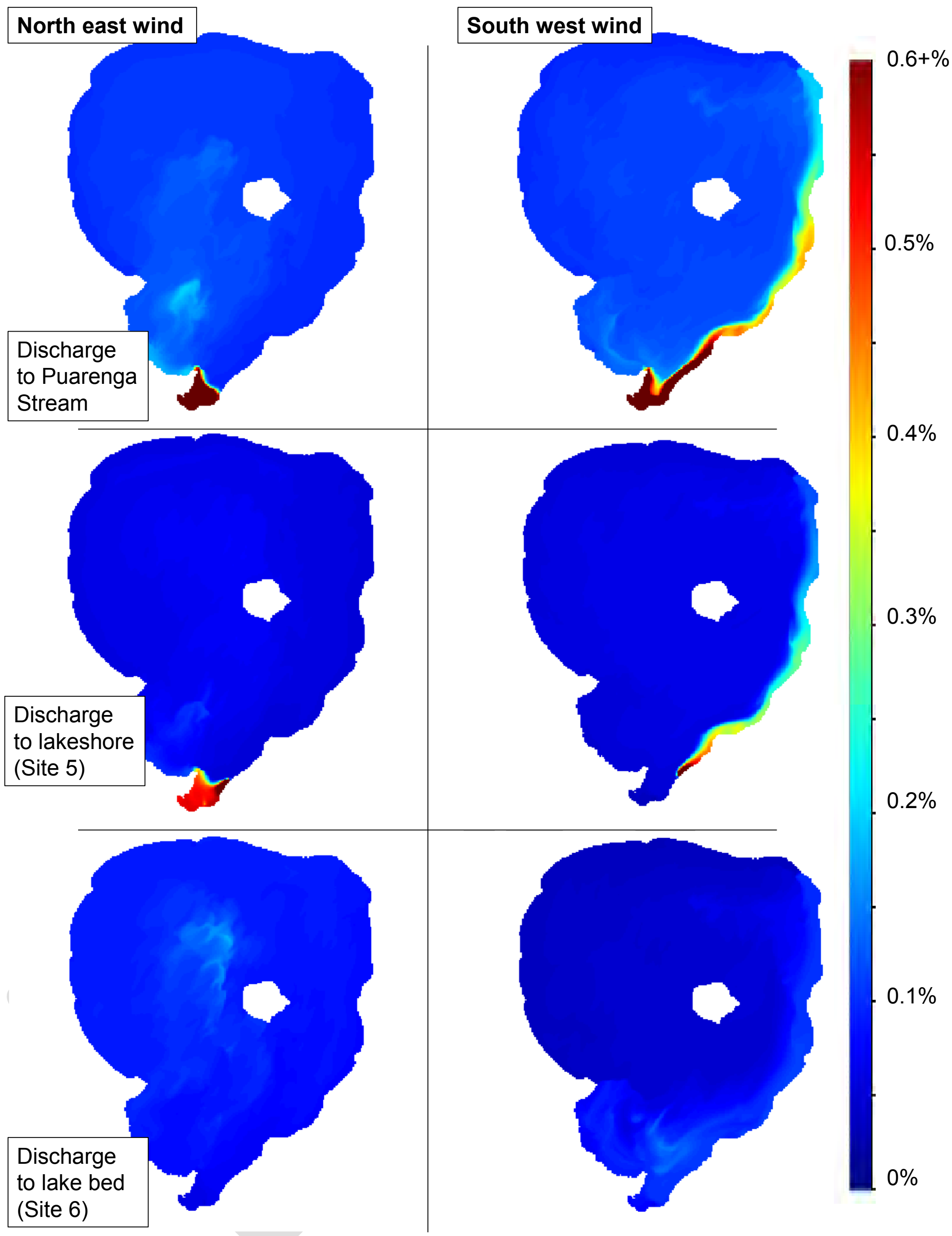

Figure 31 **Comparison of surface water (0–2 m) simulated tracer concentrations for scenarios of discharge to the Puarenga Stream, the lake shoreline (Site 5) and the lake bed (Site 6; Map 2) during summer 2013/2014 with consistent wind forcing (4 m/s) from the NE or SW. Plots show concentrations six weeks after the simulations started.**

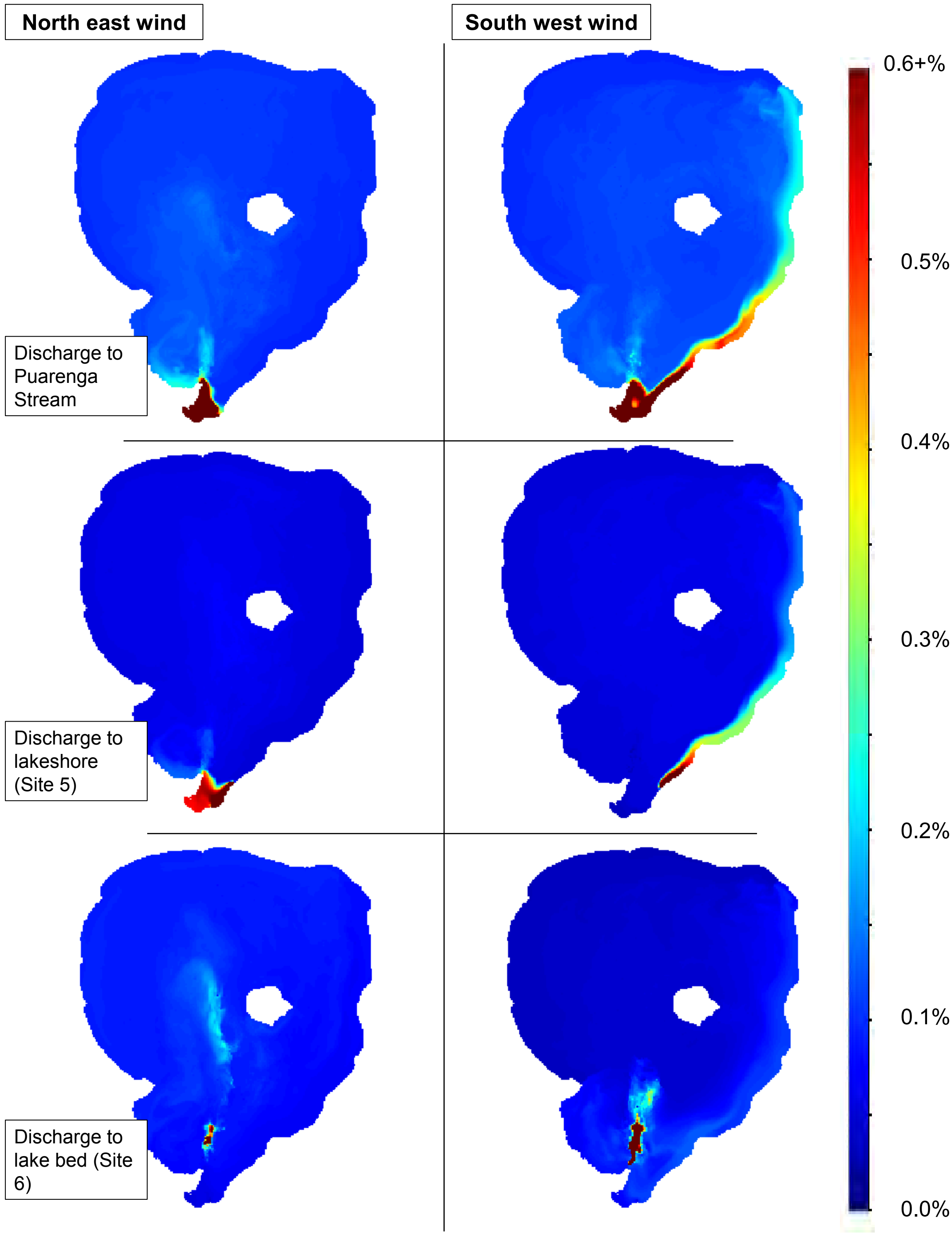

Figure 32 **Comparison of water column average simulated tracer concentrations for scenarios of discharge to the Puarenga Stream, the lake shoreline (Site 5) and the lake bed (Site 6; Map 2) during summer 2013/2014 with consistent wind forcing (4 m/s) from the NE or SW. Plots show concentrations six weeks after the simulations started.**

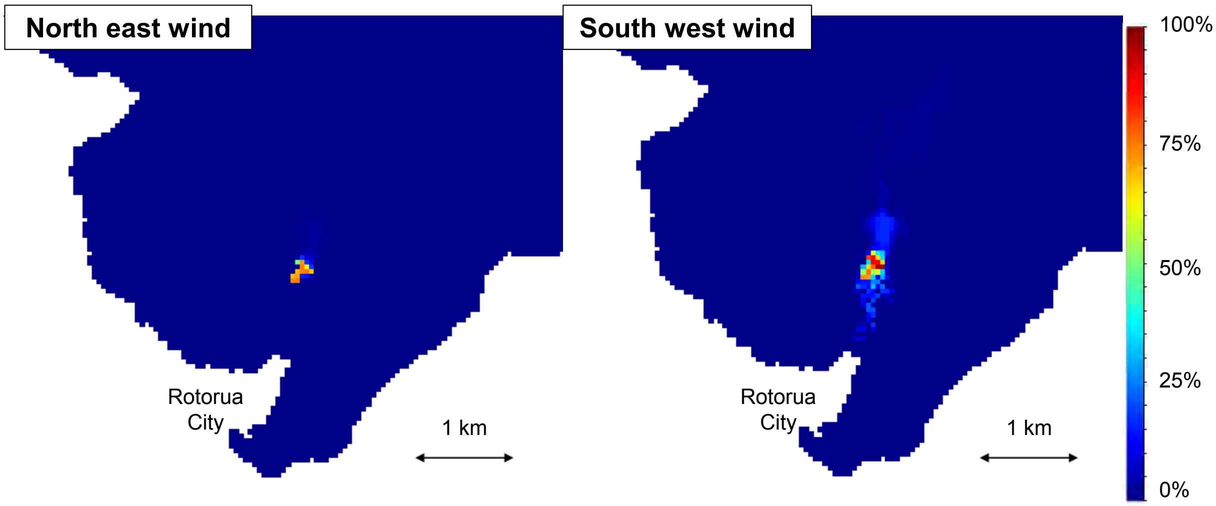

Figure 33 **Comparison of bottom water (2 m layer) tracer concentrations for a scenario of lake bed discharge (Site 6) during consistent (4 m/s) NE and SW winds. Plots show concentrations six weeks after the simulations started.**

Summer

Figure 34 compares simulated surface water (0–2 m) tracer concentrations for the summer period between scenarios involving discharge either to the Puarenga Stream (i.e., at Sites 1, 2 or 3), to a shoreline site (Site 5) or to the lake bed site (Site 6; Map 2). Precise concentrations for the dates that are presented are reported in Table 32. This table presents surface water (0–2 m) concentrations for three locations sited 100–150 m offshore, where depths are < 3 m. Surface water, bottom water and water column average concentrations are presented for Site 6 which is approximately 22 m deep.

In Figure 34, the spatial patterns in surface tracer concentration for the stream and lake shoreline discharge scenarios are very similar, reflecting the close proximity of the discharge locations. It is notable, however, that transport to the main body of the lake is higher under the scenario of stream discharge (e.g., compare the background concentrations on the final date that is shown between these two scenarios). This reflects slightly reduced retention of tracer in the lake for the lakeshore discharge scenario. Maximum concentrations observed for the scenario of lake bed discharge are generally lower than concentrations for the other two scenarios. In particular, surface concentrations in Sulphur Bay and along the eastern shoreline are lower for the scenario of lake bed discharge, compared with the other two scenarios for which concentrations are consistently ⩾ 0.3%. Note that simulated treated wastewater temperatures (Figure 5) were generally slightly lower than modelled surface water temperatures (Figure 27), resulting in a negatively buoyant discharge.

Figure 35 presents the results shown in Figure 34, except water column average tracer concentrations are presented rather than surface water concentrations. The patterns are consistent between the two figures, with the exception that concentrations are higher for the lake bed (Site 6) discharge scenario when water column average values are presented. Water column average concentrations at Site 6 are

6.8% to 11.0% on the dates shown. Concentrations in water adjacent to the lake bed at Site 6[12] were 70.7% to 86.7% on the dates shown in the figures (Table 32).

Winter

Figure 36 compares simulated tracer concentrations for the winter period between scenarios involving discharge either to the Puarenga Stream (i.e., at Sites 1, 2 or 3), to a shoreline site (Site 5) or to the lake bed site (Site 6; Map 2). Precise concentrations for the dates that are presented are reported in Table 33.

Relative to the summer period, there is more marked difference between the stream discharge and lake shoreline discharge scenarios. In particular, the plots for 16 July show transport of treated wastewater along both western and eastern shores for the stream discharge scenario (concentrations ≈ 0.6%), whereas transport is predominantly only along the eastern shore for the lake shoreline discharge scenario. This date occurred after a period of high (> 10 m/s) NE winds followed by a shift to moderate (~5 m/s) SW winds (Figure 7). Surface tracer concentrations for the lake bed discharge scenario were generally higher during the winter period than during the summer period; note that simulated treated wastewater temperatures (Figure 5) were generally higher than modelled surface water temperatures (Figure 28), resulting in a positively buoyant discharge. With lake bed discharge, treated wastewater was generally distributed throughout the lake to a greater extent than with discharge at either of the other two discharge locations. However, relatively high concentrations (~0.3%) were observed in near–shore areas on 27 August, following a period of ~6 days with SW winds. For this scenario, water column average and bottom water tracer concentrations were lower at Site 6 (i.e., the discharge site) compared with the summer period (Table 33). This reflects the positive buoyancy of the discharge and thus reduced accumulation of treated wastewater near the bed.

Figure 38 further illustrates this difference by presenting bottom water tracer concentrations for a scenario of lake bed discharge for dates during summer and winter, 2.5 months after the start of the simulation periods. The figure also illustrates the considerable dilution of treated wastewater that occurs over a relatively short distance (~500 m) from the discharge point. The gradients in tracer concentrations around the discharge site that are shown in this figure are representative of the summer and winter periods in general.

[12] I.e., in the 50 m (*x*) × 50 m (*y*) × 2 m (*z*) parcel of water above the simulated discharge point.

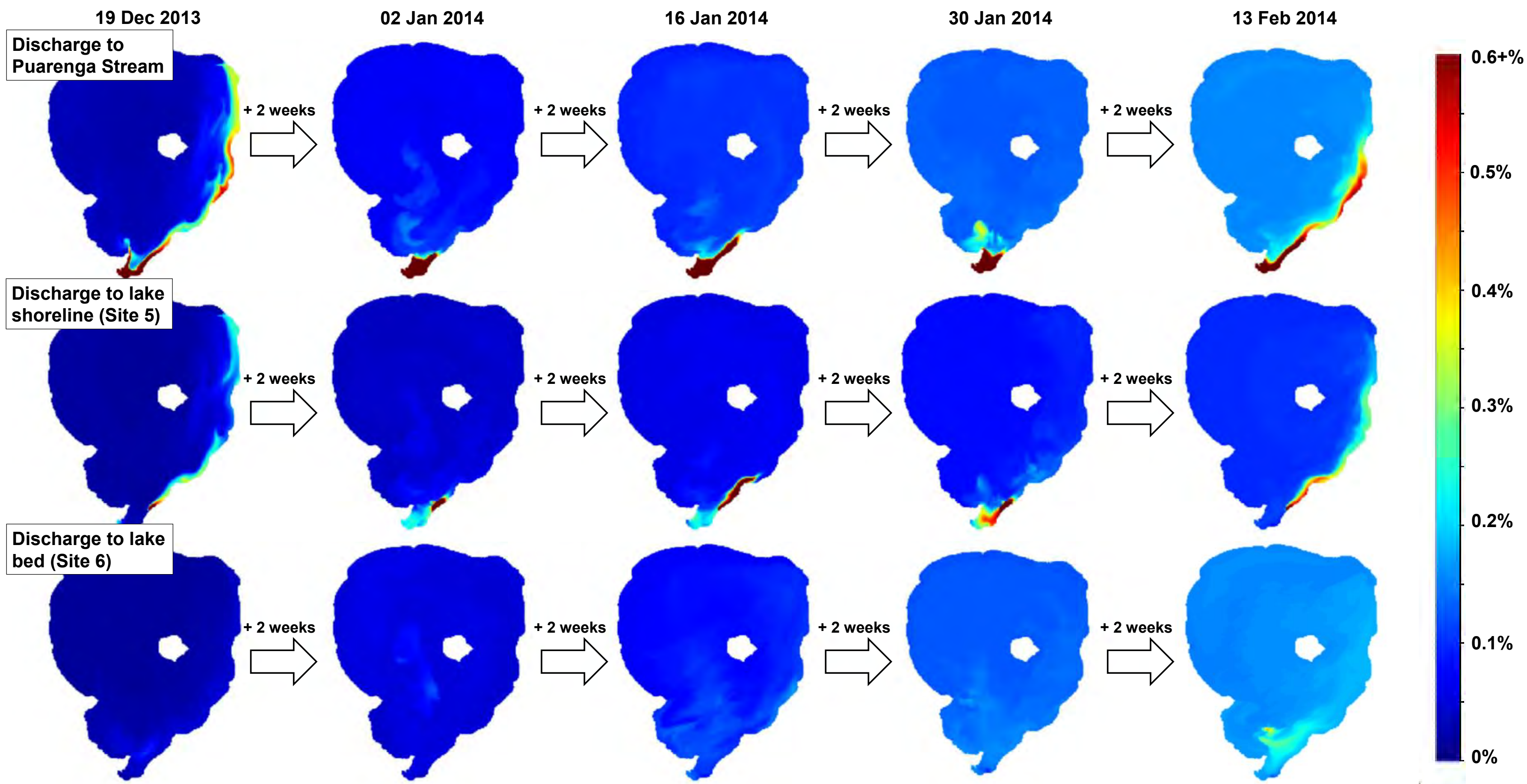

Figure 34 Comparison of simulated surface water (0–2 m) tracer concentrations for scenarios of discharge to the Puarenga Stream, Lake Rotorua shoreline (Site 5) and the lake bed (Site 6; Map 2) during summer 2013/2014. Plots are at two–week intervals, commencing two weeks after the simulation started.

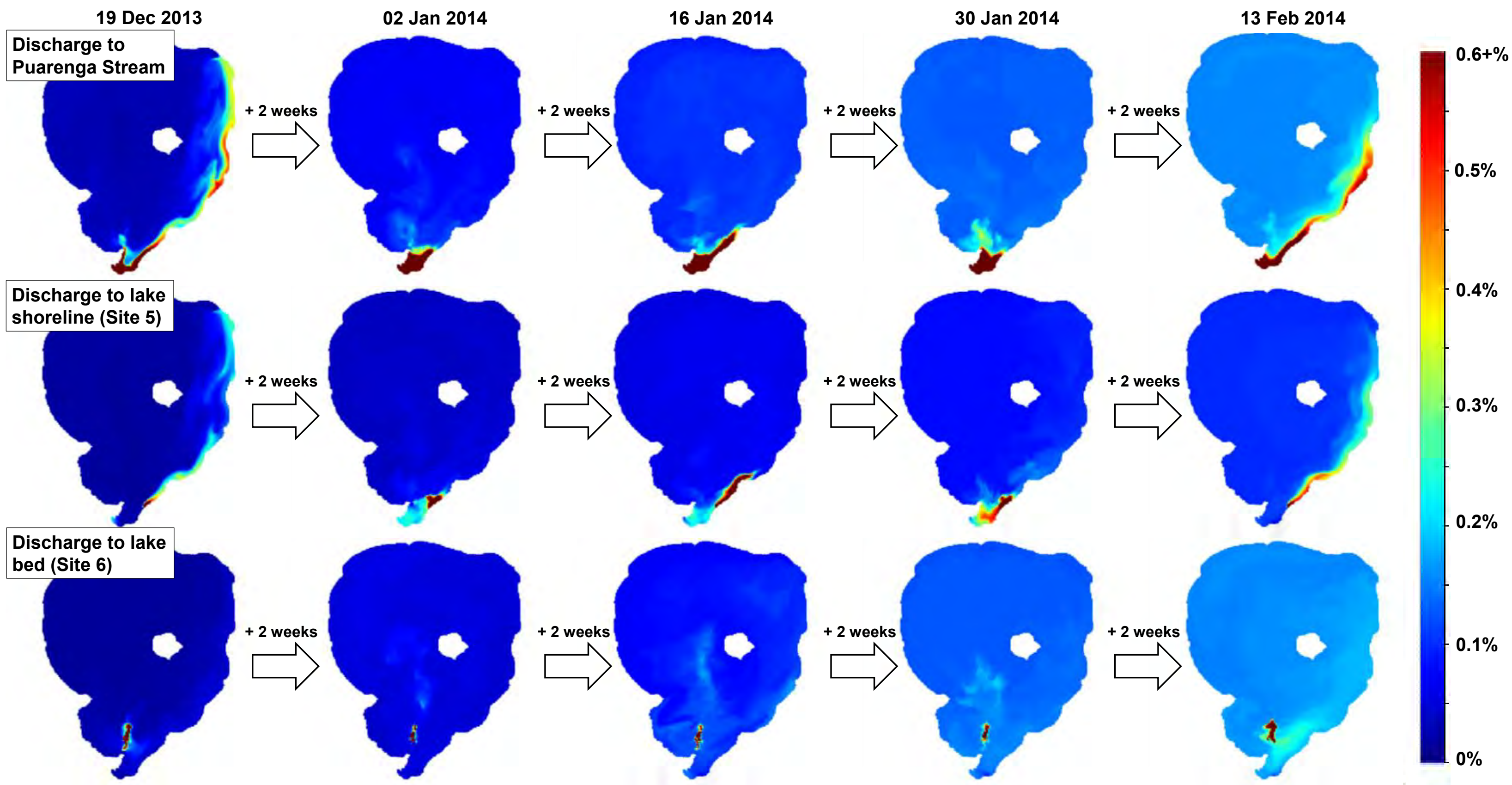

Figure 35 **Comparison of simulated water column average tracer concentrations for scenarios of discharge to the Puarenga Stream, Lake Rotorua shoreline (Site 5) and the lake bed (Site 6; Map 2) during summer 2013/2014. Plots are at two–week intervals, commencing two weeks after the simulation started.**

Table 32 **Modelled tracer concentrations (%) at four locations, on dates during summer 2013/2014 that correspond to plots shown in Figure 34 and Figure 35. Surface concentrations (0–2 m) are presented for three nearshore locations where depths are < 3 m. Surface, bottom water and water column average concentrations are shown for Site 6.**

Location	Discharge site	Depth	Date				
			19-Dec-13	02-Jan-14	16-Jan-14	30-Jan-14	13-Feb-14
Puarenga Stream mouth, 50–100 m offshore	Stream (Sites 1–3)	0–2 m	11.7	19.3	7.6	14.1	2.6
	Lake shoreline (Site 5)		< 0.1	0.2	0.1	0.3	0.1
	Lake bed (Site 6)		< 0.1	< 0.1	0.1	0.1	0.1
Site 5, 50–100 m offshore	Stream (Sites 1–3)	0–2 m	0.9	1.1	1.4	0.5	1.2
	Lake shoreline (Site 5)		2.1	3.4	1.4	3.4	1.4
	Lake bed (Site 6)		< 0.1	0.1	0.1	0.2	0.2
Rotorua City lakefront, 50–100 m offshore	Stream (Sites 1–3)	0–2 m	< 0.1	0.1	0.1	0.2	0.2
	Lake shoreline (Site 5)		< 0.1	< 0.1	0.1	0.1	0.1
	Lake bed (Site 6)		< 0.1	< 0.1	0.1	0.2	0.2
Site 6	Stream (Sites 1–3)	0–2 m	< 0.1	0.1	0.1	0.2	0.2
	Lake shoreline (Site 5)		< 0.1	0.1	0.1	0.1	0.1
	Lake bed (Site 6)		< 0.1	0.1	0.1	0.1	0.2
Site 6 (water column average)	Stream (Sites 1–3)	Water column average (0–22 m)	0.1	0.1	0.1	0.2	0.2
	Lake shoreline (Site 5)		< 0.1	0.1	0.1	0.1	0.1
	Lake bed (Site 6)		6.8	8.4	9.0	8.2	11.0
Site 6 (bottom water concentrations)	Lake bed (Site 6)	2 m layer adjacent to bed	70.8	80.0	85.8	72.0	86.7

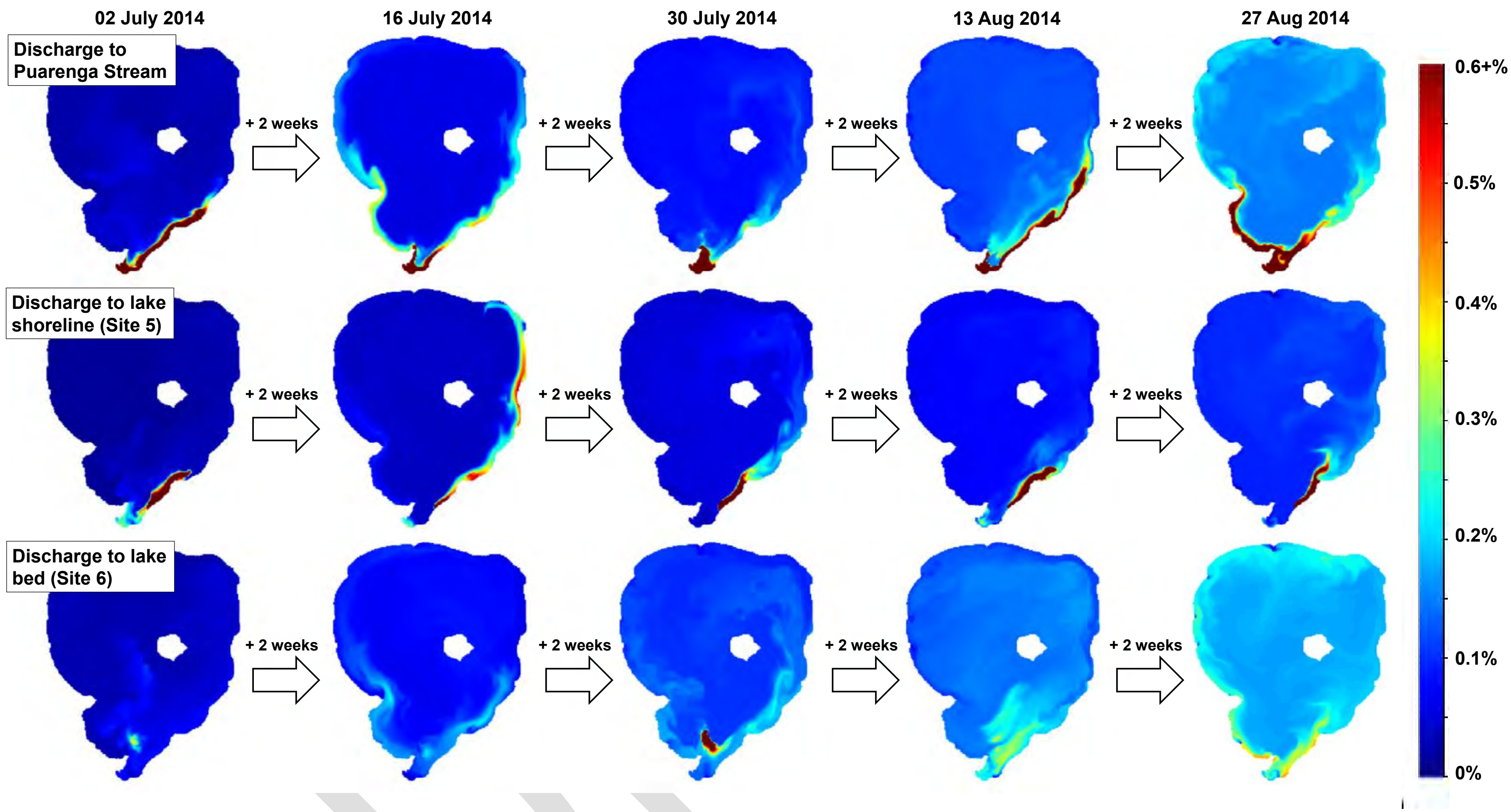

Figure 36 Comparison of simulated surface water (0–2 m) tracer concentrations for scenarios of discharge to the Puarenga Stream, Lake Rotorua shoreline (Site 5) and the lake bed (Site 6; Map 2) during winter 2014. Plots are at two–week intervals, commencing two weeks after the simulation started.

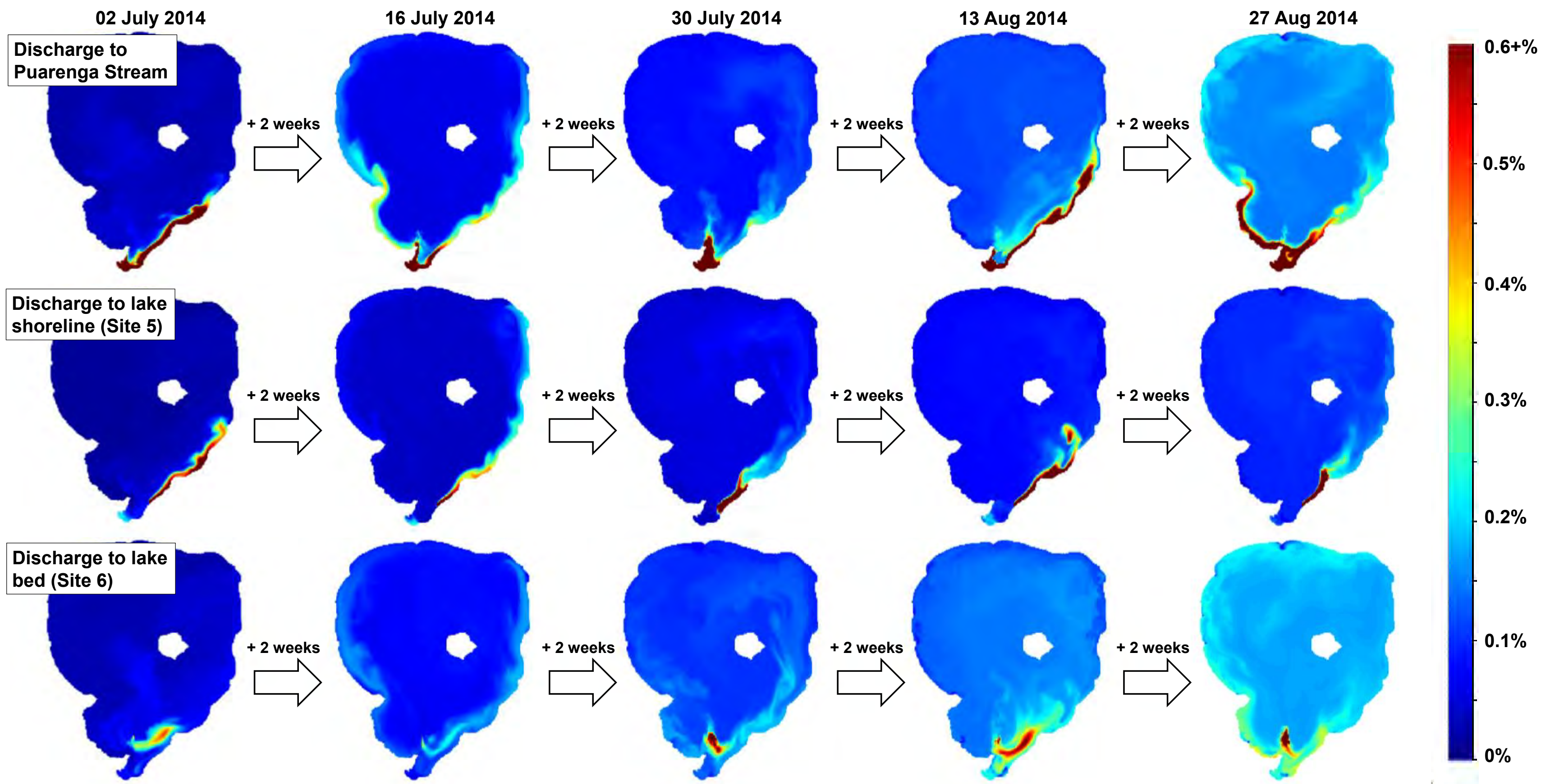

Figure 37 **Comparison of simulated water column average tracer concentrations for scenarios of discharge to the Puarenga Stream, Lake Rotorua shoreline (Site 5) and the lake bed (Site 6; Map 2) during winter 2014. Plots are at two–week intervals, commencing two–weeks after the simulation started.**

Table 33 Modelled tracer concentrations (%) at four locations, on dates during winter 2014 that correspond to plots shown in Figure 36 and Figure 37. Surface concentrations (0–2 m) are presented for three nearshore locations where depths are < 3 m. Surface, bottom water and water column average concentrations are shown for Site 6.

Location		Depth	Date				
			02-Jul-14	16-Jul-14	30-Jul-14	13-Aug-14	27-Aug-14
Puarenga Stream mouth, 50–100 m offshore	Stream (Sites 1–3)	0–2 m	4.6	8.9	9.1	17.2	23.8
	Lake shoreline (Site 5)		0.1	< 0.1	< 0.1	< 0.1	< 0.1
	Lake bed (Site 6)		0.1	0.1	0.1	0.1	0.2
Site 5, 50–100 m offshore	Stream (Sites 1–3)	0–2 m	2.0	0.7	0.2	1.4	0.7
	Lake shoreline (Site 5)		1.1	2.0	6.5	3.2	5.4
	Lake bed (Site 6)		0.1	0.1	0.2	0.3	0.3
Rotorua City lakefront, 50–100 m offshore	Stream (Sites 1–3)	0–2 m	0.0	0.3	0.1	0.1	0.9
	Lake shoreline (Site 5)		< 0.1	< 0.1	< 0.1	0.1	0.1
	Lake bed (Site 6)		0.0	0.1	0.2	0.1	0.3
Site 6	Stream (Sites 1–3)	0–2 m	0.1	0.1	0.1	0.1	0.1
	Lake shoreline (Site 5)		< 0.1	< 0.1	0.1	0.1	0.1
	Lake bed (Site 6)		0.1	0.1	2.3	0.2	0.2
Site 6 (average)	Stream (Sites 1–3)	Water column average (0–22 m)	0.1	0.1	0.2	0.2	0.2
	Lake shoreline (Site 5)		< 0.1	< 0.1	< 0.1	0.1	0.1
	Lake bed (Site 6)		1.8	1.1	3.4	2.4	5.7
Site 6 (bottom water concentrations)	Lake bed (Site 6)	2 m layer adjacent to bed	4.4	5.1	7.4	9.1	14.2

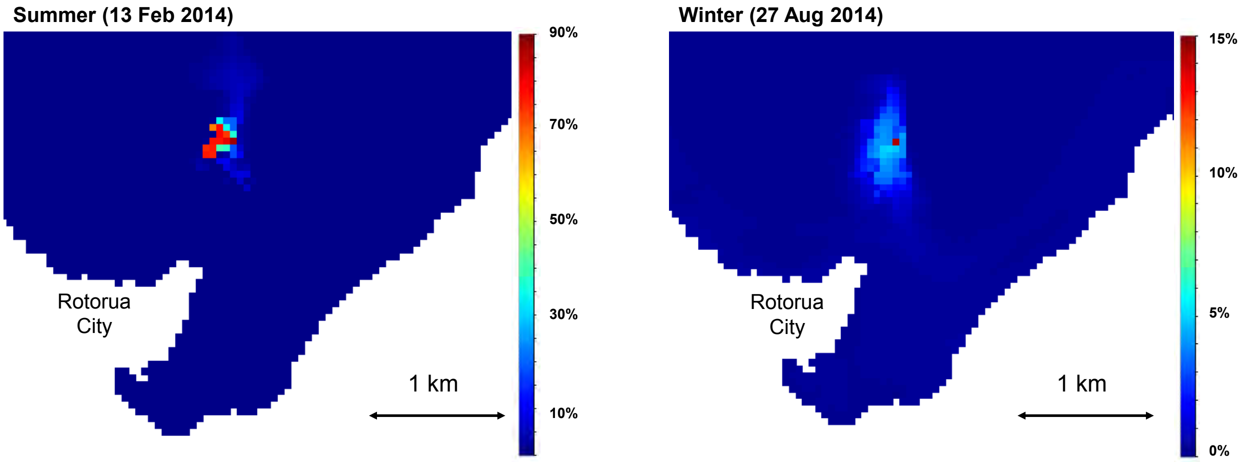

Figure 38 **Comparison of bottom water (2 m layer) tracer concentrations for a scenario of lake bed discharge (Site 6) during summer and winter. Plots show dates 2.5 months after simulations began and illustrate the differences between periods when the discharge is negatively buoyant (summer) and positively buoyant (winter) due to differences in temperatures between treated wastewater and ambient lake water. Note differences in the scales.**

Discussion

Effects on Puarenga Stream

Mass balance calculations undertaken in the context of guideline analyte concentrations in the National Policy Statement for Freshwater Management 2014 showed that proposed wastewater discharge to the stream would have either neutral (ammonium toxicity) or minor negative impacts (nitrate toxicity, dissolved oxygen). Semi–quantitative assessment based on summary statistics of projected wastewater composition (following membrane bioreactor treatment) showed that effects related to *E. coli* would be neutral or very minor (negative). Absence of data precluded a quantitative assessment regarding periphyton; however, qualitative assessment indicated that impacts on this aspect are expected to be neutral based on consideration of factors that currently limit periphyton growth in the lower Puarenga Stream (suitable substrate is limited).

The nitrate and ammonium results are based on the treated wastewater specifications that were provided for this assessment (Table 2), which are based on projections of constant concentrations. This analysis should be revisited if it becomes apparent during later design stages that variability in wastewater composition involving temporary higher concentrations is possible. Similarly, the assessment of *E. coli* concentrations was based on data provided that related to discharge following membrane bioreactor treatment. The mean value that was provided was very low (5.6/100 mL) and typically much less than the background concentrations in the stream, indicating that any associated impact would be neutral or very minor. However, significant negative impacts could occur as a consequence of short term discharge of concentrations that are much higher than the mean. In addition, the concentrations may not be representative of other treatment methods, and further analysis should be undertaken if options are considered that will result in higher concentrations. In addition, further consideration of microbial water quality could consider the characteristics of the *E. coli* and other coliform bacteria in both the stream and in the treated wastewater. Total *E. coli* counts provide only an indicator of public health risk, and techniques that quantify antibiotic resistance and virulence gene profiles (cf. Blaak *et al*. 2014) would support a more detailed assessment of this aspect (C. Dada, pers. comm. 2015).

The finding that the stream baseline Attribute State for ammonium toxicity (B) was higher than the baseline Attribute State for nitrate toxicity (A) likely predominantly reflects naturally elevated ammonium concentrations due to inputs from geological sources (cf. Morgenstern *et al*. 2015).

Effects on lake trophic status

Water quality modelling showed that treated wastewater addition is predicted to have very minor effects on lake trophic status. This result reflects the small to moderate contribution of each option to the overall external load to the lake (Figure 22; Figure 23), in addition to the high importance of internal nutrient cycling for controlling trophic status in the lake (Burger *et al*. 2007a). Overall, the performance of DYRESM–CAEDYM that was quantified during validation indicates that there is relatively low uncertainty in this general result. Model validation did indicate, however, that the model underestimated the extent to which annual TLI_3 varied in response to inter–annual differences in forcing conditions (Figure 21). This suggests that the increases in TLI_3 predicted for each scenario may have been slightly under–estimated, although the magnitude of any such error is expected to be low. With regard to differentiating between the treatment options, it is important to note that the relative differences between the TLI_3 predictions for each treatment option (0.02 to 0.03 units) are comparable to the mean error in annual TLI_3 predictions (Figure 21).

In a broader context of the RPSC decision–making process, the lack of marked difference between the six treatment options highlights the importance of carefully weighing the cultural and economic considerations (not considered in this study) associated with each option. For example, if large expenditure is required for relatively marginal improvements in wastewater treatment, then this may be more effectively invested elsewhere in the lake catchment to support lake water quality management, on the basis of $/t of nutrient load reduced. Similarly, cultural sensitivities of various disposal methods might be prioritised over small differences in final treated wastewater loads.

While the 1–D modelling showed that any impacts on long–term trophic status of the lake as a consequence of treated wastewater discharge would likely be very small, there is potential for more pronounced localised effects on productivity. These could include local increases in phytoplankton biomass in the vicinity of the outfall during periods when background nutrient concentrations in the lake are at limiting concentrations, e.g., during stratified periods in the summer. Such conditions could also occur some distance from the outfall, in areas where dominant mixing process cause the discharged treated wastewater to accumulate, e.g., potentially in the vicinity of Rotorua lakefront following prolonged NE winds and discharge to Sulphur Bay (Figure 31). In this regard, discharge to the Puarenga Stream has an advantage over discharge to the shoreline as the small proposed discharge rate of the treated wastewater compared with the stream means that dissolved nutrient concentrations (i.e., the most bioavailable fractions) will be greatly diluted, resulting in final concentrations that are only slightly above background conditions (e.g., Figure 12 and Figure 13). Remotely sensed data do show that chlorophyll *a* distributions in the Te Arawa lakes can be spatially heterogeneous at times, and high concentrations of chlorophyll *a* have been observed in Lake Rotoehu in the vicinity of geothermal inflows that have elevated nutrient concentrations (Allan 2011). In Lake Rotorua, such variations are more typically related to wind patterns that cause localised aggregations, although phytoplankton patchiness was observed in association with localised nutrient loading during a field study in early summer, at the mouth of the Ngongotahā Stream following rainfall (Abell and Hamilton 2015). The apparent localised 'blooms' that were observed were not readily visible to the eye, although dissolved nutrient concentrations in the stream[13] were less than the concentrations projected for the treatment options (Table 2).

Although it was not the primary focus of this study, the 1–D model results emphasised the significant positive contribution that stream alum dosing has had towards achieving TLI targets for the lake (cf. Hamilton *et al*. 2015). When alum representation was removed from model, the simulated values for the nitrogen and phosphorus functions (Figure 25) showed that both nitrogen and phosphorus limit

[13] DIN was ≈ 0.9 mg N/L, PO_4–P was ≈ 0.04 mg P/L

phytoplankton growth, although nitrogen limitation is typically more dominant. This is broadly consistent with observations, although the relative dominance of nitrogen limitation is perhaps slightly overestimated; experiments conducted in 2005 showed that phytoplankton was co–limited by nitrogen and phosphorus, with phytoplankton biomass responding to the greatest extent to phosphorus additions, but chlorophyll *a* concentrations responding more to nitrogen additions (Burger *et al*. 2007b). This result indicated a transition from dominance by nitrogen limitation (only) in the early 1980s (White *et al*. 1985). The shift to dominance of primary limitation by phosphorus when alum representation was included (i.e., the baseline scenario) reflects reduction of dissolved reactive phosphorus in the water column, both in the treated stream inflows, and in the lake due to downstream transport of free aluminium from the alum dosing stations (Hamilton *et al*. 2015). The major effect of alum dosing on lake water quality creates a challenge for lake ecosystem modelling, as the water quality model (CAEDYM) does not currently have the capacity to mechanistically simulate alum–related effects in a dynamic manner, without substantial model development work. Thus, the effects of alum were represented 'statically' by altering parameters that control the bed sediment phosphorus release rate and particulate organic matter settling (cf. Hamilton *et al*. 2015). Although these changes were based on mechanistic principles, there was a lack of representation of daily fluctuations in alum dosing rates and dynamic changes to sediment nutrient stores. This will have contributed to the model uncertainty, and was probably a major reason why error in modelled TLI_3 during this study was greater than for previous model applications that simulated periods prior to alum dosing commencing (Hamilton *et al*. 2012; 2015). This is also the likely reason why the model underestimated TLI_3 in 2007, and overestimated TLI_3 to a relatively large extent in 2011 and 2012 (Figure 21); 2007 was the first full year when alum dosing occurred (to one stream only), and it is likely that the free aluminium concentrations in the lake were too low to cause the full extent of in–lake effects that were represented in the model. Conversely, 2011 and 2012 were the years following initiation of dosing in a second inflow, when the total aluminium dose was the highest for the period (100–400 kg Al/d; see Figure 15 in Hamilton *et al*. 2015). Thus, it is likely that the in–lake effects of free aluminium were greater during those years than those represented in the model, and relative to the mean magnitude of those effects over the study period.

Dilution and dispersion of treated wastewater

Tracer concentrations predicted using the 3–D model (e.g., Figure 31) show the effects of advection and dispersion processes on the distribution of treated wastewater in the lake. Relative to the lake, the small volume of the proposed discharge means that the modelled concentrations in surface waters are typically low throughout most of the lake (e.g., < 0.2%), with higher concentrations (e.g., 0.5% to 1%) potentially present in near–shore areas, depending on discharge location and wind conditions (Figure 31; Figure 34). Results indicate that discharge to Sulphur Bay via the Puarenga Stream could lead to higher concentrations (up to ~20%) 100 m offshore of the stream mouth during certain periods (Table 33). Modelled discharge at the lake shoreline at Site 5 (the more northerly of the two shoreline sites; Map 2) resulted in greater dispersion around the outfall location, with concentrations ~1–7% 100 m offshore of the outfall. Discharge to the lake bed (Site 6) resulted in the lowest surface water concentrations (Figure 34; Figure 36), although bottom water concentrations could be very high in the vicinity of the discharge site during summer when the projected wastewater temperature was cooler than the lake (Figure 38). The small difference between maximum projected treated wastewater temperature (18 °C) and maximum lake water temperature (~21 °C) means that such accumulation in bottom waters would only occur during a ~2–3 month period, assuming that temperature exerts the dominant control on treated wastewater density.

The 3–D simulations highlighted the potential for wind–driven basin–scale circulation processes to greatly influence how treated wastewater mixes throughout the lake, depending on prior wind conditions and the location of the outfall. The simulations showed that wind forcing can establish alternate double gyre features that are predicted to cause transport of water added to the Puarenga Stream mostly either in a north–eastern direction along the eastern shoreline (SW winds), or northwards towards the main body of the lake, with potential for some accumulation in the vicinity of Rotorua city lakefront (NE wind), albeit resulting in concentrations in this area that are still very low (<1%). Such double gyre patterns have been observed in large lakes elsewhere (Beletsky *et al*. 1999), although single gyres are more typical (Emery and Csanady 1973; Csanady 1977). In the case of Lake Rotorua, Mokoia Island apparently acts as an axis around which a second gyre rotates (Gibbs *et al., in prep*.). It is important to note that model predictions of lake currents have not been validated in the vicinity of Sulphur Bay, and the model configuration did not include fine scale details of bathymetric characteristics in the embayment, nor detailed representation of the temperatures of geothermal inflows that are likely to influence mixing process. Therefore there is moderate uncertainty in the predicted basin–scale circulation patterns, and moderate to high uncertainty regarding predictions of localized mixing processes in Sulphur Bay. Field studies undertaken elsewhere in the lake do, however, support the model predictions in relation to the gyre features. Abell and Hamilton (2015) used high frequency water sampling to study the propagation of the Ngongohatā Stream in the lake following a rainstorm. Measurements were collected up to a distance of ~5 km from the stream mouth, and they showed excellent correspondence between observed mixing processes and those simulated with ELCOM–CAEDYM. In addition, Gibbs *et al.* (*in prep.*) validated ELCOM predictions using data collected using two acoustic Doppler current profilers (ADCPs) sited to the west and north of Mokoia Island. Modelled and measured current speeds and directions showed high correspondence, with the results also indicating that the double gyre pattern described above is highly–influential in controlling mixing processes in the lake at the basin scale. Further validation of mixing processes would require field studies in the vicinity of the proposed discharge locations. These could involve the deploying instruments such as ADCPs to measure current velocity, or drifter buoys to track currents. Alternatively, field tracer studies (cf., Gibbs *et al*. 2007) would be valuable to validate model predictions.

The non–equilibrium state of mean tracer concentrations in the lake (see Section 0) means than the modelled concentrations are underestimates with respect to analytes that are considered truly conservative, e.g., chloride. However, the predictions provide a good basis to examine how advection and dispersion affect the contribution to background lake concentrations of analytes in the treated wastewater that exhibit high loss rates in the lake. Such analytes typically include total coliform bacteria such as *E. coli*, which are commonly used as indicators of fecal pollution. Such organisms have a loss rate that combines losses due to base mortality, sunlight and settling (Chapra 2014). In a study of Lake Michigan, Liu *et al*. (2006) showed that sunlight exerted a stronger control than settling on the concentration of *E. coli*, with concentrations well–described using a model with an overall first–order inactivation coefficient in the range of 0.5 to 0.2 per day. With respect to total coliform bacteria, the modelled conservative tracer concentrations are therefore likely to represent overestimates of the likely extent of the distribution such organisms originating from treated wastewater, as the model predictions ignore attenuation processes. It should be stressed, however, that detailed microbial transport modelling was outside of the scope of this study, and the above discussion ignores the potential for resuspension of viable sediment–bound bacteria that can survive in lake sediments for several days (Laliberte and Grimes 1982).

Land Treatment System loads

A key uncertainty concerns the rate that nutrient loads from the LTS are expected to decline following its closure. The baseline scenario includes LTS loads in the Puarenga Stream, while the '-LTS' scenarios (Table 14) involve complete removal of the LTS loads; clearly, nutrient loads associated with the LTS are expected to be intermediate between these two conditions for a period after LTS operations cease. Furthermore, the response characteristics will be different for nitrogen and phosphorus due to the higher dominance of sub–surface transport in the case of nitrogen. Thus, the rate of decline in nitrogen loads will be highly dependent on catchment groundwater characteristics (which are not fully understood), whereas the rate of decline in phosphorus loads will be more closely related to processes of erosion and plant uptake in surface soil layers. Two specific processes that affect this uncertainty are described below. These may have the effect of reducing the short–medium term improvement in water quality relative to the model predictions; however, it is important to note that, regardless, the modelled improvement was extremely minor, comprising a difference in TLI_3 of only 0.02 between the baseline scenarios with and without LTS loads.

Regarding phosphorus loads, it should be noted that the configuration of the Puarenga Stream phosphorus concentrations in the '-LTS' scenario potentially overestimates the likely extent of the decline in load that would occur, even after any lag period has passed. Phosphorus concentrations in this scenario were reduced to achieve a 1.7 t P/y reduction in load, which relates to the 5–year 'sewage–derived' load estimated from LTS consent monitoring during 2007–2012 (A. Lowe, pers comm. 2013). This monitoring is conducted in the Waipa Stream, and inspection of water quality data collected by BoPRC in the lower Puarenga Stream suggests that the full extent of this load does not reach the lake. This indicates that removal processes in the stream attenuate the transport of LTS phosphorus loads to the lake. Such processes may include uptake by plants or adsorption to sediments, with the latter process known to exert a particularly important control on dissolved phosphorus concentrations in the stream (Abell and Hamilton 2013).

Of the two nutrients, understanding how nitrogen loads will be attenuated is of greatest importance, since losses are highest for this nutrient (Tomer *et al*. 2000). Knowledge of mean groundwater residence time is particularly pertinent to this understanding. The mean residence time of the Waipa Stream sub–catchment (where the LTS is sited) is reported as five years (Ray and Rutherford 2004, cited in Rutherford *et al*. 2009). This implies that the rate of decline in legacy nitrate in the stream will be relatively fast (i.e., years not decades), and is consistent with the observation that it took ~10 years for nitrate concentrations in the Puarenga Stream to reach a new quasi–equilibrium following the initiation of the LTS. However, recent work by Morgenstern *et al*. 2015 further highlights the complexity of local groundwater systems, and estimates that the mean residence time of the wider Puarenga Stream groundwater catchment is 40 years. Thus, the rate of decline in background nitrate loads could be slower than indicated by the estimated residence time for the Waipa Stream catchment if transport of nitrate has occurred beyond the shallow groundwater in this part of the catchment. Furthermore, experiences elsewhere highlight the potential for considerable lag times to occur in nutrient export following changes to management practices, and the rate of decline in nutrient export following the implementation of best management practices to manage nutrient pollution is typically slower than initial increases (Meals *et al*. 2009)[14]

Summary

Findings are summarised in Table 34 overleaf.

[14] Currently, a PhD research project is being undertaken by Ms W. Me at the University of Waikato to develop a detailed computer model of the catchment, and Ms Me's research has considerable potential to advance understanding of these issues.

Table 34 Summary of the environmental effects assessment

Focus of assessment	Environmental aspect	Methods	Results	Comments/uncertainties
Puarenga Stream impacts	Nitrate nitrogen toxicity		●Baseline corresponds to Attribute State A (high conservation value system). ●Option 3a results in no change. ●Options 1, 2, 4 and 5 result in median concentrations that correspond to low end of Attribute State B (some growth effect on up to 5% of species). ●Options 3b and 6 result in median concentrations that correspond to Attribute State A or the low end of Attribute State B.	Concentrations in the treated wastewater are assumed constant.
	Ammoniacal nitrogen toxicity	Mass balance calculations to consider efffects on the Puarenga Stream in the context of the National Policy Statement for Freshwater Management 2014. Based on baseline data for 2007–2014.	●Baseline corresponds to Attribute State B (some growth effect on up to 5% of species). ●The options considered resulted in no change.	Concentrations in the treated wastewater are assumed constant.
	Dissolved oxygen		●Baseline generally corresponds to Attribute State A (high conservation value system). ●A 'worst case scenario' involving discharge of anoxic wastewater would result in ~30% of monthly measurements corresponding to Attribute State B. This is consistent with "occasional minor stress on sensitive organisms".	Monitoring data (spot measurements) were assumed to be equal to daily minima. High frequency data were unavailable to test this assumption.
	E. coli		●Baseline conditions are variable but generally correspond to Attribute State B, which corresponds to a low (<1%) risk of infection to water users. ●Projected mean *E. coli* concentrations in the treated wastewater are very low and are predicted to have a neutral to very minor (negative) impact.	●Projections have only been defined for mebrane bioreactor treatment. These may not be relevant to all treatment options. ●Maximum projected concentrations have not been defined. If occassional high counts are expected to occur then the risk may be higher for temporary periods. ●The virulence of *E. coli* strains in the treated wastewater is unknown.
	Periphyton	Qualitative assessment	Options have the potential to increase periphyton abundance by causing minor increases in stream nutrient concentrations. However, the substrate and depth characteristics of the stream are poorly suited for periphyton growth hence effects are predicted to be neutral.	
Lake trophic status	TLI_3	1–D hydrodynamic–water quality modelling	●All proposed options are predicted to result in neutral or very minor (negative) impacts, comprising a eight–year mean increase in TLI_3 of ≤ 0.02 units. These changes are within the range of model error. ●All proposed options are not predicted to alter baseline Lake Ecosystem Health Attribute State values defined for total nitrogen, total phosphorus and chlorophyll *a* concentrations.	Model predictions under estimated the variability in measured TLI3, indicating that the model was not fully responsive to between year differences in nutrient loads. Nevertheless, there is high confidence in the general result that the options will have very minor effect on lake trophic status.
Treated wastewater dilution and dispersion	Treated wastewater distribution	3–D hydrodynamic mode	●Modelled surface water concentrations of treated wastewater (represented using a conservative tracer) are generally very low (<1%) througout the lake. ●Wind patterns drive basin–scale mixing processes that will affect how treated wastewater is dispersed in the lake. ●Discharge to Puarenga Stream (Sites 1, 2, 3) predicted to result in moderate accumulation (~3–30%) in Sulphur Bay near the stream mouth at times. ●SW winds predicted to cause accumulation of treated wastewater along eastern shoreline for scenarios of discharge to the Puarenga Stream and the lake shore (Sites 4 and 5); however, concentrations are still low (< 1%). ●Discharge to the lake bed (Site 6) generally predicted to result in lowest surface concentrations and greatest dispersion throughout the lake. Treated wastewater is predicted to accumulate in bottom waters (concentrations 70–90%) over a small area (~< 1 km^2) during the summer.	Model predictions of basin–scale mixing processes have not been fully validated; to do so would require an extensive field study. Nevertheless, there is high certainty in the general result that the treated wastewater will be highly diluted throughout most of the lake, including shoreline areas by Rotorua City lakefront.

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