Lake Nutrient Mass Balance Modelling

To Determine Nutrient Loads Required to Achieve Target Attribute States

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Climate, Freshwater & Ocean Science

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Executive summary

The Bay of Plenty Regional Council (BOPRC) requires catchment nutrient loads to be determined that will ensure that each of the 12 Rotorua Te Arawa lakes in the Region meets draft target attribute state options (TAS) for the NPS-FM attributes median total nitrogen, median total phosphorous and median and maximum chlorophyll *a*.

To aid with the BOPRC limit setting process, BOPRC requested NIWA to provide estimates of nutrient loads required to achieve TAS in the 12 largest Rotorua Te Arawa lakes, using the best available information in accordance with the NPS-FM. To better understand causes of change in lake water quality attributes and potential effects on the attributes by changes in nutrient loads it was necessary as well to examine seasonality in the magnitude of the attributes, magnitude of nutrient retention, differential nutrient retention (differences in retention between nitrogen and phosphorous and therefore in their ratios), internal loading (phosphorous release from the sediment), bottom water oxygen concentrations (relevant for magnitude of internal loading), removal of nitrogen by denitrification, and nutrient burial in the lake sediment.

To convert catchment load export to estimates of in-lake concentrations we compared six mass balance models for phosphorus and three for nitrogen to estimate the relationships between loading and in-lake concentrations. The models were set up to predict both nutrient concentrations from loads and vice versa. In addition, we analysed the BOPRC water quality database for each of the lakes to provide supporting information regarding average water quality, trends and seasonality, and to provide context to the mass balance modelling.

There were clear differences between the lakes in processing of nutrients and therefore in the relationships between nutrient loads and in-lake concentrations. There was a remarkable range of TN:TP ratios in the surface layer water among the lakes, which were not related to the nutrient ratios in the loads. In most lakes TN:TP ratios increased from the average in the inflow (i.e., the loads) to in-lake concentrations, suggesting greater retention of P than of N. In lakes Tarawera, Rotoiti, Rotoehu and Okataina TN:TP ratios decreased from inflow to in-lake concentrations. This may suggest relatively more denitrification and/or less P burial in these four lakes, compared with the other lakes.

Total phosphorus and chlorophyll concentrations were highest in winter in nine of the 12 lakes, excepting lakes Rotorua, Rotoehu and Rerewhakaaitu. The latter three lakes mix frequently down to the bottom, agreeing with their stratification potential being lowest of the 12 lakes, as defined by mean depth and lake area. Among the 12 lakes mean TP, TN and chlorophyll *a* in summer (January – March) decreased with increasing stratification potential. However, exceptions were lakes Okaro and Rotomahana which had both relatively high mean summer nutrient and chlorophyll *a* concentrations and high stratification potential. The lake stratification potential strongly correlated with the month of maximum chlorophyll *a* ($R^2 = 0.90$, $p < 0.000005$). Lakes where the chlorophyll *a* maximum falls in summer tend to have higher maximum algal biomass.

Apart from sediment contents, bottom water oxygen is an important driver of internal loading in most lakes. Anoxic bottom water enhances phosphorus release from the sediment. Bottom water becomes anoxic during late summer to autumn in monomictic lakes (lakes where the full water column mixes once a year, in winter, stratified during all of summer) Okareka, Okaro, Okataina and Rotoiti. Lake Tikitapu can become anoxic at its deepest point. In shallow polymictic lakes (lakes that mix repeatedly during the year, also during summer) during short periods of stratification in summer (December – March) bottom water can rapidly deoxygenate and become anoxic.

In shallow lakes Rerewhakaaitu, Rotoehu and Rotorua bottom water DO was lower in December – March than in April – May and was lowest in February (on average about 2-3 mg L^{-1} at near maximum depths). However, the monthly sampling (low frequency) may have missed more severe anoxic events in shallow polymictic lakes.

The gross internal load is P release from the sediments and can be measured, for instance by using benthic chamber incubations (Burger et al. 2007). The gross internal load does not account for sediment recycling, i.e., it is the total internal loading rate before deducting sedimentation rates, and therefore is a measure of internal loading that is expected to be related to the average standing crop of phytoplankton. We predicted gross internal load from sediment iron concentrations and previously-established empirical relationships. There was a good correlation between the predicted gross internal P load per unit area divided by mean depth, and annual mean chlorophyll (R^2 = 0.62). The correlation with annual mean chlorophyll improved by calculating the total P load (external + gross internal) per unit area divided by mean depth (R^2 = 0.76). The estimate for the gross internal P load in Lake Rotorua (4.1 g m^{-2} y^{-1}) was similar to the average observed internal P load in Lake Rotorua at 14 m depth (3.3 g m^{-2} y^{-1}, Burger et al. 2007).

The lowest rates of denitrification per unit area occurred in the least productive lakes, Lake Tikitapu (0.27 g N m^{-2} y^{-1}) and Lake Rotoma (0.01 g N m^{-2} y^{-1}), and the highest denitrification rates in Lake Rotorua (3.0 g N m^{-2} y^{-1}) and Lake Rotoiti (2.3 g N m^{-2} y^{-1}), while denitrification as a proportion of the N inputs ranged from 0.6%. to 71%.

As the proportion lost by denitrification increased, the ratio of N:P in the lake decreased relative to volume-weighted average in the inflows (mass balance results are based on long-term averages). When denitrification exceeds about 50% of the external N load, the TN:TP ratio in the lake becomes less than that in the external loads, suggesting a larger proportion of N is lost than of P from the surface waters in the lake, and accordingly the N:P ratio decreases relative to the loads.

In none of the Rotorua Te Arawa lakes did the export of N or P through the outlet exceed the external inputs. Therefore, there was a net retention of TN and TP in all 12 lakes by burial in the sediment and loss of N by denitrification. Retention averaged 71% of the P loads and 70% of the N loads. The change in the TN:TP ratio from the external loads to the lake concentrations was significantly related to nutrient retention of P. Also, the proportion of the gross internal P load to the external P load increased the more negative the difference between observed and predicted P retention became (R^2 = 0.72, $p < 0.0005$).

The relationships given by the mass balance models are a means by which to describe the average statistical behaviour of a large population of lakes, rather than the specific behaviour of a single lake. Nevertheless, model performance was unsatisfactory in many cases especially for P, in that predicted concentrations were substantially higher than observed concentrations in a large proportion of the Rotorua Te Arawa lakes for all models. Likewise, in nine out of 12 lakes the P load estimates of McBride et al. (2021) were outside the range of loads estimated by the six mass balance models for P. In addition, there were large differences in load estimates between models. Furthermore, it was unexpected that the only model developed from a New Zealand lakes data set (Abell et al. 2019) performed less well than the five other examined models for P, and was also not the best model for N. The difference between the P loads estimated by the six models based on 2016-2020 mean water quality and the loads of McBride et al. was greater (average 61%, range 53-86%) than the desired reductions in the lake P concentrations (average 17%, range 0 – 65%).

Therefore, instead of using modelling to find the catchment loads needed to achieve TAS in each lake, the best method to estimate the change in loads needed to achieve TAS is to simply apply the ratio of the desired TAS to the present concentration proportionally to the external loads. In other words, the external nutrient loads should reduce by the same percentage as the in-lake concentrations to achieve TAS.

A linear relation between water quality improvement and load reduction is reasonable because for individual lakes (when variables such as long-term average residence time are in fact constants) the model fits are all either linear or indistinguishable from linear (excepting the Abell et al. model which fitted least well). In lakes where internal loading is substantial it may take a decade after reduction of the external load before a new equilibrium is reached.

1 Introduction

The Bay of Plenty Regional Council (BOPRC) is required to determine catchment nutrient loads that ensure that each monitored lake in the region meets draft target attribute state (TAS) options (i.e., proposed targets for water quality variables) for relevant attributes in the National Policy Statement for Freshwater Management (NPS-FM 2020).

The final TAS set will be set by BOPRC after engagement with tangata whenua and the community. At a minimum, TAS are set above national bottom lines, and at or above a baseline state (calculated at 1 July 2017), but can be more conservative, depending on the long term vision and environmental outcomes sought by the community, and assessment of implications.

BOPRC requires estimation, through best practice methods, of in-lake concentrations for the Rotorua Te Arawa Lakes, resulting from nutrient loads exported from the surrounding catchments, that would achieve TAS. BOPRC has chosen to use catchment loads calculated in McBride et al. (2021), as this is the best available information, to represent the current loading state, and mass balance models of the Vollenweider type (Verburg et al. 2018) to convert catchment load export to estimates of in-lake concentrations.

The relevant NPS-FM lake attributes for this work are:

- Phytoplankton chlorophyll *a* (Chl-*a*) – annual median and annual maximum.
- Total Nitrogen (TN) – annual median.
- Total Phosphorus (TP) – annual median.

The 12 lakes of relevance for this work are in Table 1. This table includes the latest compilation of 'current state' as of 1 July 2020.

Table 1: List of lakes relevant to this work and the 'current state' as of 1 July 2020. Current state following NPS-FM. Table provided by BOPRC.

Lake	Sampling site number	TN median	TP median	Chl-*a* median/maximum
Lake Okareka	1	A	A	B/A
Lake Okaro	1	C	C	C/B
Lake Okataina	1	A	A	A/A
Lake Rerewhakaaitu	1	B	A	B/A
Lake Rotoehu	3	B	B	C/C
Lake Rotoiti	3	B	C	B/B
Lake Rotokakahi	Outflow	B	B	B/B
Lake Rotoma	1	A	A	A/A
Lake Rotomahana	2	B	B	B/A
Lake Rotorua	2	B	B	C/B
Lake Tarawera	5	A	A	A/A
Lake Tikitapu	1	B	A	B/A

To aid with the BOPRC limit setting process BOPRC requested NIWA to provide estimates of nutrient loads required to achieve TAS in the 12 largest Rotorua Te Arawa lakes (Figure 1), using mass balance models. In addition, BOPRC required an examination of the relationships between TAS for the three attributes and TLI values for each lake. TLI has been used historically by the BOPRC to assess lake trophic state, but it is not an NPS-FM attribute.

To better understand causes of change in lake water quality attributes and potential effects on the attributes by changes in nutrient loads it was necessary as well to examine seasonality in the magnitude of the attributes, magnitude of nutrient retention, differential nutrient retention (differences in retention between nitrogen and phosphorous and therefore in their ratios), bottom water oxygen concentrations (relevant for magnitude of internal loading), internal loading (phosphorous release from the sediment), removal of nitrogen by denitrification, and nutrient burial in the lake sediment. These subjects have not received much attention in most of the 12 Rotorua Te Arawa Lakes.

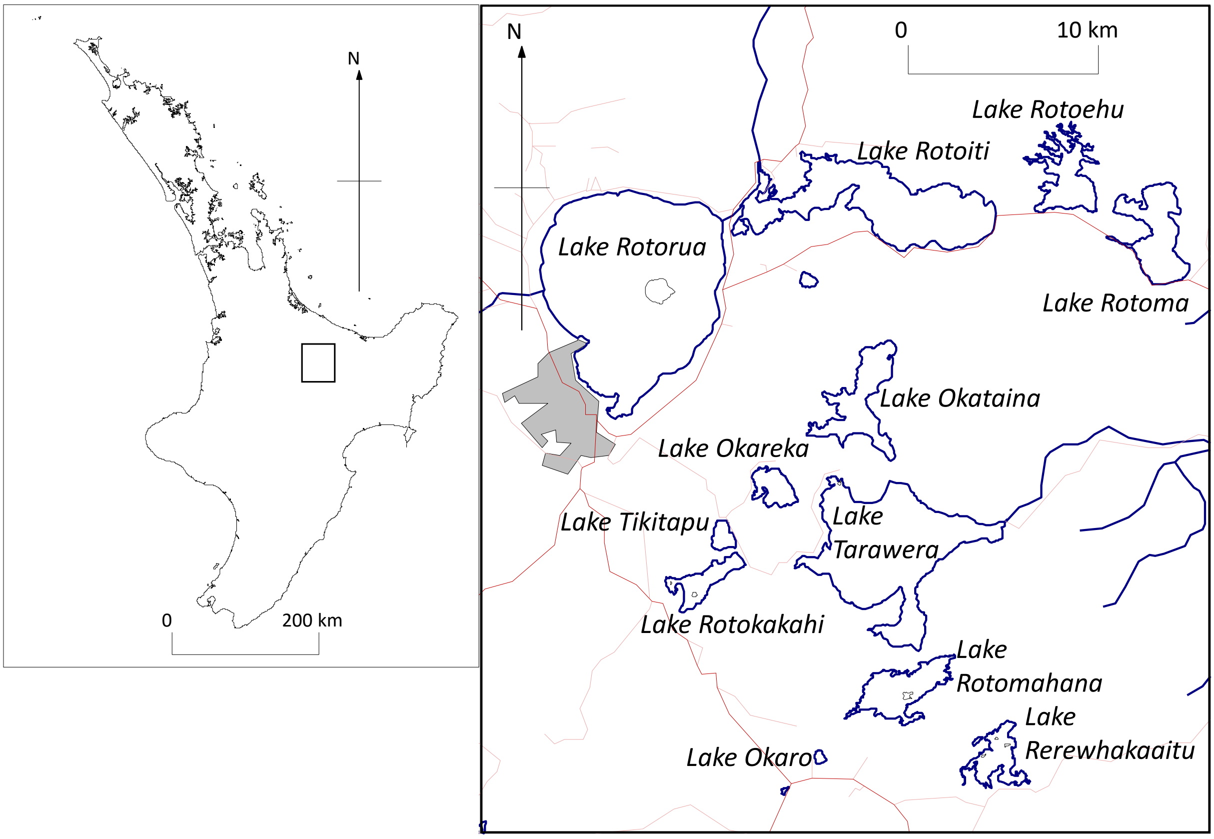
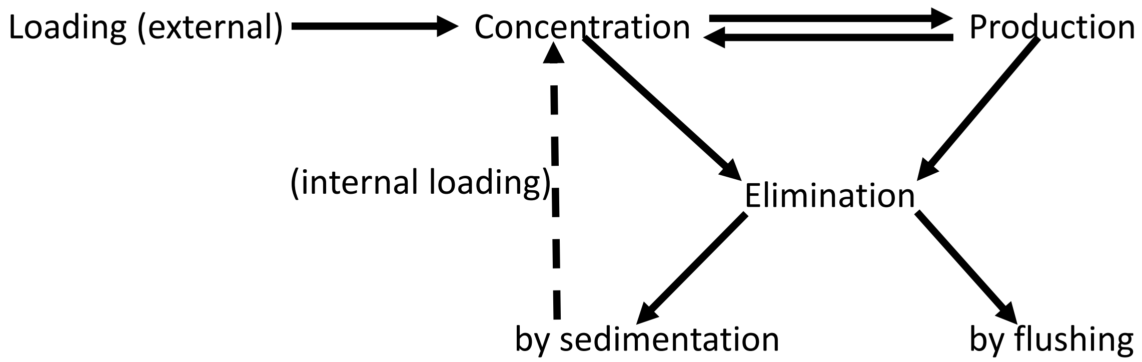


Figure 1: The positions of 12 lakes in the Bay of Plenty Region. The square in the left panel shows the location on the North Island.

2 Methods

2.1 Nutrient mass balance modelling

A simple conceptual mass balance model is given below (for P, for N add elimination by denitrification), adapted from OECD (1982).



To construct mass balance models for the relevant attributes TN and TP, we used the models of the Wisconsin Lake Modelling Suite (WiLMS 2003) for phosphorus, in addition to the model by Abell et al. (2019). Model variables are defined in Table 2. The models from WiLMS (2003) used to estimate the relationship between phosphorus loading and in-lake TP concentrations in the upper mixed layer were:

1) Dillon and Rigler (1974):

$$P = \frac{L_P(1-R)}{zp}$$

with predicted retention of phosphorus R (by sedimentation and burial) from Kirchner and Dillon (1975):

$$R = 0.426\ \mathrm{EXP}(-0.271\ q_S) + 0.574\ \mathrm{EXP}(-0.00949\ q_S)$$

Retention is the proportion of the external load that does not leave the lake in its outflow.

2) Larsen-Mercier (1976):

$$P = [TP]_{in}(1-R)$$

with predicted retention of phosphorus given by

$$R = \frac{1}{(1+\sqrt{p})}$$

Because $\sqrt{p} = \sqrt{\frac{1}{T_W}} = \frac{1}{\sqrt{T_W}}$ the Larsen-Mercier (1976) equation is equivalent to the

Vollenweider (1976) model:

$$P \quad = (1 - \mathrm{R})\frac{L_P}{q_S} = \frac{L_P/q_S}{\left(1 + \sqrt{T_W}\right)}$$

with predicted retention of phosphorus given by

$$R \quad = \frac{\sqrt{T_w}}{\left(1 + \sqrt{T_W}\right)}$$

3) Nurnberg (1984):

$$P \quad = (1 - \mathrm{R})\frac{L_P}{q_S}$$

with predicted retention of phosphorus given by

$$R \quad = \frac{15}{18 + q_S}$$

4) Reckhow (1979):

$$P \quad = \frac{L_P}{11.6 + 1.2 q_S}$$

The unit for P concentration (mg L^{-1}) in the equation of Reckhow (1979) and in WiLMS (2003) is incorrect, it is corrected here (mg m^{-3}).

5) OECD (1982):

$$TP \quad = 1.55 \left[\frac{[TP]_{in}}{\left(1 + \sqrt{T_W}\right)}\right]^{0.82}$$

In addition, we used the model of

6) Abell et al. (2019):

$$log_{10}(\mathrm{TP}) = \frac{log_{10}([TP]_{in})}{1 + (k_1 + \Delta k_1 d) T_W^{k_2}}$$

where k_1, Δk_1, and k_2 are fitted coefficients with values of 0, 0.44, and 0.13 respectively, and d = 0 for shallow lakes and d = 1 for deep lakes. A maximum depth threshold of = 7.5 m was used by Abell et al. (2019) to differentiate shallow lakes, which means all of the Rotorua Te Arawa lakes are in the deep lakes category.

Three models were used for the relationship between nitrogen loading and in-lake TN concentrations:

1) Abell et al. (2019):

$$log_{10}\, TN = 1.60 + 0.54\, log_{10}\, TN_{in} - 0.41\, log_{10} z_{MAX}$$

or in non-logarithmic form:

$$TN \quad = 39.8\, \frac{[TN]_{in}^{0.54}}{z_{MAX}^{0.41}}$$

2) A combination of the main mass balance equation from Vollenweider (1976) and predicted retention of nitrogen R from Harrison (2009):

$$TN \quad = (1 - R)\, \frac{L_N}{q_S}$$

and predicted retention of N is

$$R \quad = 1 - \exp\, (\frac{-a\, T_W}{z})$$

with a = 6.83 (for natural lakes, for reservoirs Harrison (2009) suggests a higher value). Retention of nitrogen covers both burial in the sediment and loss by denitrification.

3) OECD (1982):

$$TN \quad = 5.34 \left[\frac{[TN]_{in}}{\left(1 + \sqrt{T_W}\right)}\right]^{0.78}$$

The models predict lake-wide annual mean nutrient concentrations in the upper mixed layer and assume steady state conditions.

We also examined N loads based on the equation given by Jeppesen et al. (2005). It produced a linear relation with lake nutrient concentrations fitting closely to that of OECD (1982). However, the Jeppesen et al. model was developed for only 16 shallow lakes. Moreover, Jeppesen et al. (2005) cite Windolf et al. (1996) for the source of the model and its methods but the model in Windolf et al. (1996) differs in its coefficients from that given without further explanation by Jeppesen et al. (2005) and delivers results that are substantially different from the model as given by Jeppesen et al. (2005). Because the methods of the equation of Jeppesen et al. (2005) were not well recorded it was not used in this report.

Annual maximum chlorophyll concentrations were predicted using the equation in OECD (1982):

$$Chla_{MAX} = 0.74 \left[\frac{[TP]_{in}}{\left(1 + \sqrt{T_W}\right)}\right]^{0.89}$$

and compared with the same relationship in the Rotorua Te Arawa lakes data set.

Table 2: Model variables.

Symbols	Definition	Unit
$P = TP$	Predicted lake TP concentration	mg m^{-3}
$N = TN$	Predicted lake TN concentration	mg m^{-3}
L or L_P or L_{EXT}	Areal P load	mg m^{-2} lake area yr^{-1}
L_N	Areal N load	mg m^{-2} lake area yr^{-1}
z	Mean depth	m
z_{MAX}	Maximum depth	m
T_W	Lake residence time	year
p	Lake flushing rate	yr^{-1}
q_s	areal water loading rate	m y^{-1}
R	Fraction of the nutrient load that is retained in the lake	dimensionless
$[P]_{in}$ or $[TP]_{in}$	Average inflow TP concentration	mg m^{-3}
$[TN]_{in}$	Average inflow TN concentration	mg m^{-3}
O	Outflow rate	m^{-3} s^{-1}
A	Lake area	m^2

We used the estimates for lake area, lake volume, outflow rate, residence time, mean depth, and external N and P loads (which include aerial nutrient deposition onto the lake) provided for each of the 12 lakes by the McBride et al. (2021) modelling report (Table 1 and Table A in that report – Tables 4 and 6 below). McBride et al. (2019) estimated P and N loads using the areal export coefficient method, and outflow rates by using CLUES. P and N loads were estimated based on land use data of 2001-2004 but these were expected not to have changed much since then (McBride et al. 2021). Maximum depths were taken from Scholes and Hamill (2016). The areal water loading rate is calculated as $q_s = z/T_w = O/A$. The flushing rate is $p = 1/T_w$. T_w was calculated by McBride et al. (2021) as lake volume divided by outflow rate (V/O).

Observed lake specific retention coefficients for P and N were calculated, using equations in Verburg et al. (2018), and expressed as a proportion of the external nutrient load by:

$$R_{obs} = 1\text{-}\frac{O[P]_{\text{lake}}}{L_{\text{ext}}}$$

where $[P]_{lake}$ is the observed average total P concentration in the lake (mg m-3), and L_{ext} is the total annual external loading rate of P (mg y^{-1}).

Estimates of internal loading can be derived from the combination of catchment loading, and predicted and observed in-lake nutrient concentrations, using equations in Verburg et al. (2018), when mass balance models are derived from a lake data base that does not include lakes where anoxia and substantial internal loading occurs. Apart from screening the lakes database for lakes where anoxia occurs this is generally achieved by excluding lakes where nutrient concentrations in the lake are similar to or exceed those in the inflow.

2.2 Water quality measurements and TAS

The models were used to estimate load reductions required for total nitrogen (TN) and total phosphorus (TP) to meet TAS for median TN, median TP, median and maximum chlorophyll *a* (Chl-*a*) concentrations as defined by BOPRC.

Water quality data was provided by BOPRC for each lake, including monthly oxygen depth profiles. This data base was analysed to provide relationships between annual mean and median concentrations of TN, TP and chlorophyll *a*, to provide annual averages and seasonal patterns in each lake, and was examined for indicative information on trends (see Appendix). The analysed database included TP and DRP values from the new silica blocking method (for TP and DRP) with a retrospective adjustment for historical data (carried out by Keith Hamill). The analysed data, providing the phosphorus averages for 2016-2020 (frequently used in this report for comparisons with model outputs), therefore was corrected for interference by silica as were the phosphorus averages of July 2012- June 2017 (also referred to in this report) from the McBride et al. (2021) report.

Because the nutrient mass balance models produce annual mean and not median concentrations, to find loads required to achieve TAS the model results were converted to estimates of medians by using relationships between medians and means of TN and TP. The water quality data collected more or less monthly by BOPRC since 2001 in each of the 12 lakes were used to derive relationships between annual mean and median nutrient concentrations. In addition, because the models predict relations between nutrient loads and in-lake nutrient concentrations and do not predict in-lake concentrations of chlorophyll *a*, the BOPRC lake water quality database was used to derive the relationships between annual mean nutrient and annual median chlorophyll *a* concentrations. Maximum annual chlorophyll *a* concentrations, also not predicted by mass balance models, was estimated by a relationship with P concentrations given by OECD (1982).

Annual means, medians and maxima of TN, TP and chlorophyll *a* in the surface mixed layer (labelled "TOP" in the BOPRC database) were calculated separately for each lake and site within lakes (20 years and 15 sites). Multiple samples per month (sometimes multiple samples per day) were averaged before processing. Years with fewer than 10 monthly measurements were excluded (averages per lake: 18.8 years of data for TN and TP and 18.7 years for chlorophyll *a*). Medians were on average lower than means of TP (6.3% ±12.0% standard deviation), TN (2.8% ±6.5%) and chlorophyll *a* (12.4% ±14.5%). While regression analysis (Figure 2) illustrates the close similarity of means and medians, the differences were nevertheless highly significant (t-Test: Paired Two Sample for Means).

Relationships between annual means and medians of TN (R^2 = 0.98), TP (R^2 = 0.97) and chlorophyll *a* concentrations (R^2 = 0.95) in the BOPRC database of 2001-2020 were similar to those in a national lake database (Verburg et al. 2010, data of 2005-2009 in 119 lakes monitored by regional councils, 113 for chlorophyll *a* concentrations) for TN (R^2 = 0.99), TP (R^2 = 0.95) and chlorophyll *a* (R^2 = 0.96, Figure 2). Relationships are fitted with power equations. R^2 values were lower for linear regressions, although in some cases there was little difference. Relationships between various measures of chlorophyll (annual means, medians and maxima) and nutrients are shown in Figure 3. Regression equations given in the figures were used to convert the median TAS values to annual means before application in the nutrient mass balance models.

R^2 values for annual mean (R^2 = 0.71), median (R^2 = 0.64) and maximum chlorophyll (R^2 = 0.68) against annual mean TN (not shown) were similar as against annual mean TP (R^2 = 0.71, 0.66 and 0.67, respectively). One reason for the variance in the TP – chlorophyll relationship is related to the fact that the ratio chlorophyll:TP varies substantially, for instance about four-fold between the averages of the Rotorua Te Arawa lakes (from the lowest in Lake Tarawera to the highest in Lake Rotorua, the ratio was not strongly correlated with trophic state, i.e., the average TP or chlorophyll concentration). While in part this may be explained by non-linear relationships, presumably a more important reason for part of the variability are differences in composition of the phytoplankton species assemblages, with differences in chlorophyll content relative to nutrient requirements between species. However, applying separate annual mean TP – chlorophyll relationships for each lake is not an option because the relationships were very weak in several lakes (Okareka, Tarawera, Tikitapu: $R^2 < 0.01$, Rotomahana, Rotoma: $R^2 < 0.1$). The fits were not related to average TN:TP ratios, suggesting the weak fits in these lakes were not the result of N limitation being dominant.

Annual mean and median chlorophyll fitted equally well with annual mean Secchi depth (R^2 = 0.84, not shown) and better than TP (R^2 = 0.66).

We estimated the trophic level index (TLI) at the TAS for chlorophyll *a*, TN and TP, and compared with the average TLI in recent years (2016-2020), for each lake, using equations for the TLI components as in Burns et al. (2009).

BOPRC has not set TAS yet, and instead has identified the range of potential TAS that could be set (after community engagement and analysis of implications), that would be above national bottom lines and at or above baseline state. These are shown in Table 3. Note that the minimum TAS value, as defined by BOPRC, is the concentration value that will require the least nutrient load reduction. In other words, the minimum TAS is higher than (or in some cases equal to) the maximum TAS.

Table 3: TAS values provided by BOPRC. Target Attribute States proposed by BOPRC for annual median TP, median TN, median chlorophyll *a* and annual maximum chlorophyll *a* in Rotorua Te Arawa Lakes at monitored sites. min = minimum, max = maximum. Units are mg m^{-3}.

Lake	Median TP min	Median TP max	Median TN min	Median TN max	Median Chl-*a* min	Median Chl-*a* max	Maximum Chl-*a* min	Maximum Chl-*a* max
Okareka	9	8	191	160	4	2	8	8
Okaro	30	10	653	160	11	2	60	10
Okataina	7	7	90	90	2	2	7	4
Rerewhakaaitu	9	8	300	300	4	2	8	4
Rotoehu	22	10	460	300	9	2	44	10
Rotoiti	26	10	204	160	6	2	12	8
Rotokakahi	16	10	245	160	3	2	15	7
Rotoma	3	3	100	100	1	1	1	1
Rotomahana	28	10	190	160	3	2	7	5
Rotorua	14	10	315	300	9	2	12	10
Tarawera	13	9	100	84	1	1	3	3
Tikitapu	4	4	170	160	1	1	3	3

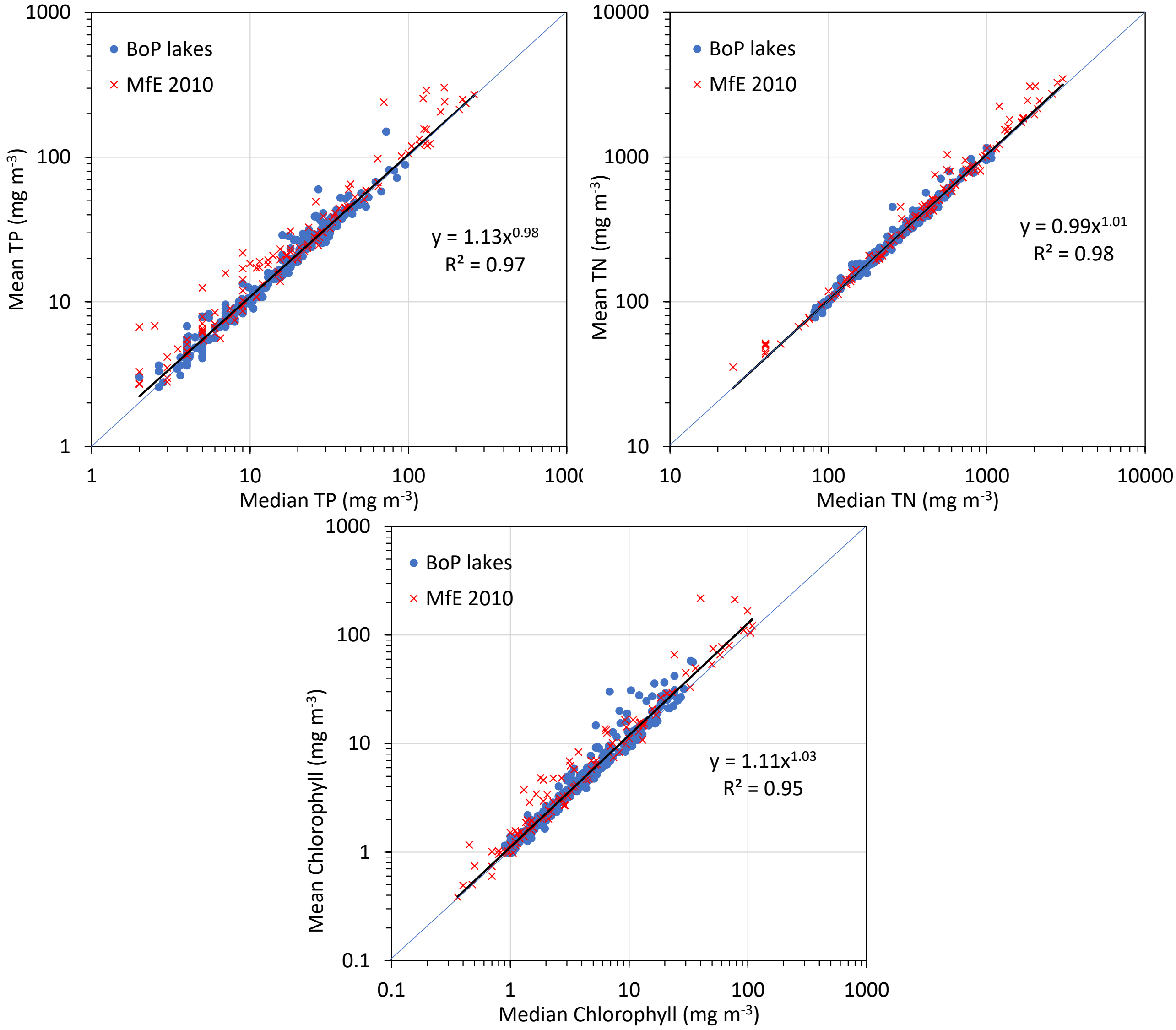


Figure 2: Relationships between annual median (as in TAS) and annual mean values of TP, TN and chlorophyll a. Shown are the annual data for 2001-2020 for all 12 lakes and the data of the MfE database (5-year averages) for 119 lakes (Verburg et al. 2010). The regression line and its equation are for the BoP lakes, used to convert TAS values to annual means. n is 279-282 (15 lake sites, up to 20 years). The light blue line is the 1:1 line.

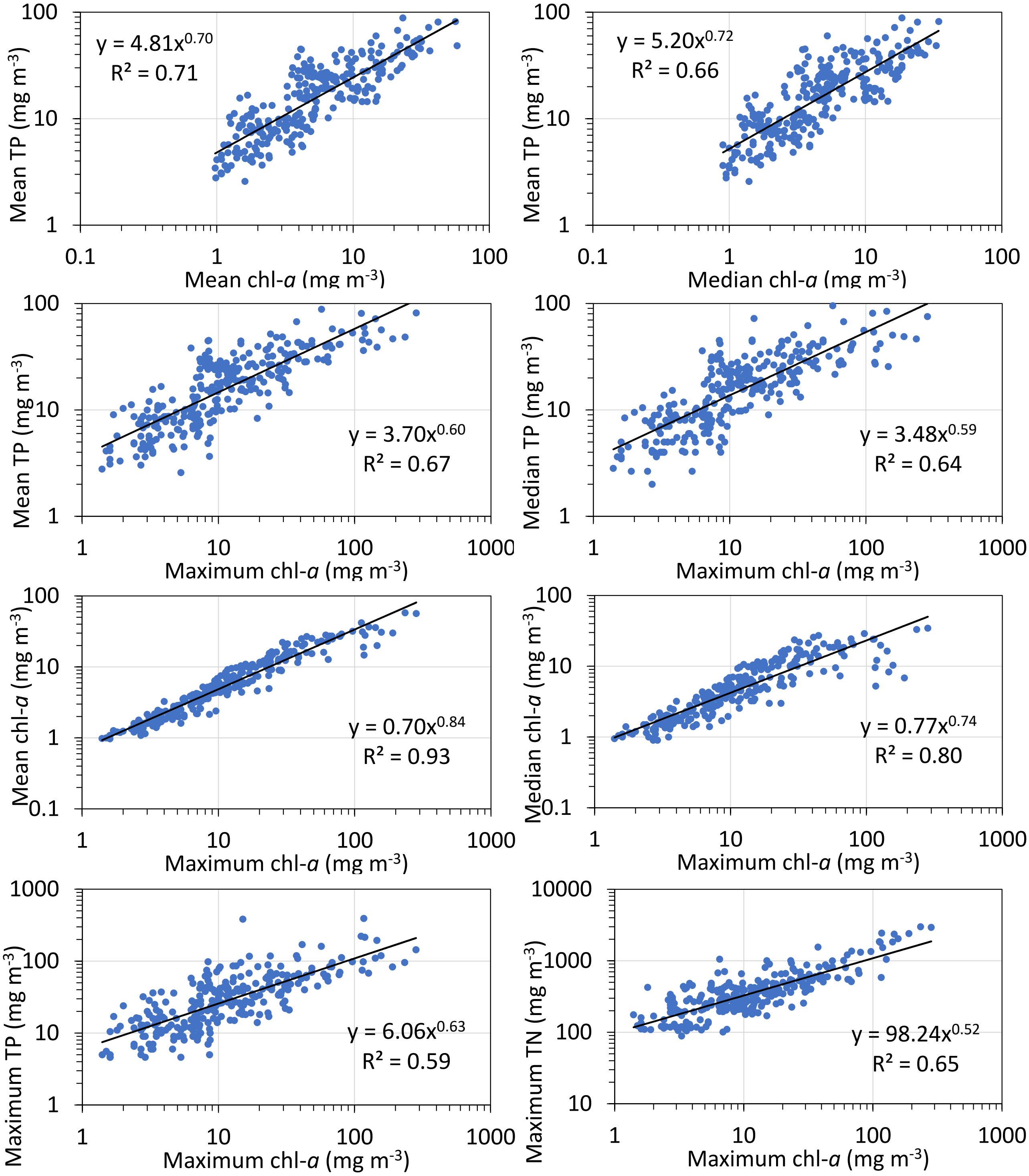


Figure 3: Relationships between statistical measures of chlorophyll and nutrients. Shown are the annual data (annual means, medians and maxima) for 2001-2020 for all 12 lakes (15 sites). Regression equations were used to convert TAS values from medians to means and chlorophyll *a* to nutrient concentrations.

2.3 The water balance and inflow nutrient concentrations

The water balance (*I* = inflow, *O* = outflow, *P* = rain on the lake, *E* is evaporation) is given by:

$I + P = O + E$, assuming long term average lake volume change to be negligible.

If estimates for *O*, *E* and *P* are available then the inflow can be derived as $I = O - P + E \Rightarrow I = O - (P - E)$. This would account both for surface inflows and net groundwater inflow.

It seems logical that $[TP]_{in}$, the average inflow TP concentration, would be estimated as the P load to the lake divided by the inflow rate. However, the literature is ambiguous. Whether the inflow rate was estimated or recorded and used to estimate $[TP]_{in}$ is often not expressly stated and at least in some of the literature it is indicated that this was not the case. It appears that, often if not always, the literature on the mass balance models (section 2.1) assumed the total inflow rate to be equal to the outflow rate, perhaps because the latter is easier to monitor or estimate.

For instance, Vollenweider writes in OECD (1982), which is probably fair to say is the most authoritative and most cited nutrient mass balance modelling study (see citations for a list of the main papers in the lake phosphorus mass balance literature in Brett and Benjamin 2008), that $T_W = V/O$ (as it was also calculated by McBride et al. 2021) $= z/q_S$ and $q_S = O/A = z/T_W$ (on page 123). Furthermore, OECD (1982) defines the annual inflow P concentration as L_P/q_S (page 123) and the total P load as $[TP]_{in}*z*A/T_W$ (page 94), which is equal to $[TP]_{in}*O$, and therefore $[TP]_{in}$ = total P load/*O*.

Based on this combination of equations in OECD (1982) it is clear that what is called the inflow concentration is not the actual inflow concentration but the load divided by the outflow rate. It is possible that nevertheless actual average inflow concentrations were used by OECD (1982) to develop the models linking nutrient loads to lake nutrient concentrations (i.e., plots of $[P]_{lake}$ versus $[TP]_{in}/(1-\sqrt{T_W})$) but because the equations in OECD (1982) contradict this it is impossible to know for certain.

Likewise, Vollenweider (1976, page 70) has $[TP]_{in} = L_P/q_S$, from which follows $[TP]_{in} = (L_P\, A)/O$, which is the total external P load divided by the outflow.

In addition, it is distinctly stated in the program documentation of the Wisconsin models (WiLMS 2003 – page 13 according to the Table of contents, page 19 if counting from the first of the unnumbered pages) that $[TP]_{in} = L_P*T_W/z$. The same page also confirms that $q_S = z/T_W$. On page 13 of WiLMS 2003 (page 7 according to the Table of contents): $T_W = 1/p$, *p* = the "hydraulic loading divided by lake volume", and "the hydraulic loading …. represents the total annual water loading to the water body. This includes point and nonpoint sources as well as the net (precipitation-evaporation) to the lake surface" which means hydraulic loading $= I + P - E = O$. On page 30: "Areal Water Load [q_S] = A calculated value of Lake Area divided by Lake Outflow" and "Residence Time = the estimated time that water stays in the lake before it flows out" $= V/O$. Altogether this means I = O is assumed by WiLMS (2003), because $[P]_{in} = L_P*T_W/z = L_P/q_S = L_P * A/O$, which is the total external P load divided by the outflow, as above for OECD (1982) and Vollenweider (1976). However, the WiLMS program documentation is also contradictive, by stating (on page 13): "The water residence time is calculated as the lake volume divided by the annual inflow or the time it takes one lake volume to be replaced by inflow", which contradicts all of the above concerning the WiLMS (2003) methodology (and contradicts also OECD 1982, while it is included as one of the WiLMS models), except if by "annual inflow" the hydraulic loading (above) is meant $= I + P – E = O$.

Therefore, to follow the same approach to the nutrient mass balance modelling as the cited modelling studies, $[TP]_{in}$ was assumed equal to L_P*T_W/z and $[TN]_{in}$ was assumed equal to L_N*T_W/z, effectively suggesting the flow rates into the lake to be equal to outflow rates.

However, to get a better idea of the consequences, inflow rates were estimated (Table 5). Annual mean rainfall data 1991-2020 for the lake locations (Figure 4) was taken from NIWA climate maps (https://niwa.co.nz/climate/research-projects/national-and-regional-climate-maps). Most rainfall occurs at Okataina, Rotoehu and Rotoma with an annual 2000-2200 mm, Rotoiti and Tarawera receive around 1800 mm, and the remaining lakes around 1500 mm (Figure 5). NIWA climate maps have not been prepared for open water evaporation. Therefore the 2016-2020 average open water evaporation, estimated from climate data recorded at two land-based weather stations at Rotorua, was taken from the Cliflo website, and the same average value (846 mm y^{-1}) was applied to all 12 lakes. Rainfall on the lake minus evaporation (P-E) from the lake data was on average about 18% of the outflow rates (Table 5). In other words, if the CliFlo derived estimate is used for E, the inflow rates would be less than the outflow rates by about 18%, on average across all lakes.

However, it is known that actual on-lake evaporation, derived from lake-based buoy data, is typically greater and can be much greater than estimated open water evaporation based on land-based measured climate variables (Verburg 2021, Woolway et al. 2017, Woolway 2018). For instance, on Lake Taupo, where evaporation is roughly 50% greater than suggested by CLiFlo open water evaporation based on land-based data (Taupo airport), P-E derived from lake-based data (buoy data) is more than 10 times smaller than when derived from land-based data (i.e., CliFlo). In other words, P and E are nearly equal. The main reasons for the greater evaporation estimated from on-lake weather data are on average higher wind speeds and higher air temperatures, especially on large lakes, compared with shore-based weather stations (Woolway et al. 2017, Woolway 2018).

Therefore, in view of 1) the probable relatively small numbers for average P-E, 2) the uncertainty of the real value of P-E for each lake, and 3) the fact that in the nutrient mass balance modelling literature the outflow rates appear to have been used as a proxy for inflow rates, it may be justifiable to assume annual mean flow rates into the lake as equal to outflow rates. However, there is one caveat, that the effect of using lake-based data instead of land-based data to estimate open water evaporation likely increases with the size of a lake. Therefore, in reality outflow and inflow rates may differ more in small lakes than in large lakes.

This study applies models from literature and the models should be applied in the same way. Nevertheless, for those models that use $[TP]_{in}$ or $[TN]_{in}$ as part of the equation the calculations were done both based on inflow and outflow rates to estimate $[TP]_{in}$ and $[TN]_{in}$. This applies only to the models of Abell et al. 2019, Larsen-Mercier (1976) and OECD (1982) for P, and Abell et al. (2019) and OECD (1982) for N. The remaining models are not affected, as they do not use the symbols $[TP]_{in}$ or $[TN]_{in}$ in their equations and they effectively assume $I = O$ by multiplying L_P/q_S or L_PT_W/z by (1-R) to estimate in-lake nutrient concentrations. T_W and q_S remained unchanged, functions of outflow rates, as in the literature (cited above) and in McBride et al. (2021). Therefore, in the case of inflow based calculations in the mentioned five models the inflow nutrient concentrations are *not* equal to L_XT_W/z, with x standing for P or N.

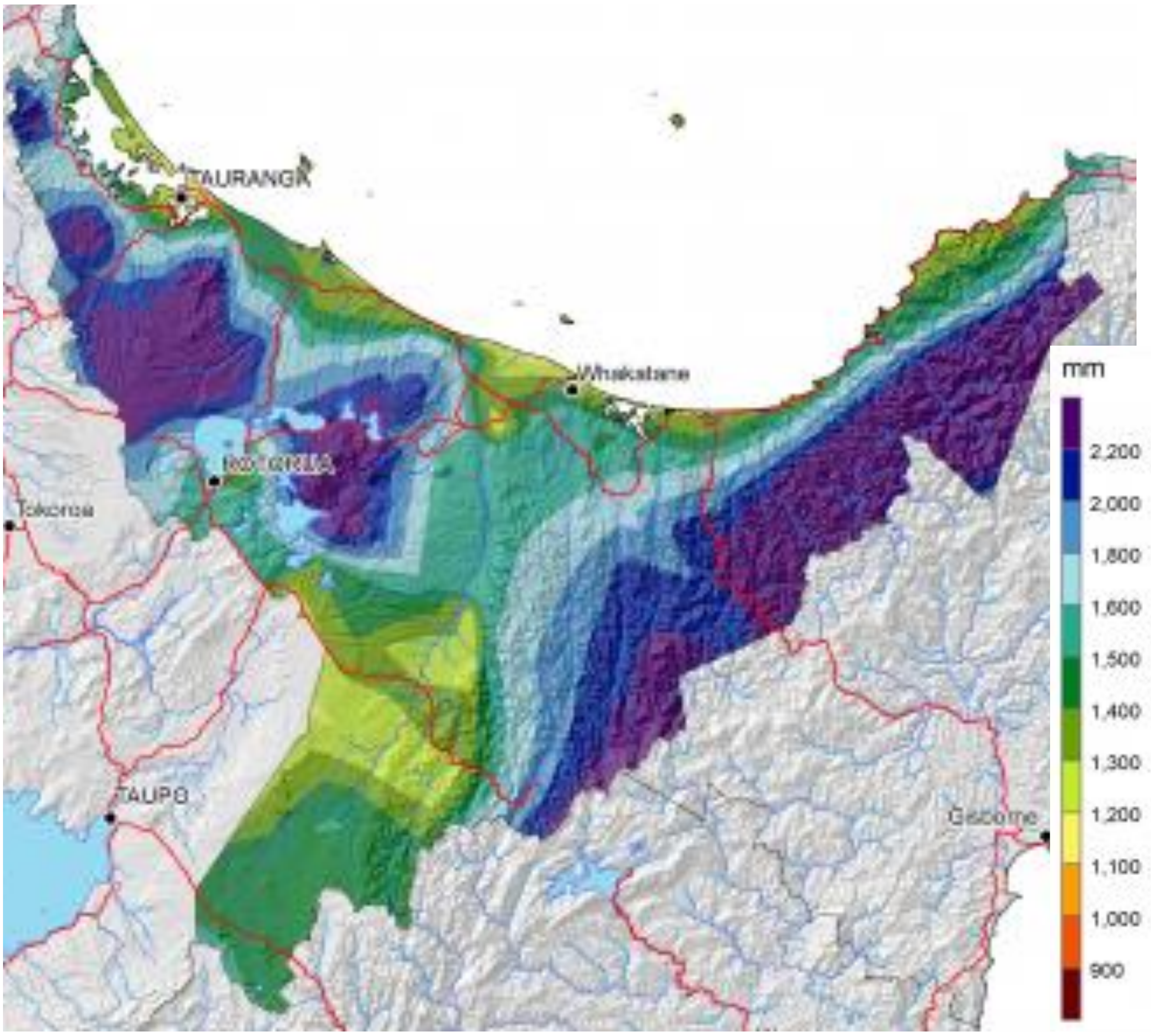


Figure 4: Rainfall distribution across the Bay of Plenty Region. Annual mean rainfall of 1991-2020. NIWA climate maps (Niwa.co.nz).

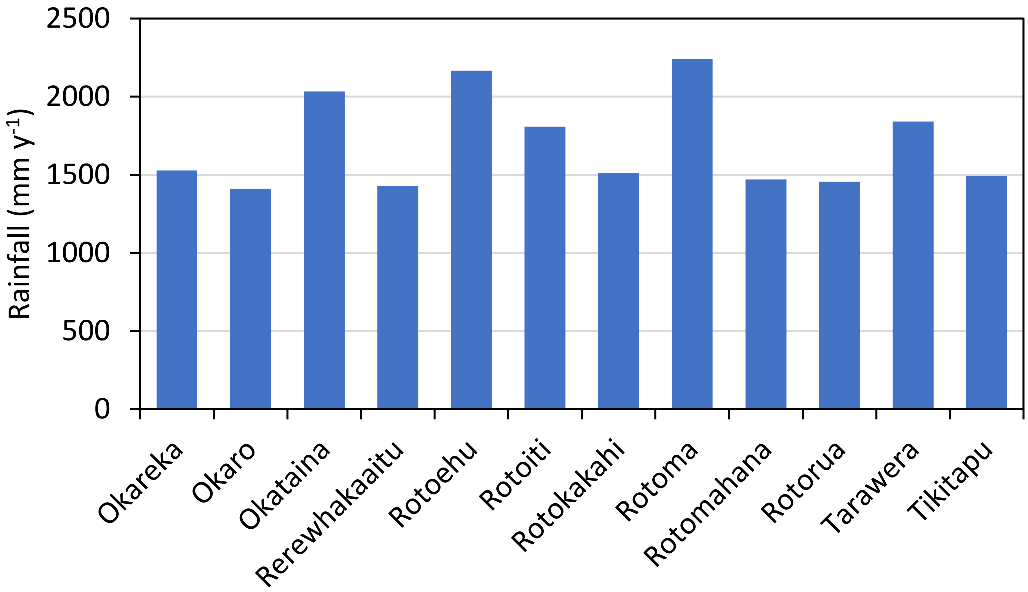


Figure 5: Rainfall at the lake locations. Mean annual rainfall in 1991-2020.

Table 4: Lake morphology, hydrology and water quality. V = volume, z = mean depth, A = lake area, T_w = residence time = V/O (McBride et al. 2021), z_{max} = maximum depth (Scholes and Hamill 2016), q_s = areal water loading rate (O/A), TN and TP concentrations are the means of 2016-2020.

Lake	V	z	z_{max}	A	O	T_w	q_s	TN_{lake}	TP_{lake}
	1000 m^3	m	m	km^2	m^3s^{-1}	y	my^{-1}	mg m^{-3}	mg m^{-3}
Okareka	63594	19.0	33.5	3.34	0.53	3.80	5.0	189	8.9
Okaro	3445	10.4	18	0.33	0.11	0.99	10.5	687	68.0
Okataina	499040	46.5	78.5	10.73	2.33	6.79	6.9	89	7.0
Rerewhakaaitu	36647	7.1	15.8	5.17	0.32	3.63	2.0	369	11.2
Rotoehu	61001	7.7	13.5	7.9	1.98	0.98	7.9	477	34.9
Rotoiti	1042300	30.9	126	33.69	4.83	6.84	4.5	188	23.4
Rotokakahi	76832	17.7	32	4.33	0.5	4.87	3.6	220	15.4
Rotoma	455803	45.4	83	10.03	1.27	11.37	4.0	109	4.1
Rotomahana	479050	53.1	125	9.02	2.12	7.16	7.4	203	24.6
Rotorua	801690	10.0	45	80.48	15.57	1.63	6.1	324	17.6
Tarawera	2273700	55.3	87.5	41.15	5.93	12.15	4.5	92	11.5
Tikitapu	26320	18.3	27.5	1.44	0.13	6.42	2.8	174	4.4

Table 5: Estimation of inflow rates. Precipitation = rainfall on the lake, P-E = precipitation minus evaporation from the lake (calculated by multiplying precipitation and evaporation by lake area), I = estimated total inflow rate including net groundwater, (P-E)/O = the proportion (%) relative to the outflow. The same evaporation rate was applied for each lake (846 mm y^{-1}) while rainfall averages were applied separately for each lake (see methods).

Lake	Precipitation mm y^{-1}	Precipitation $m^3 s^{-1}$	*P-E* $m^3 s^{-1}$	*I* $m^3 s^{-1}$	(*P-E*)/*O* %
Okareka	1527	0.16	0.07	0.46	14
Okaro	1412	0.01	0.01	0.10	5
Okataina	2033	0.69	0.40	1.93	17
Rerewhakaaitu	1429	0.23	0.10	0.22	30
Rotoehu	2168	0.54	0.33	1.65	17
Rotoiti	1809	1.93	1.03	3.80	21
Rotokakahi	1511	0.21	0.09	0.41	18
Rotoma	2240	0.71	0.44	0.83	35
Rotomahana	1471	0.42	0.18	1.94	8
Rotorua	1455	3.71	1.55	14.02	10
Tarawera	1841	2.40	1.30	4.63	22
Tikitapu	1494	0.07	0.03	0.10	23

Table 6: Lake catchment loads. N_{EXT} and P_{EXT} are the total external nutrient loads (from McBride et al. 2021) and average inflow concentrations are given by [nutrient]$_{in}$, estimated as total load either divided by outflow rate (see methods text) or divided by the inflow rate.

Lake	N_{EXT} kg y^{-1}	P_{EXT} kg y^{-1}	N_{EXT}/O $[TN]_{in}$ mg m^{-3}	P_{EXT}/O $[TP]_{in}$ mg m^{-3}	N_{EXT}/I $[TN]_{in}$ mg m^{-3}	P_{EXT}/I $[TP]_{in}$ mg m^{-3}
Okareka	14912	984.7	892	59	1032	68
Okaro	4045	633.6	1165	183	1231	193
Okataina	22821	1385.4	310	19	375	23
Rerewhakaaitu	15823	1080.8	1567	107	2233	153
Rotoehu	45418	3370	727	54	873	65
Rotoiti	109876	4087.9	721	27	916	34
Rotokakahi	9847	857.4	624	54	763	66
Rotoma	15332	2072.6	383	52	587	79
Rotomahana	95579	20705.8	1429	310	1560	338
Rotorua	721427	56760.5	1468	115	1631	128
Tarawera	106767	10652.6	571	57	730	73
Tikitapu	2187	116.5	533	28	690	37

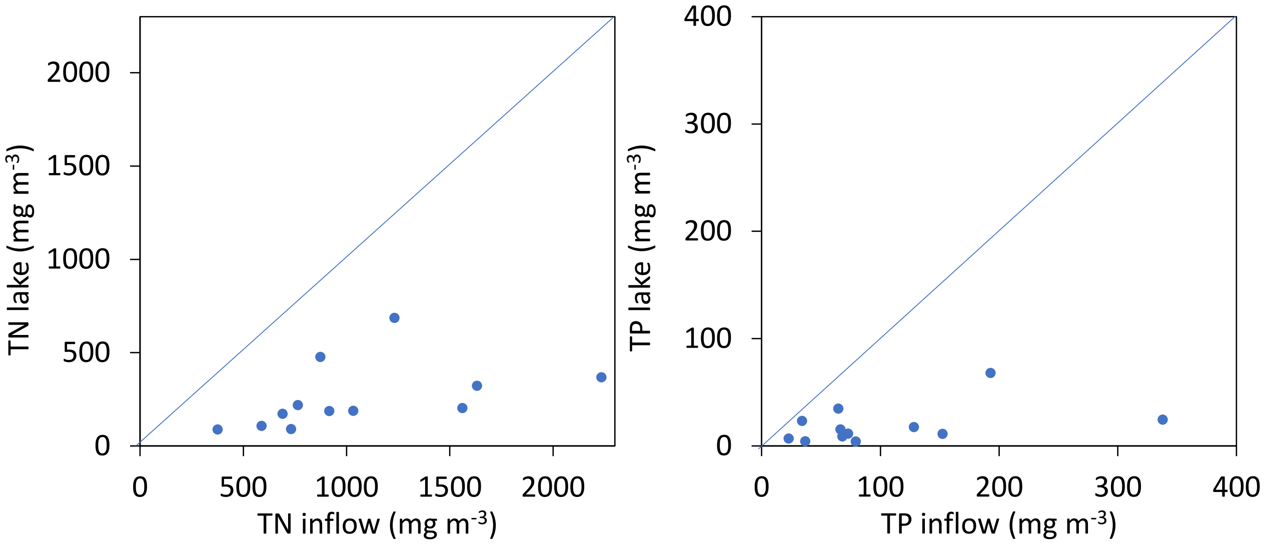


Figure 6: Lake nutrient concentrations versus inflow concentrations. Blue line is the 1:1 line. Inflow concentrations were calculated as the loads (McBride et al. 2021) divided by the inflow rate estimated as $I = O - P + E$.

The average lake nutrient concentrations (2016-2020) are lower and in some cases much lower than the average inflow concentrations, as estimated by dividing the estimated load by the estimated inflow rate (Figure 6). The difference is explained by retention, by burial in the sediment and for N by denitrification. Therefore, the estimation of the retention is an important aspect in lake nutrient mass balance modelling (section 2.1).

2.4 Denitrification

Fixed nitrogen is nitrogen that is contained in a compound such as nitrate or in organic material, as opposed to N_2 gas, which is a form of nitrogen in general not available to phytoplankton. The removal of fixed nitrogen from the lakes by denitrification (reduction of nitrate ultimately to N_2) was estimated from the available data for TN and TP loads, in-lake TN and TP concentrations, outflow rates, and sediment TN and TP contents. The variables used in this analysis are defined in Table 7.

Table 7: Definitions of variables in the mass balance equation for denitrification. Units t y^{-1}.

Variables	Meaning
TN_{in}, TP_{in}	Total external loads into the lake
TN_{out}, TP_{out}	Nutrients lost from the lake via the outflow
$TP_{retained}$	P not lost from the lake via the outflow = TP_{in} - TP_{out} = Sediment burial (TP_{sed})
$TN_{retained}$	N not lost from the lake via the outflow = Sediment burial (TN_{sed}) + Denitrification ($TN_{denitrified}$)
TN_{sed}, TP_{sed}	Burial rates of nutrients in the lake sediment. Nutrient concentrations in the sediment were taken from Trolle et al. (2008). Note that in the equation used to estimate denitrification only the ratio TN_{sed}:TP_{sed} is used, therefore the magnitude of the fluxes need not be known.
$\left[\frac{TN_{sed}}{TP_{sed}}\right]$	Ratio of concentration of N and P in the lake sediment, taken from Trolle et al. (2008).

The method to estimate denitrification from the mass balance is based on the following three points:

1) The amount of TP buried per year in the sediment is known from the difference of the P load into the lake and the load exiting the lake via the outlet:

$$TP_{sed} = TP_{in} - TP_{out}$$

2) The rate of N burial from mass balance is:

$$TN_{sed} = TN_{in} - TN_{out} - TN_{denitrified}$$

3) The ratio of the rates of TN to TP burial will be the same as the ratio of the TN to TP concentrations in the buried sediment. By combining TP_{sed} with the measured ratio of the concentrations of TN and TP in the sediment the rate of TN burial in the sediment (TN_{sed}) can be estimated:

$$TN_{sed} = TP_{sed}\left[\frac{TN_{sed}}{TP_{sed}}\right]$$

These principles result in the following equation:

$$\left[\frac{TN_{sed}}{TP_{sed}}\right] = \frac{TN_{in} - TN_{out} - TN_{denitrified}}{TP_{in} - TP_{out}}$$

It follows that denitrification (t yr^{-1}) can be estimated as:

$$TN_{denitrified} = TN_{in} - TN_{out} - (TP_{in} - TP_{out})\left[\frac{TN_{sed}}{TP_{sed}}\right]$$

$$TN_{denitrified} = (TP_{in} - TP_{out})\left(\frac{TN_{in} - TN_{out}}{TP_{in} - TP_{out}} - \left[\frac{TN_{sed}}{TP_{sed}}\right]\right)$$

which reduces to:

$$TN_{denitrified} = TP_{retained}\left(\frac{TN_{retained}}{TP_{retained}} - \left[\frac{TN_{sed}}{TP_{sed}}\right]\right)$$

It follows that $TN_{retained}$:$TP_{retained}$ cannot be less than TN_{sed}:TP_{sed}. $TN_{retained}$:$TP_{retained}$ > TN_{sed}:TP_{sed} can be explained by denitrification.

The method assumes that an additional source of nitrogen by N fixation is negligible relative to the TN load. However, actual denitrification may be larger where this process occurs. The method also places all microbial processes resulting in loss of fixed nitrogen under the label “denitrification”.

3 Results and Discussion

3.1 Water quality measurements and TAS

Appendix A shows figures with the annual mean TN, TP and chlorophyll *a* concentrations in each lake since 2000 with the minimum and maximum TAS indicated. A brief summary of the general trends is included.

The proportions by which the average in-lake concentrations in 2016-2020 were higher than the minimum TAS, if any, are in Table 8.

Median TP concentrations were higher than the minimum TAS in seven lakes: Okaro, Okataina, Rerewhakaaitu, Rotoehu, Rotoma, Rotorua and Tikitapu. Median TN concentrations were higher than the minimum TAS in four lakes: Rerewhakaaitu, Rotoma, Rotomahana and Rotorua. Median Chl-*a* concentrations were higher than the minimum TAS in nine lakes: Rerewhakaaitu, Rotoehu, Rotoiti, Rotokakahi, Rotoma, Rotomahana, Rotorua, Tarawera and Tikitapu. Maximum Chl-*a* concentrations were higher than the minimum TAS in nine lakes: Okareka, Okaro, Rerewhakaaitu, Rotoiti, Rotoma, Rotomahana, Rotorua, Tarawera and Tikitapu. Clearly the TAS have been set more rigorously for chlorophyll than for the nutrient concentrations (Figure 7). For instance, in Lake Tikitapu large reductions in chlorophyll are suggested but virtually none in nutrients. Maximum Chl-*a* concentrations requires more reduction than median Chl-*a* concentrations in seven lakes, and on average across all 12 lakes.

Table 8: Proportions by which recent in-lake concentrations were higher than the minimum TAS. Bold values indicate where reductions are required to meet the minimum TAS. Based on 2016-2020 water quality data.

	Median TP	Median TN	Median Chl-*a*	Maximum Chl-*a*
	%	%	%	%
Okareka	-4	-5	-21	**9**
Okaro	**65**	-8	-9	**24**
Okataina	**3**	-1	0	-25
Rerewhakaaitu	**23**	**22**	**22**	**35**
Rotoehu	**54**	**0**	**78**	-6
Rotoiti	-14	-9	**12**	**32**
Rotokakahi	-5	-13	**30**	-32
Rotoma	**39**	**7**	**4**	**102**
Rotomahana	-16	**6**	**23**	**52**
Rotorua	**18**	**2**	**20**	**71**
Tarawera	-20	-8	**44**	**19**
Tikitapu	**4**	0	**108**	**68**

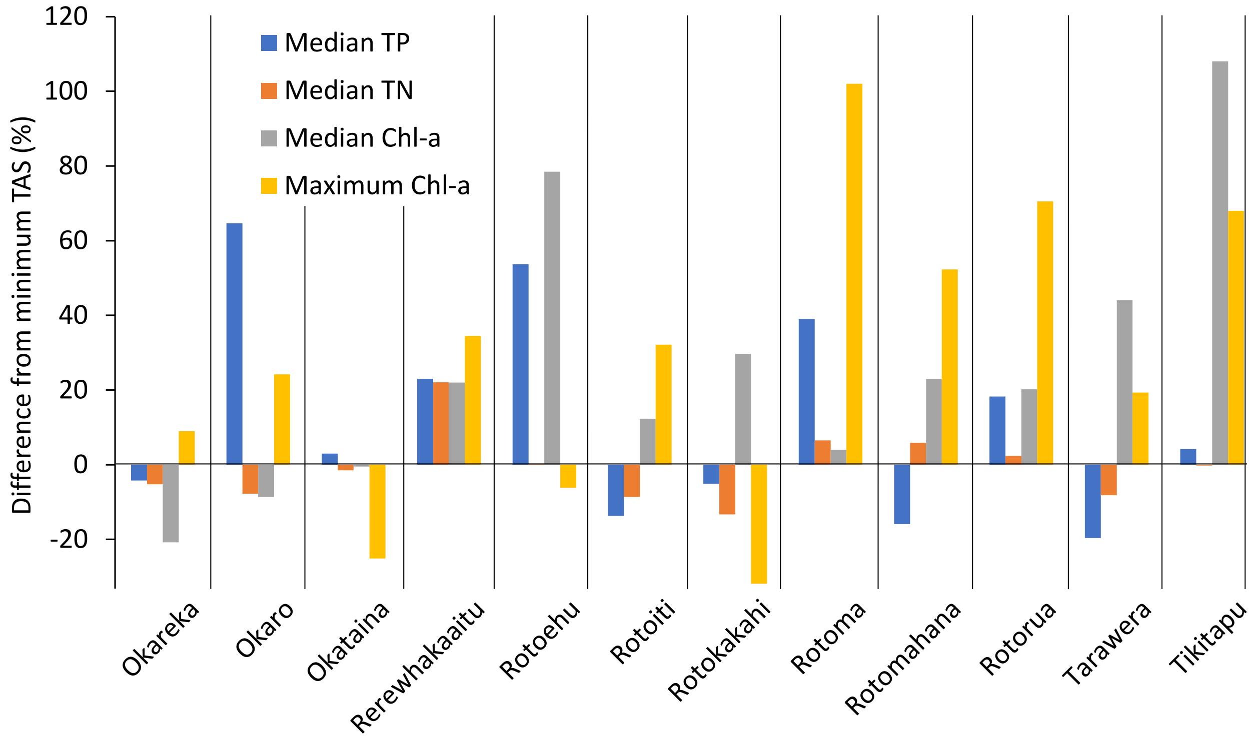


Figure 7: The proportions by which the in-lake concentrations in 2016-2020 were higher than the minimum TAS. Positive values indicate when lake concentrations exceed the minimum TAS. Note that in the BOPRC terminology minimum TAS > maximum TAS (see Table 3 for TAS).

The proportion change needed to achieve TAS (Table 8) can be high when observed concentrations are in fact already very low. For instance, in Lake Tikitapu, the TAS for the median chlorophyll *a* concentration is only 1 mg m^{-3}, while the median in 2016-2020 was 2.1 mg m^{-3}, 108% higher (which means TAS for median chlorophyll *a* concentration is 52% lower than the recent average). When concentrations are already that low more caution should be taken to measure achievement of goals set due to the potentially greater influence of analytical noise.

In Figure 8 the TAS values for annual median TP, median TN, median chlorophyll *a* and maximum chlorophyll *a* are compared with their median concentrations in each lake as observed in 2016-2020. The median TP concentrations in 2016-2020 exceeded the minimum TAS in seven lakes: Okaro, Okataina, Rerewhakaaitu, Rotoehu, Rotoma, Rotorua and Tikitapu. The median TN concentrations in 2016-2020 exceeded the minimum TAS in four lakes: Rerewhakaaitu, Rotoma, Rotomahana, and Rotorua. The median chlorophyll *a* concentrations in 2016-2020 exceed the minimum TAS in eight lakes: Rerewhakaaitu, Rotoehu, Rotoiti, Rotokakahi, Rotomahana, Rotorua, Tarawera and Tikitapu. The mean annual maximum chlorophyll *a* concentrations in 2016-2020 exceeded the minimum TAS in nine lakes: Okareka, Okaro, Rerewhakaaitu, Rotoiti, Rotoma, Rotomahana, Rotorua, Tarawera and Tikitapu. Only in Lakes Okataina, Rotoehu and Rotokakahi did the annual maximum chlorophyll *a* concentrations in 2016-2020 not exceed the minimum TAS, even though in two of those three lakes median concentrations did exceed the minimum TAS.

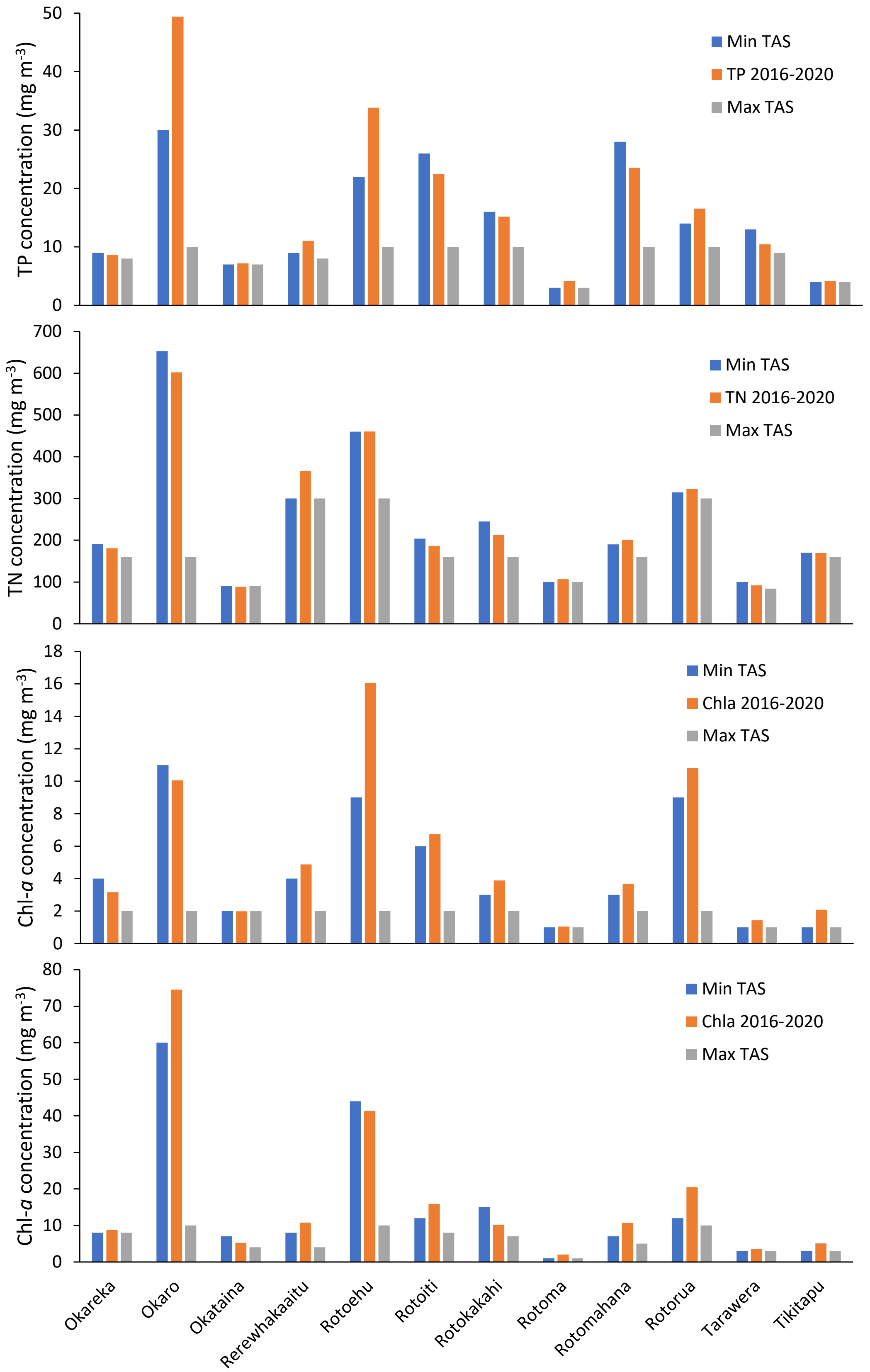


Figure 8: The TAS for nutrients and chlorophyll a compared with median concentrations in each lake observed in 2016-2020. From top to bottom: annual median TP, median TN, median chlorophyll *a* and maximum chlorophyll *a*.

3.2 Predicted lake nutrient concentrations

Figure 9 shows lake P concentrations predicted by all P models and compares the predictions with average measured P concentrations in 2012-2017 (from McBride et al. 2021) and in 2016-2020. While there are differences in average P concentrations between the two periods (higher in the later period in seven of the lakes), in most lakes most models appeared not to agree well with the observed P concentrations.

Figure 10 shows lake N concentrations predicted by the three models using the loads in McBride et al. (2021) and compares the predictions with average N concentrations in 2012-2017 (from McBride et al. 2021) and in 2016-2020.

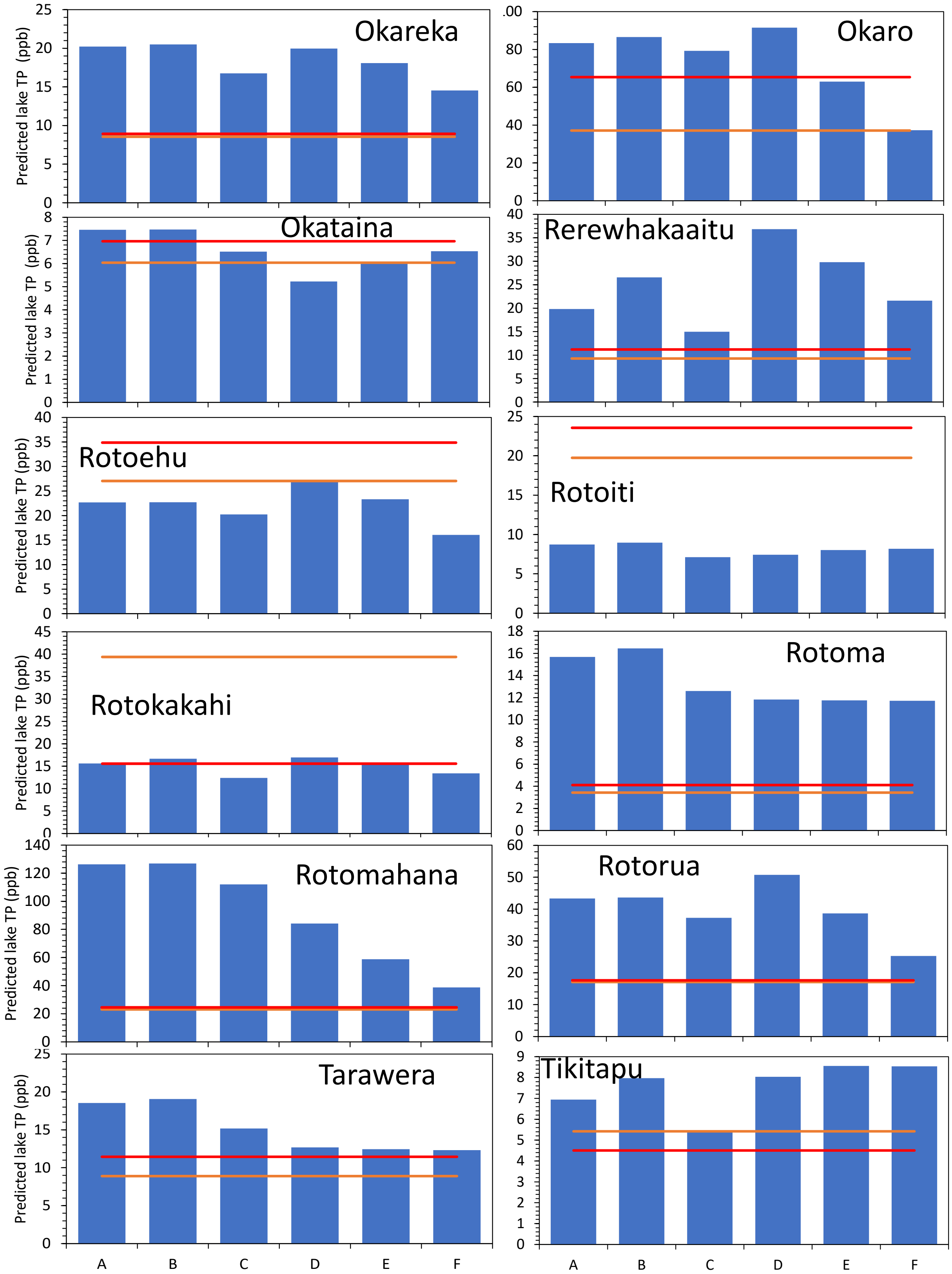


Figure 9: **Predicted lake P concentrations for all models.** Compared with observed average nutrient concentrations in 2012-2017 (from McBride et al. 2021) and in 2016-2020. Models are: A. Dillon, Rigler (1974), Kirchner (1975), B. Nurnberg 1984, C. Reckhow 1979, D. Vollenweider 1976, E. OECD 1982, F. Abell et al. 2019. Orange line indicates the average lake concentrations in McBride et al. 2021 (2012-2017). Red line indicates the 5-year averages of 2016-2020 (listed as "Top" data in the BOPRC data files). ppb = mg m^{-3}.

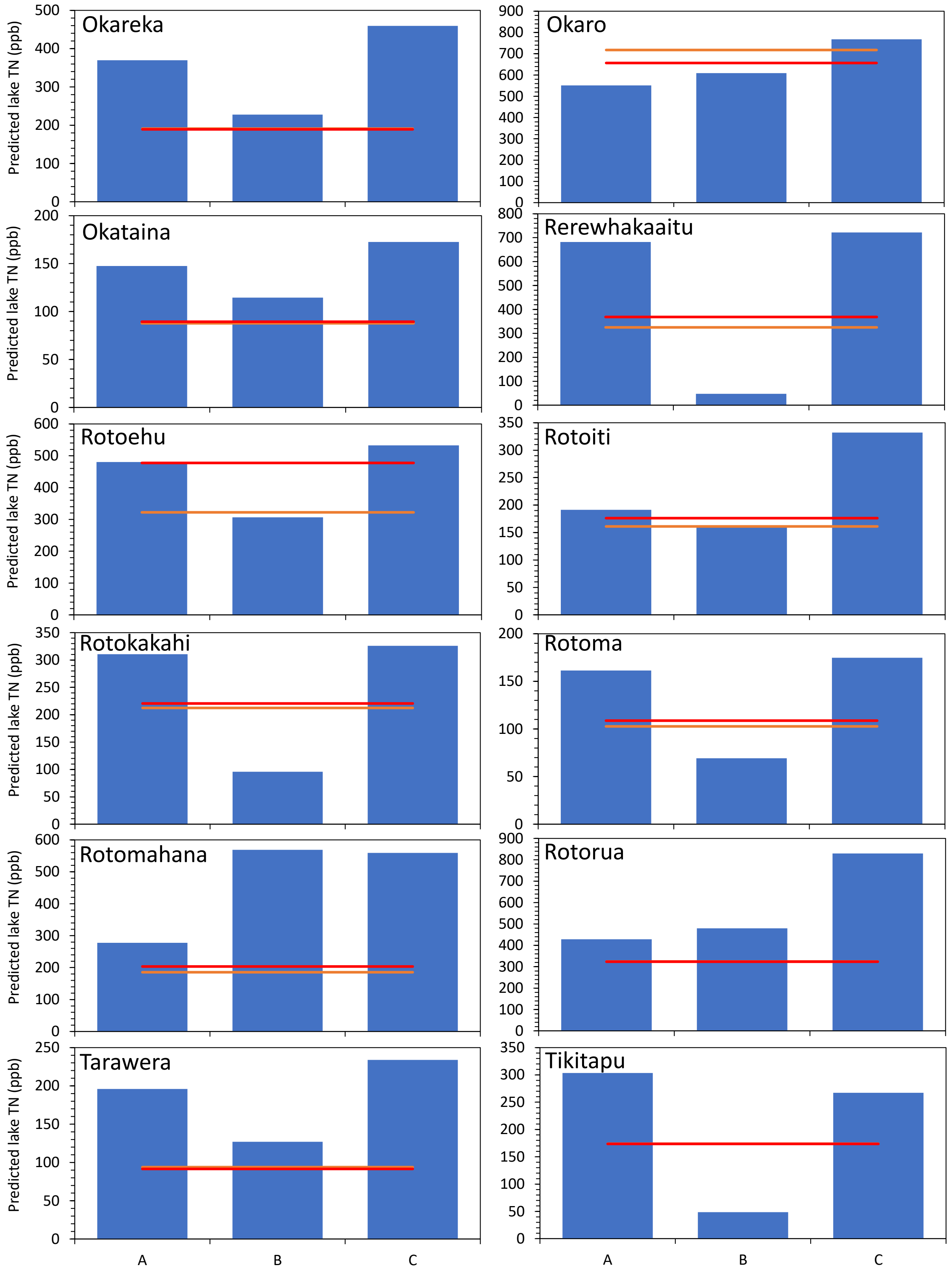


Figure 10: Predicted lake N concentrations for three models. Compared with observed average N concentrations in 2012-2017 (from McBride et al. 2021) and in 2016-2020. Models are: A. Abell et al. 2019, B. Harrison 2009/Vollenweider 1976 combination, C. OECD 1982. Orange line indicates the average lake concentrations in McBride et al. 2021 (2012-2017). Red line indicates the 5-year averages of 2016-2020 (“Top” data). (Identical if only 1 line shows).

3.3 Predicted nutrient loads to achieve minimum and maximum TAS

Figure 11 shows predicted P loads corresponding with minimum and maximum TAS according to the six models with the estimate of the present load indicated (McBride et al. 2021), and Figure 12 shows the same for nitrogen. Figure 13 shows the P loads predicted to achieve minimum and maximum TAS for median chlorophyll *a* according to the six models.

In seven lakes both the minimum and the maximum TAS for TP required loads that were lower than the estimates of the current P load, according to all six models. In three of the remaining lakes (Rotoehu, Rotokakahi, Tarawera) the required TP load to meet the maximum TAS for TP was lower than the estimate of the current P load, according to all six models, but some of the models suggested that the minimum TAS could be achieved with the estimate of the current P load. In two lakes (Rotoiti and Okataina) the models suggested that the maximum TAS for TP could be achieved with the current estimate of the P load.

In seven lakes both the minimum and the maximum TAS for TN required loads that were lower than the estimates of the current N load, according to at least two of the three models. In four of the remaining lakes (Rotoehu, Rotokakahi, Okaro, Rotoiti) at least two of the three models suggested that the minimum TAS could be achieved with the estimate of the current N load but not the maximum TAS. In lake Tikitapu at least two of the three the models suggested that the maximum TAS for TN could be achieved with the current estimate of the N load.

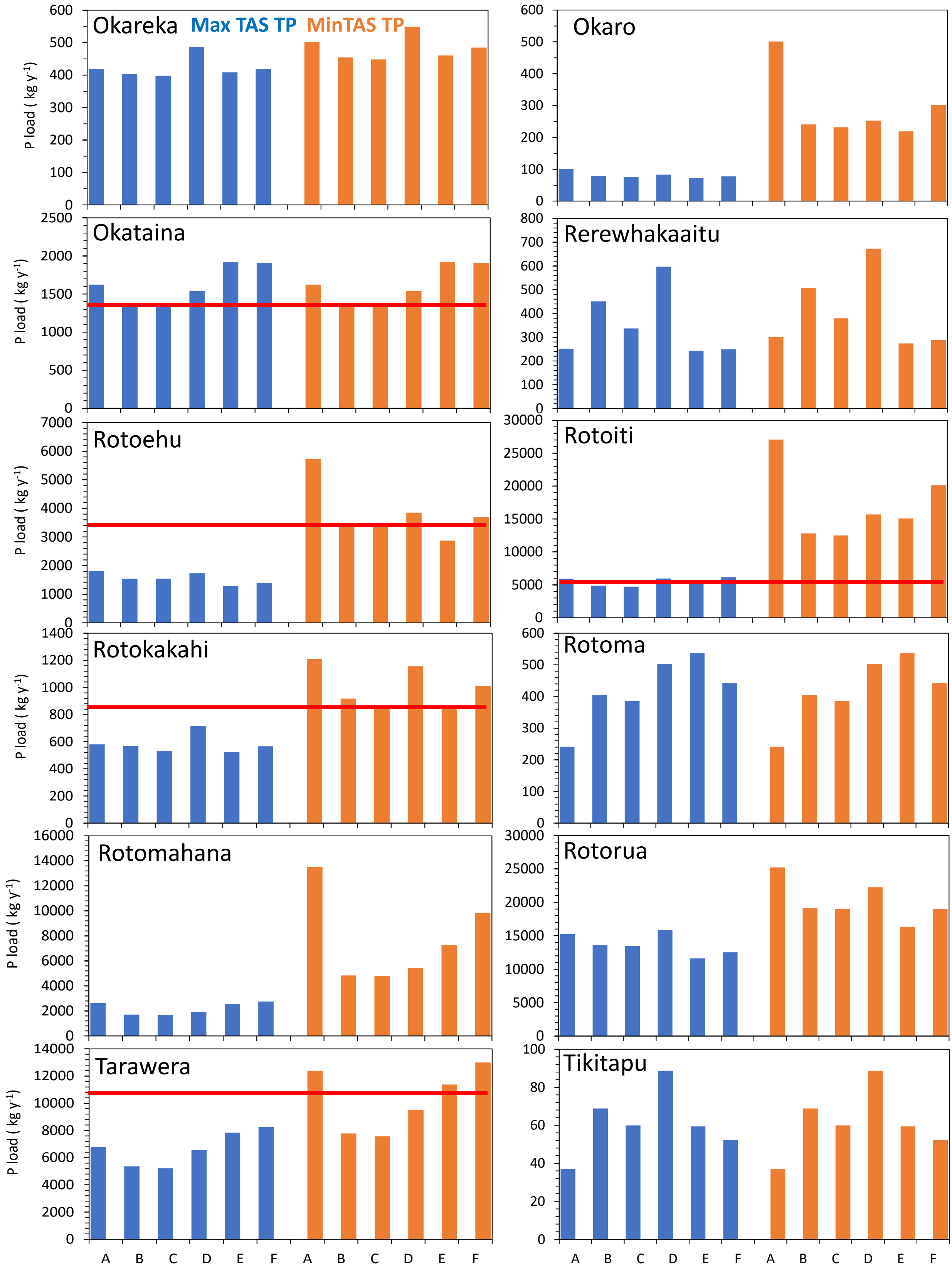


Figure 11: Predicted P loads to achieve minimum and maximum TAS according to six models. Models: A. Abell et al. 2019, B. Dillon, Rigler (1974) & Kirchner (1975), C. Nurnberg 1984, D. Reckhow 1979, E. Vollenweider 1976, F. OECD 1982. Red bars indicate the estimate of the present load (McBride et al. 2021). Where there are no red bars the loads are higher than the range on the y-axis (7 lakes).

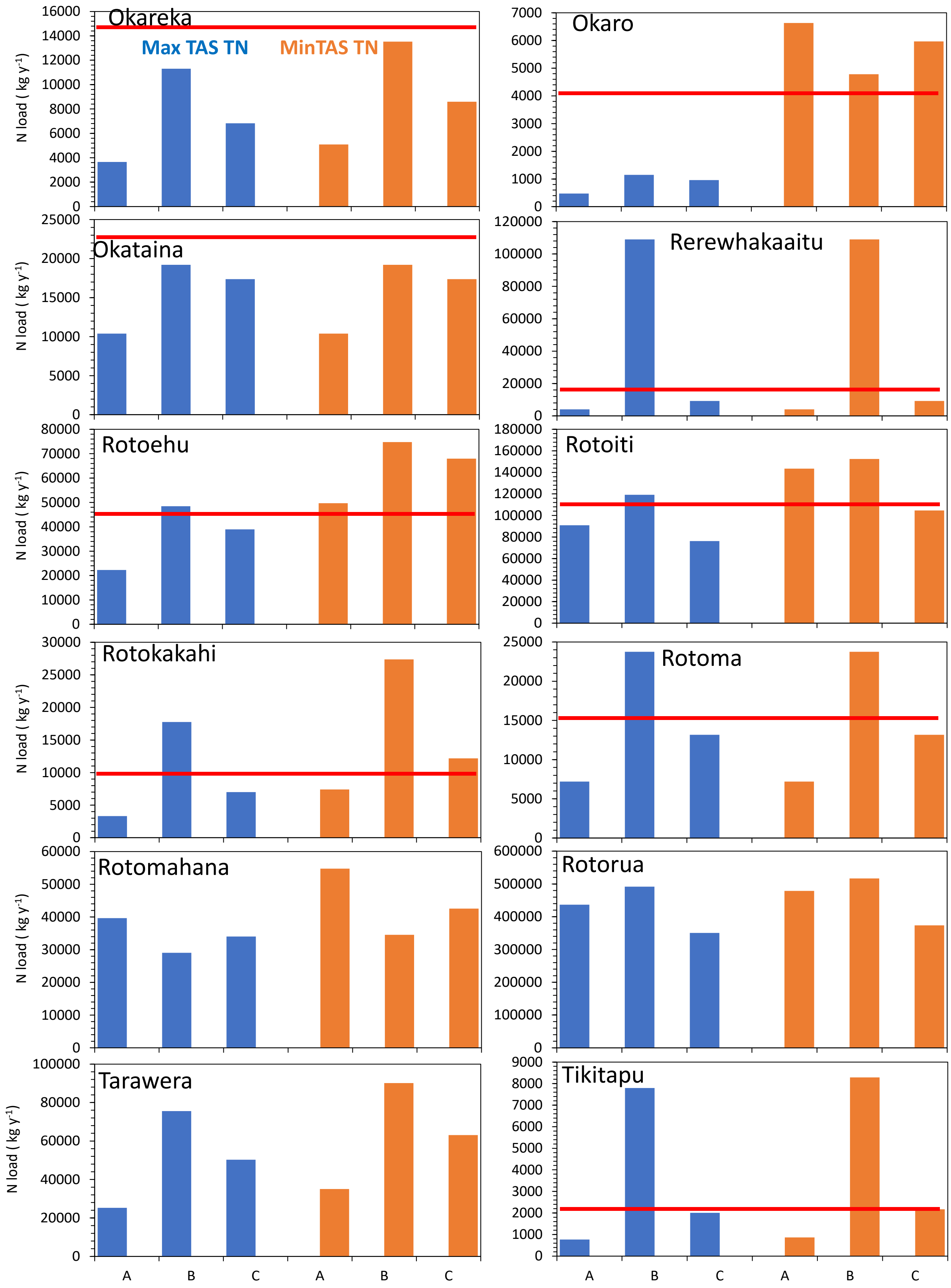


Figure 12: Predicted N loads to achieve minimum and maximum TAS using three models. Models: A. Abell et al. 2019, B. Harrison 2009/Vollenweider 1976 combination, C. OECD 1982. Red bars present the estimated load (McBride et al. 2021). Where red bars are absent the loads are higher than the y-axis range.

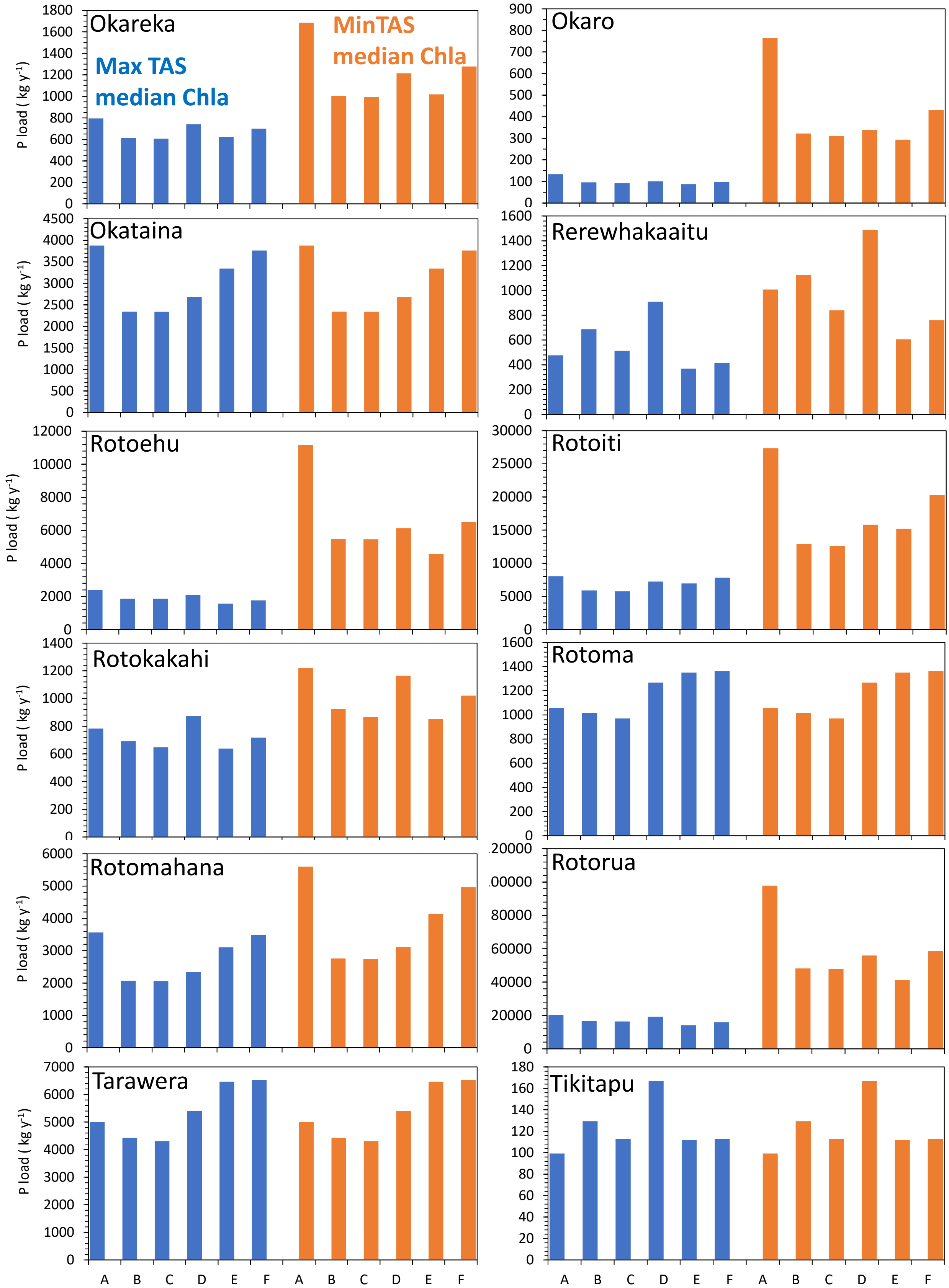


Figure 13: Predicted P loads to achieve minimum and maximum TAS for median chlorophyll a according to six models. Models: A. Abell et al. 2019, B. Dillon, Rigler (1974) & Kirchner (1975), C. Nurnberg 1984, D. Reckhow 1979, E. Vollenweider 1976, F. OECD 1982.

3.4 Relationship between TLI and TAS, and the effect of Secchi depth

Figure 14 shows the TLI as calculated from the values of minimum and maximum TAS for TP, TN and chlorophyll *a*.

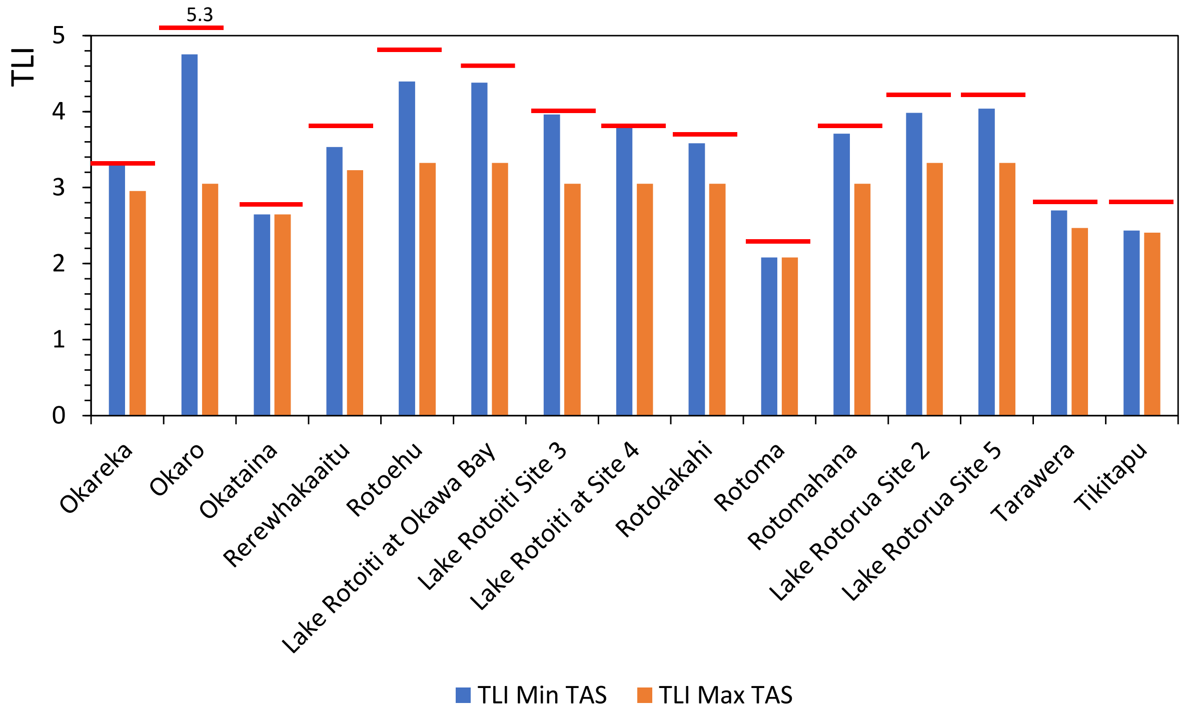


Figure 14: The TLI calculated from the values of minimum and maximum TAS for TP, TN and chlorophyll *a*. Red bars indicate the TLI based on average observed TN, TP and chlorophyll *a* concentrations in 2016-2020. There are no TAS for Secchi depth (a component of the full TLI).

The TLI (Figure 14) is here based on TP, TN and chlorophyll *a*, excluding Secchi depth because there is no TAS for Secchi depth. It is important to note that the TLI normally includes a component for Secchi depth, when available. This is important because the TLI component derived from Secchi depth (using the equation in Burns et al. 2009, revised from Burns et al. 2000) is typically better (i.e., lower) than the overall TLI, as illustrated by Figure 15. This is curious because Burns et al. 2000 state that "The assumption underlying this normalisation is that on average: TLc = TLs = TLp = TLn", i.e. all four components of the TLI should be similar, on average, across lakes.

However, while the four components would assumedly be equal for the original data set used to construct the equations, for new data sets they would never be exactly equal depending on which lakes and years of their data are used to fit the equations. In other words, when a data set is used different from the original data set there will always be one or more TLI components that are larger than the average TLI and one or more that are smaller.

The TLI equation was updated by Burns et al. (2009) for the Rotorua Te Arawa lakes, to remove the impact of peat lakes outside the Region (personal communication Paul Scholes), because in peat lakes Secchi depth is relatively low and less related to phytoplankton biomass than in non-peat lakes such as those in the Bay of Plenty.

This revision of the equation for the Secchi depth TLI component (TLI_SD = 5.1 + 2.6 x log [1/SD-1/40]) actually reduces it further than the equation of Burns et al. (2000), because the equation is a linear function of log [1/SD-1/40], which produces negative values, and the slope in the updated equation is greater (2.60, while it was 2.27 in the earlier equation), and a greater slope times negative values means more negative values.

In summary, it is likely that all TLI values in Figure 14 would be somewhat lower when Secchi depth is included in the TLI.

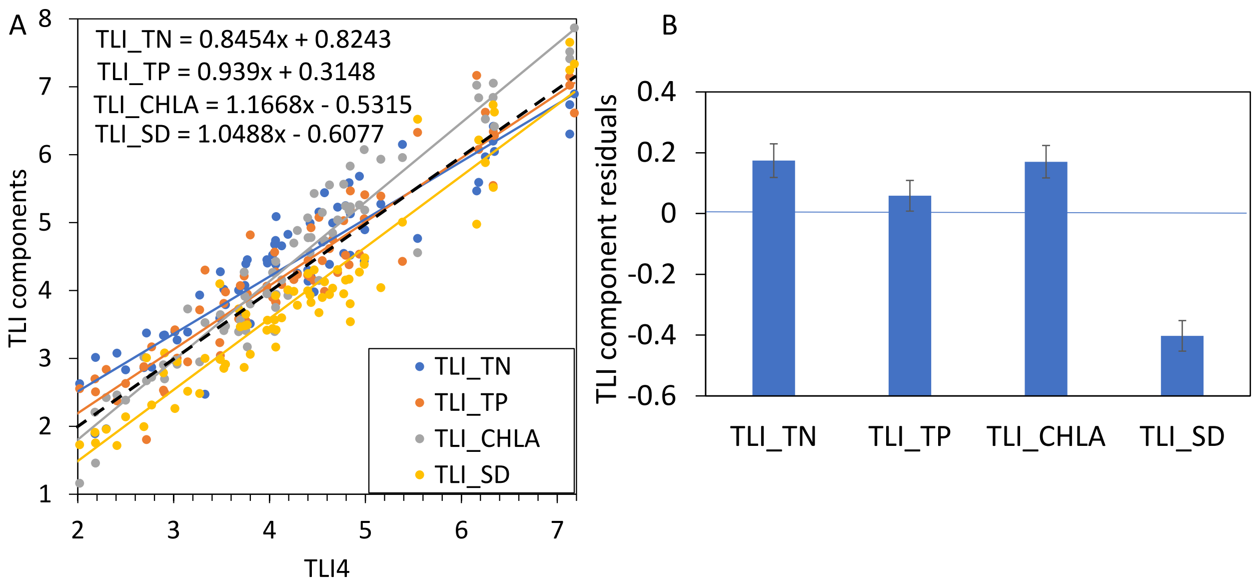


Figure 15: Comparing TLI components including Secchi depth. (A) The graph shows long-term mean data for 70 lakes (Verburg et al. 2010) with linear regressions. The black broken line indicates 1:1. TLI4 means the TLI is based on four components. (B) Shows, for the same data, the average divergence of each of the four TLI components from their average, the TLI4, with standard error bars.

3.5 Loads estimated by models for 2001-2020

This section examines the behaviour of the models in their prediction of nutrient loads across a range of in-lake nutrient concentrations as occurred in each lake in 2001-2020, and compares with the estimate of the current load. Figures 16 and 17 show P and N loads as predicted from annual mean concentrations in the lakes in 2001-2020, when available - in some of the lakes data started later than 2001. N = 20 years for each model, except for Lake Rotorua for which data are included for two sampling sites (sites 2 and 5 combined, n = 40 annual means for each model). Regression equations are only shown in the figure for Lake Rotorua. All R^2 values are 1. While the regression equations account for only 2 variables (loads and in-lake concentrations), unlike the models themselves which account for information such as residence time, lake depth, lake area and areal water loading, these additional variables are assumed not to vary for individual lakes, which explains the perfect correlations. The Dillon and Rigler model for P is usually almost the same as the Nurnberg model when $5 < q_s < 8$, and because of this overlap is not always visible in the plots (exceptions: lakes Rerewhakaaitu and Tikitapu), with model abbreviations as in the caption of Figure 16.

The Abell et al. (2019) models for N and P are the only models that are visibly non-linear (i.e., the OECD models are indistinguishable from linear within the range of P and N concentrations in the BoP lakes).

All six models underestimated the P load in lakes Rotorua, Okareka, Rerewhakaaitu, Rotoma, Rotomahana, Tarawera and Tikitapu and all models overestimated the P load in Lake Rotoehu and Rotoiti. All three models underestimated the N load in lakes Rotorua, Okareka, Okataina, Rotomahana and Tarawera and all models overestimated the N load in Lake Okaro.

The findings of Figures 16 and 17 are summarized in Figure 19, showing the average absolute deviations (%) of P and N load estimates from expected for each model.

The Abell et al. model (2019) for P performed least well for lakes Rotoiti (417% deviation), Rotoehu (207% deviation) and Okaro (138% deviation). However, even when excluding these three lakes still three of the models outperformed the Abell et al. model.

For predicted P loads, the Abell et al. 2019 model was the most divergent relative to the other models, especially at high P concentrations (Figure 16). Overall, the Abell et al. 2019 fitted the expected P loads least well, although this model was closest to the estimates for 2 lakes (Rotomahana and Rotorua). However, in the case of Lake Rotorua the fit between P load and concentration is not expected to be realistic because of the large amount of alum added to its inflows. Model D-R-K performed the best (53% deviation), closely followed by the R and N models. An average of these three models did not perform better than model D-R-K alone.

Model H-V fitted the expected N loads the least well, based on mean deviations. It usually estimated loads higher than the other two models. However, model H-V is heavily dependent on the choice of its coefficient a, in the equation for predicted retention. Harrison et al. (2009) provided a number of coefficients, depending on forms of N (two classes), lake size (two size classes), latitude, lakes versus reservoirs (but not cross sections of these classes, such as TN in large temperate lakes), and an overall mean. We chose the coefficient given for lakes (greatest sample size apart from the overall mean coefficient, which was higher). It is possible that with another coefficient the H-V model would have fitted better. For instance, with the coefficient for temperate lakes (5.13) the model would produce lower load estimates, although still usually too high.

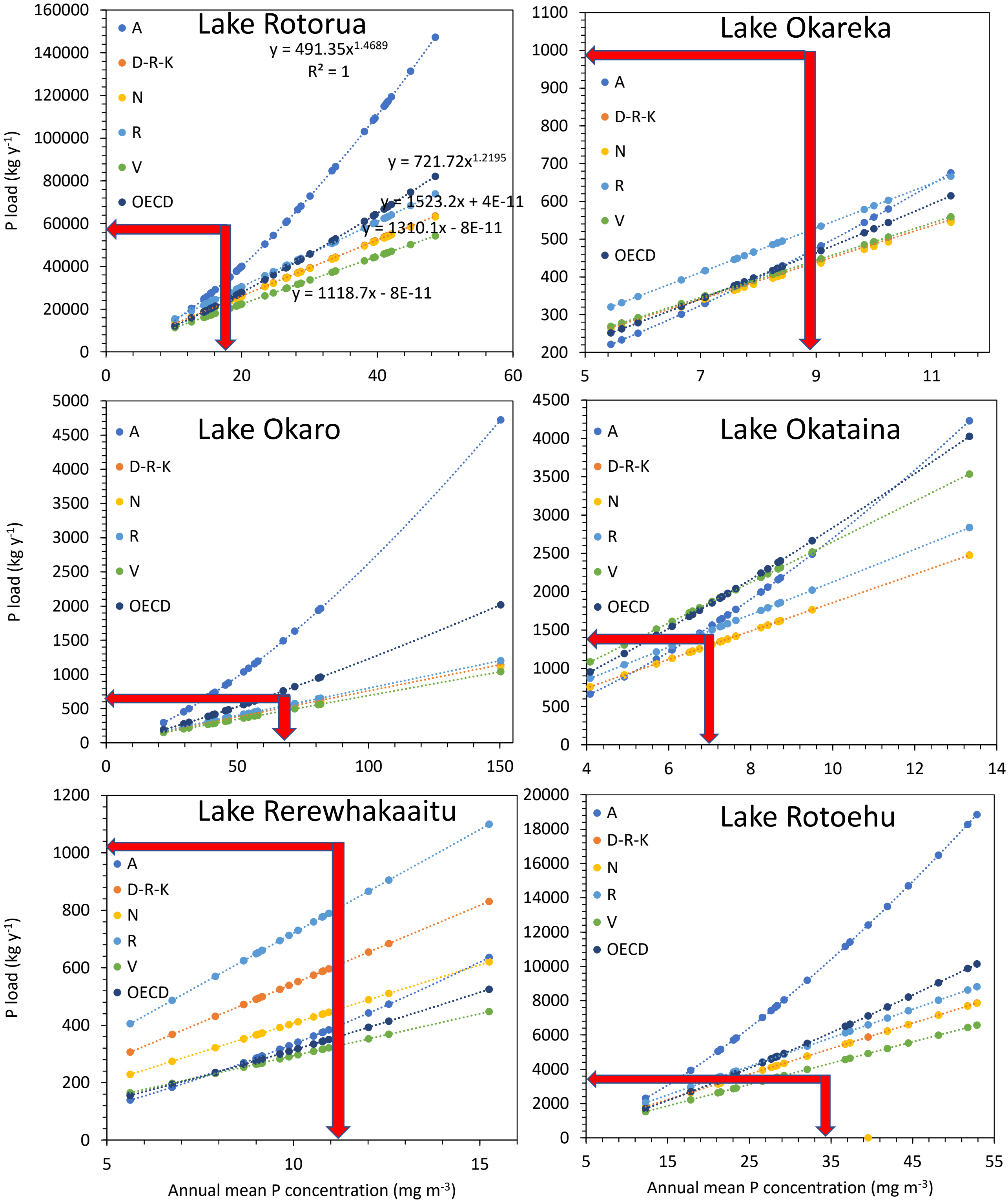


Figure 16: P loads as predicted from annual mean in-lake P concentrations (I). P loads as predicted from the annual mean P concentrations of Lake Rotorua (sites 2 and 5), Okareka (site 1), Okaro (site 1), Okataina (site 1), Rerewhakaaitu (site 1), Rotoehu (site 3), Rotoiti (site 3), Rotokakahi, Rotoma (site 1), Rotomahana (site 2), Tarawera (site 5), Tikitapu (site 1), in 2001-2020. n = 40 for each model for lake Rotorua, n = 20 or less for other lakes. Models are: A = Abell et al. (2019), D-R-K = Dillon and Rigler (1974), with R from Kirchner (1975), N = Nurnberg (1984), R = Reckhow (1979), V = Vollenweider (1976), OECD = Vollenweider (1982). Two regressions are power fits (A and OECD, the latter is indistinguishable from linear within the range of P concentrations in

the BoP lakes) and all others are linear. Regression equations only shown for Lake Rotorua. All R^2's are 1. The red arrows indicate the estimated load (McBride et al. 2021) on the y-axis and the average of the annual mean concentrations in 2016-2020 on the x-axis.

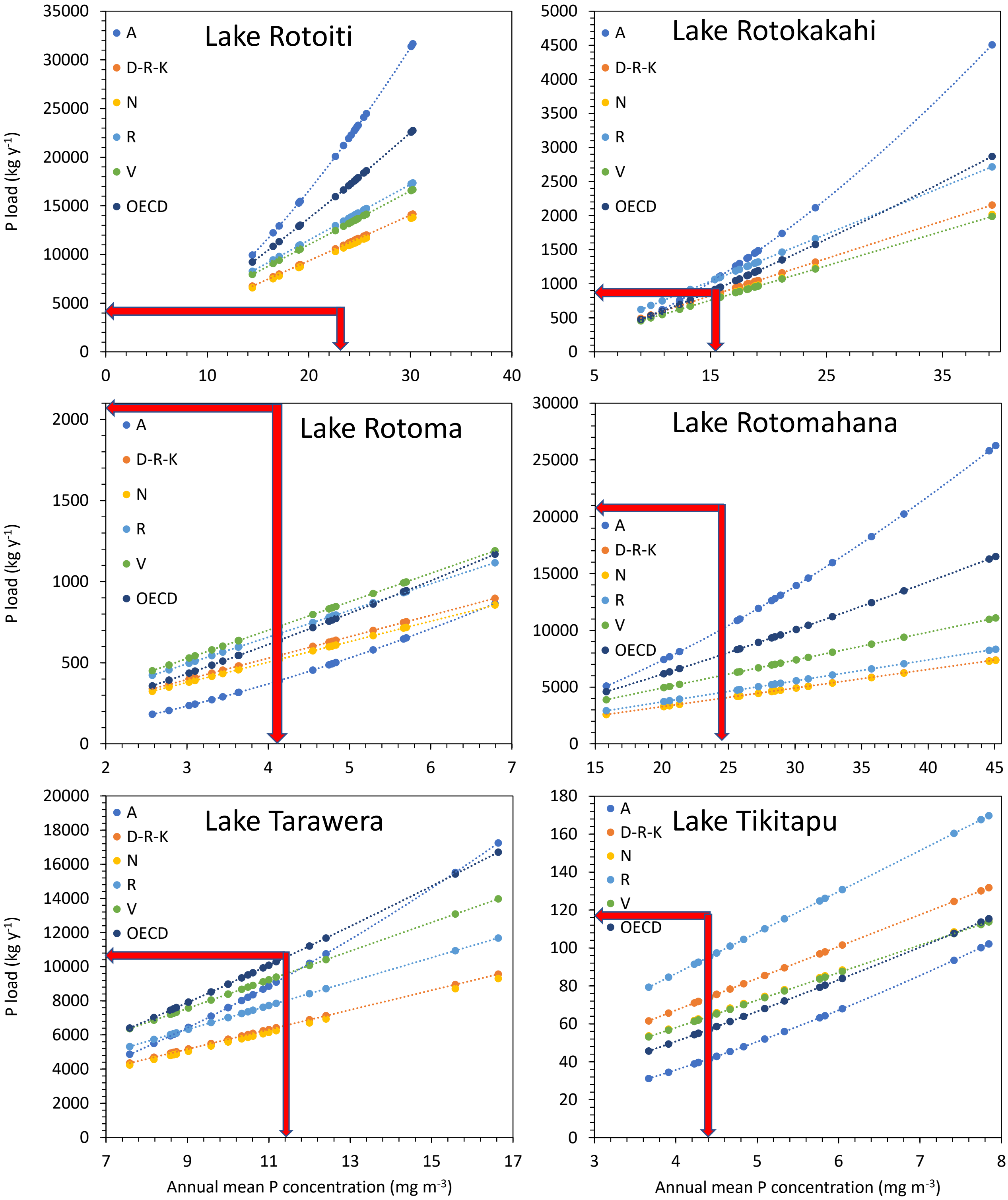


Figure 16: **Continued.**

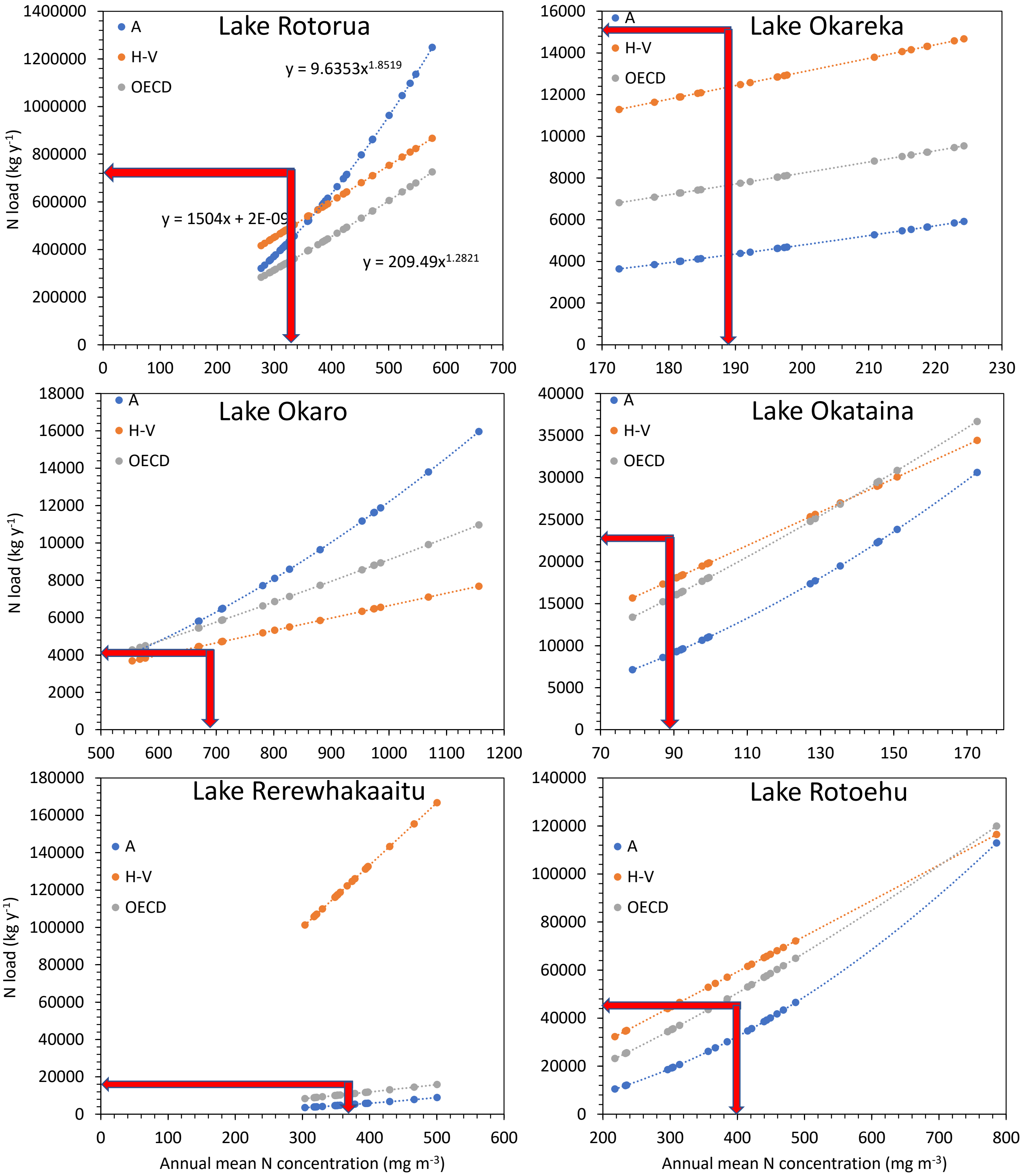


Figure 17: N loads as predicted from the annual mean in-lake N concentrations (I). N loads as predicted from the annual mean N concentrations in 2001-2020. Lake site as in previous Figure. n = 40 for each model for lake Rotorua, n = 20 or less for other lakes. Models are: A = Abell et al. (2019), H-V = Harrison et al. (2009) and Vollenweider (1976) combination, OECD = Vollenweider (1982). Two regressions are power fits (models A and OECD, the OECD model is almost linear) and the remaining one (H-V) is linear. Regression equations only shown for Lake Rotorua. All R2's are 1. The red arrows indicate the estimated load (McBride et al. 2021) on the y-axis and the average of the annual mean concentrations in 2016-2020 on the x-axis.

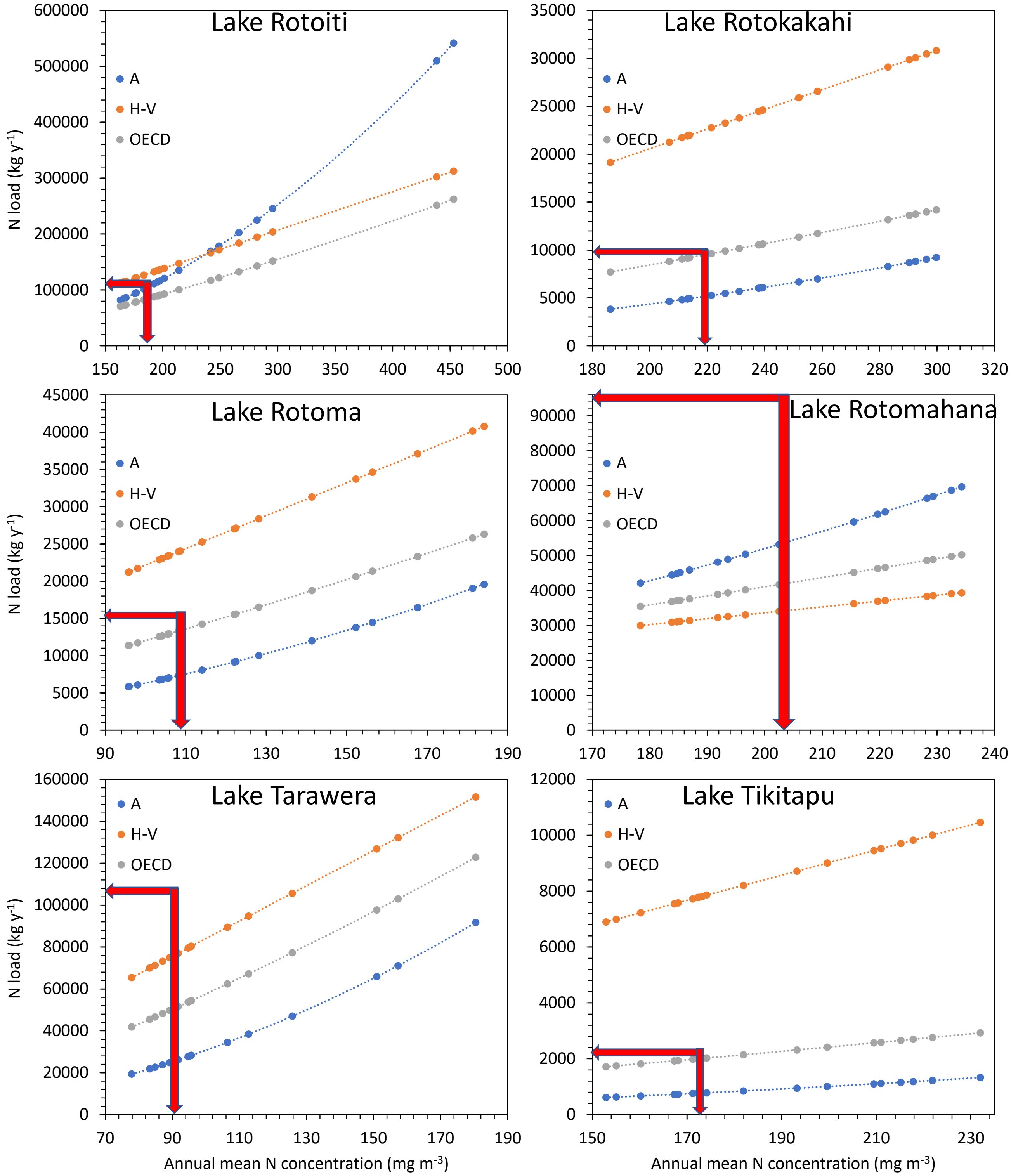


Figure 17: **Continued.**

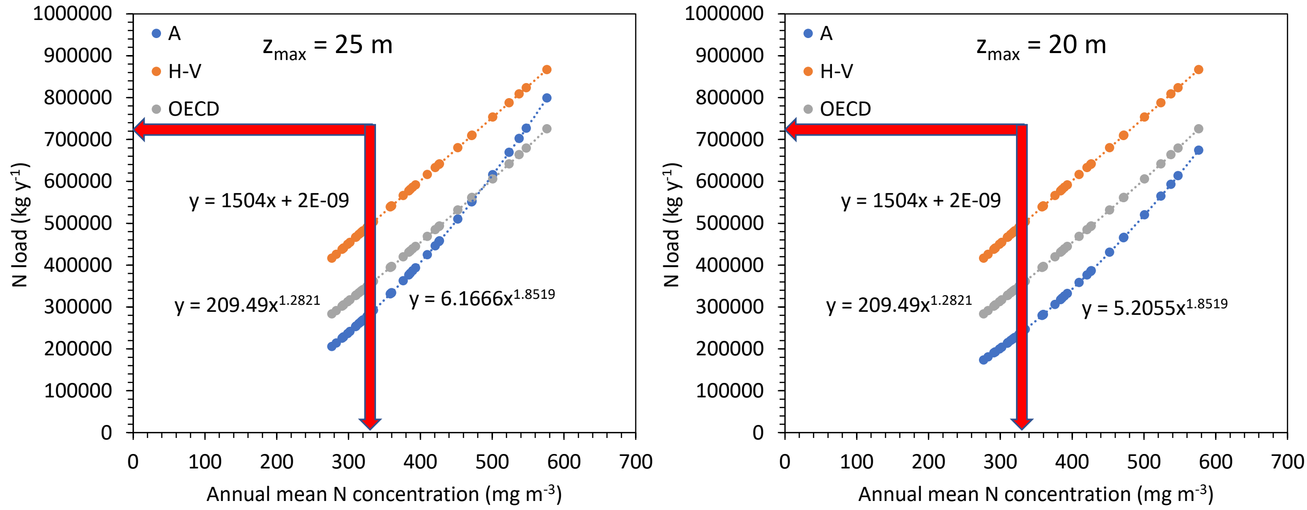


Figure 18: The effect of changing maximum depth in the data input of the Abell et al. (2019) model for nitrogen. Data of Lake Rotorua.

The model for nitrogen of Abell et al. 2019 is strongly affected by the value for the maximum depth, while the other equations either use mean depth (H-V) or are not dependent on depth (OECD). Figure 17 used the maximum depth = 45 m (Scholes and Hamill (2016) in the Abell et al. (2019) model for Lake Rotorua. Because very little of the area of lake Rotorua is deeper than 20 m (7% of lake area) to 25 m (4%), Figure 18 shows plots using z_{max} = 25 m and 20 m. For maximum depths less than 45 m the N load of Abell et al. 2019 reduces and becomes more similar to the other two models, especially at high N concentrations and loads. However, the load predicted for the 2016-2020 mean lake concentration becomes even less than the estimated current load compared with the model using z_{max} = 45 m (Figure 17). The summary plot Figure 19 does not take into consideration these alternative results (it applies the Abell et al. 2019 model as in Figure 17).

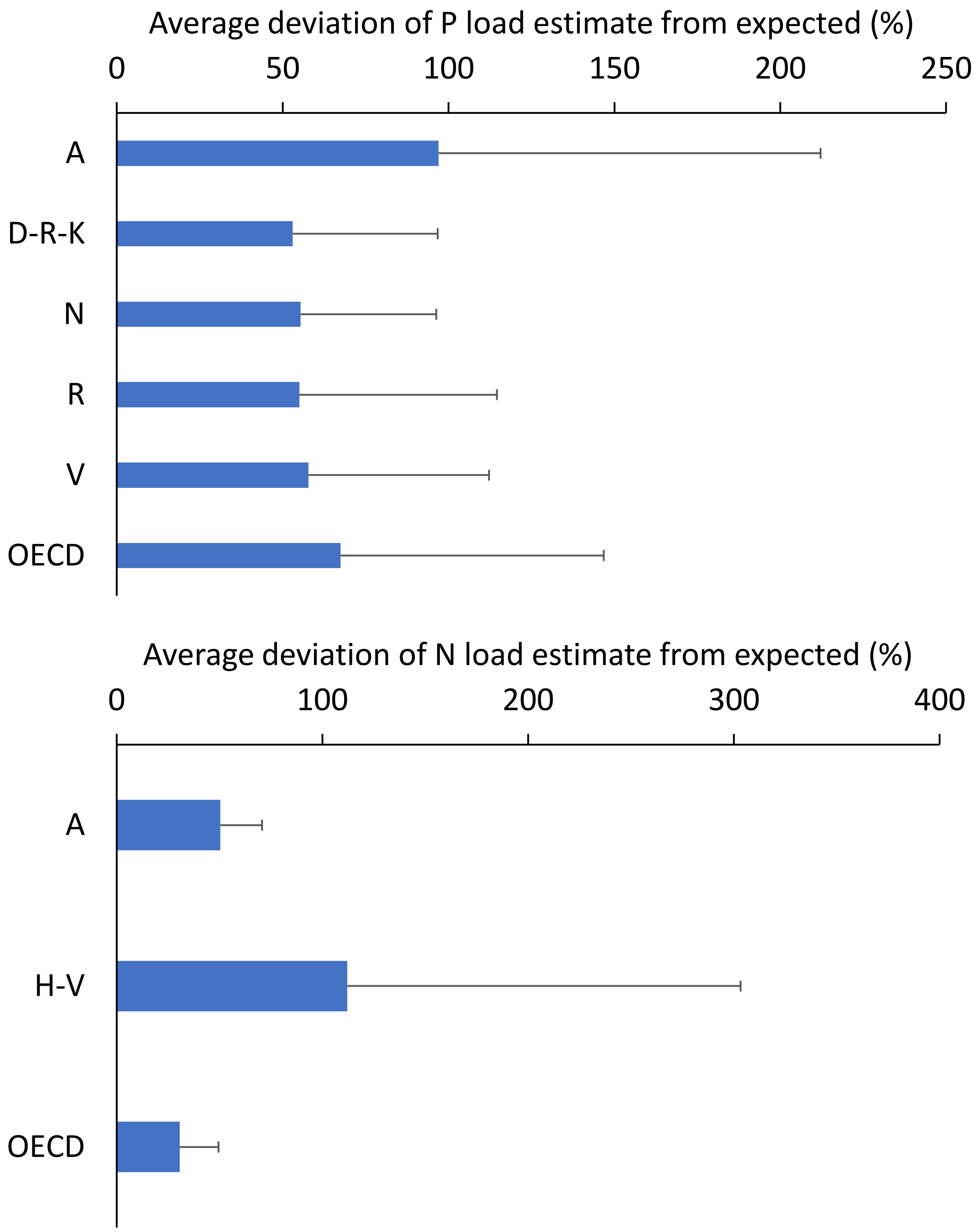


Figure 19: The average absolute deviations (%) of P and N load estimates from expected for each model (I). Based on the averages of N and P concentrations in 2016-2020, with expected loads taken from McBride et al. (2021). Error bars are standard deviations (n = 12 lakes). Abbreviations for models as in Figures 16 and 17. Absolute deviations are used because negative (smaller) and positive (larger) deviations would partially cancel.

In nine out of 12 lakes the estimated loads (McBride et al. 2021) were outside the range of estimates of the models for P (Figure 16). The absolute difference between the loads estimated by the models and the loads of McBride et al. (based on 2016-2020 mean water quality) was greater (average 64%, range 53-97%, Figure 19) than the desired reductions in the lake P concentrations (average 17%, range 0 – 65%).

Model A (Abell et al. 2019) fitted the expected N loads least well based on median deviations (Figure 19 shows mean deviations), although model A was closest to the estimates for 2 lakes, lakes Rotomahana and Rotoiti. However, in Lake Rotomahana all models including model A much underestimated the N load, and in lake Rotoiti the load estimates of the 3 models were close together at the average N concentration applied to the models. The OECD Model performed the best (31% mean absolute deviation). Averages of models in various combinations did not perform better than the OECD model alone.

Figures 20-22 include results with $[TP]_{in}$ and $[TN]_{in}$ calculated based on the estimated inflow rates (rather than estimated outflow rates). This applies only to the models of Abell et al. 2019, Larsen-Mercier (1976) and OECD (1982) for P, and Abell et al. (2019) and OECD (1982) for N. The remaining models in Figures 20-22 don't use inflow nutrient concentrations and are the same as in Figures 16-19. However, the Larsen-Mercier (1976) model estimated this way diverges from the Vollenweider (1976) model. Therefore, Figure 20 shows the Larsen-Mercier (1976) model instead of the Vollenweider (1976) model.

Figures 20-21 show, apart from the estimated load of McBride et al. (2021) also its range, as estimated by McBride et al. (2021) by varying the assumed attenuation of nutrients between the root zone/block boundary and lake edge.

The performance of the P models improved only slightly by 3% (average deviation from expected loads 61%, range 53-86%, Figure 22) while the average deviation of the three N models became greater by 7%. Each of the three P models that use $[TP]_{in}$ as part of their equation improved by using the estimated inflow rates. As a result, the average deviation from expected of the Abell et al. (2019) equation for P, the only one of the models developed from New Zealand lake data, came slightly closer to the results for the other models, but was still the least well-fitting model. The Abell et al. model (2019) for P performed least well for lakes Rotoiti (307% deviation), Rotoehu (156% deviation) and Okaro (125% deviation). However, even when excluding these three lakes, still three of the P models outperformed the Abell et al. model. While the Abell et al. model provided the best fit for P loads in lakes Rotomahana and Rotorua, those fits were still not very good (55% and 48% deviation, respectively). In eight out of 12 lakes Abell et al. (2019) provided either the lowest or the highest estimate of the P load of all models, for the average lake concentrations of 2016-2020.

Both of the two N models that use $[TN]_{in}$ as part of their equation (OECD 1982 and Abell et al. 2019) deteriorated by using the estimated inflow rates.

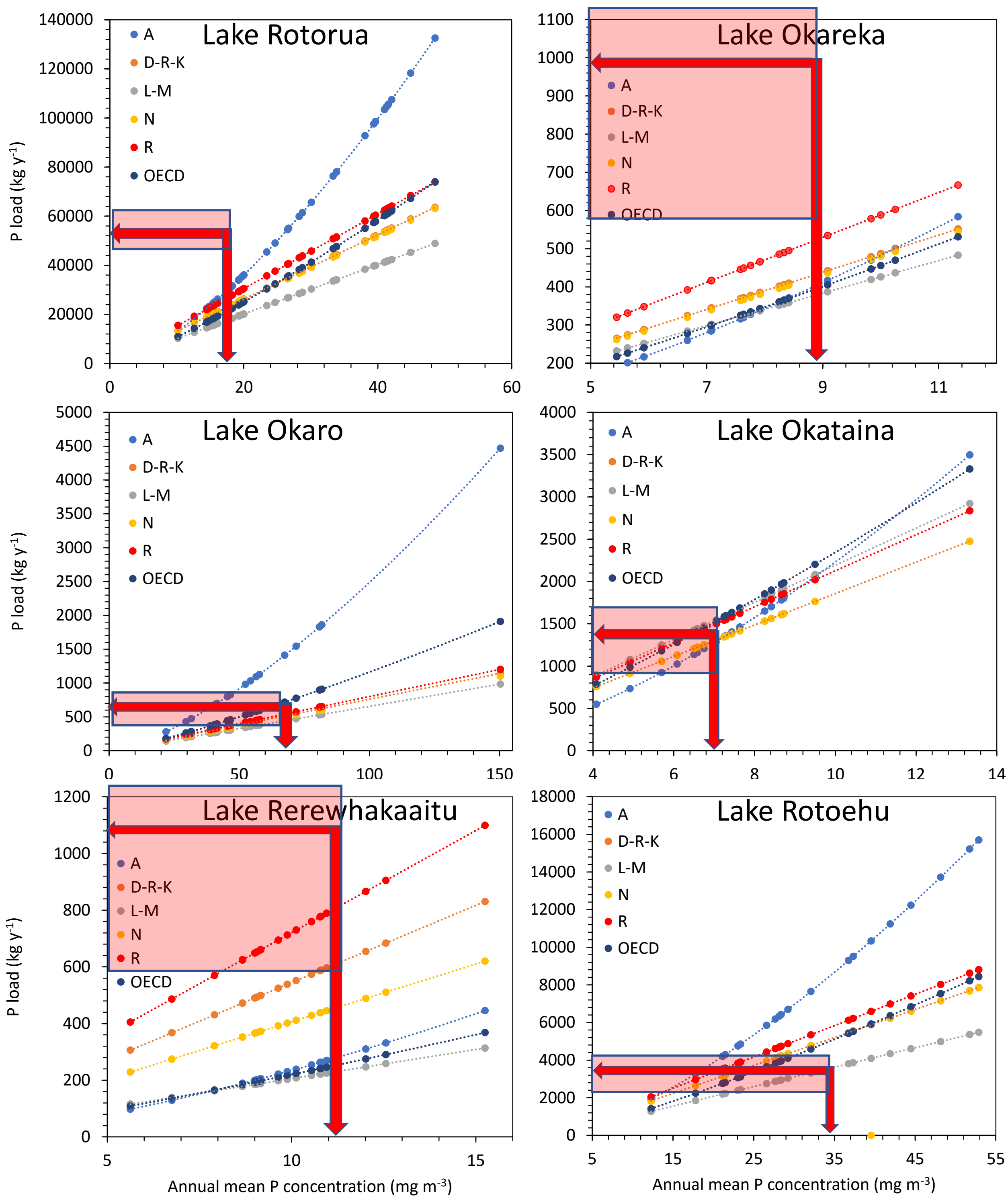


Figure 20: P loads as predicted from annual mean in-lake P concentrations (II) (using inflow flow rates). P loads as predicted from the annual mean P concentrations in 2001-2020. Models are: A = Abell et al. (2019), D-R-K = Dillon and Rigler (1974), with R from Kirchner (1975), L-M = Larsen-Mercier (1986) N = Nurnberg (1984), R = Reckhow (1979), OECD = Vollenweider (1982). The red arrows indicate the estimated load on the y-axis with red transparent blocks indicating its range, and the average of the annual mean concentrations in 2016-2020 on the x-axis. The difference with Figure 16 is in three models (A, L-M and OECD) where inflow and outflow rates are not assumed equal, as opposed to Figure 16. The remaining models are unchanged.

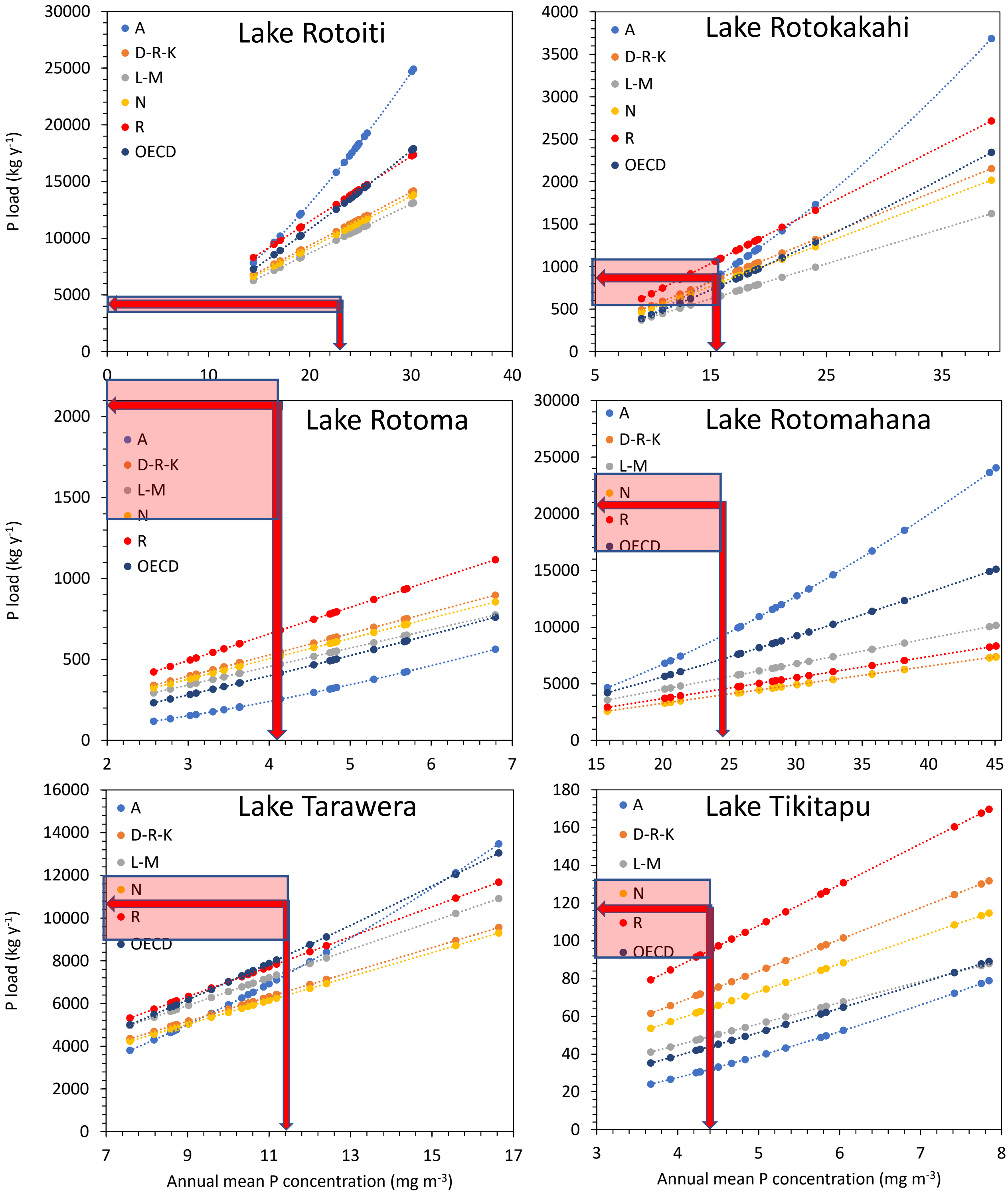


Figure 20: **Continued.**

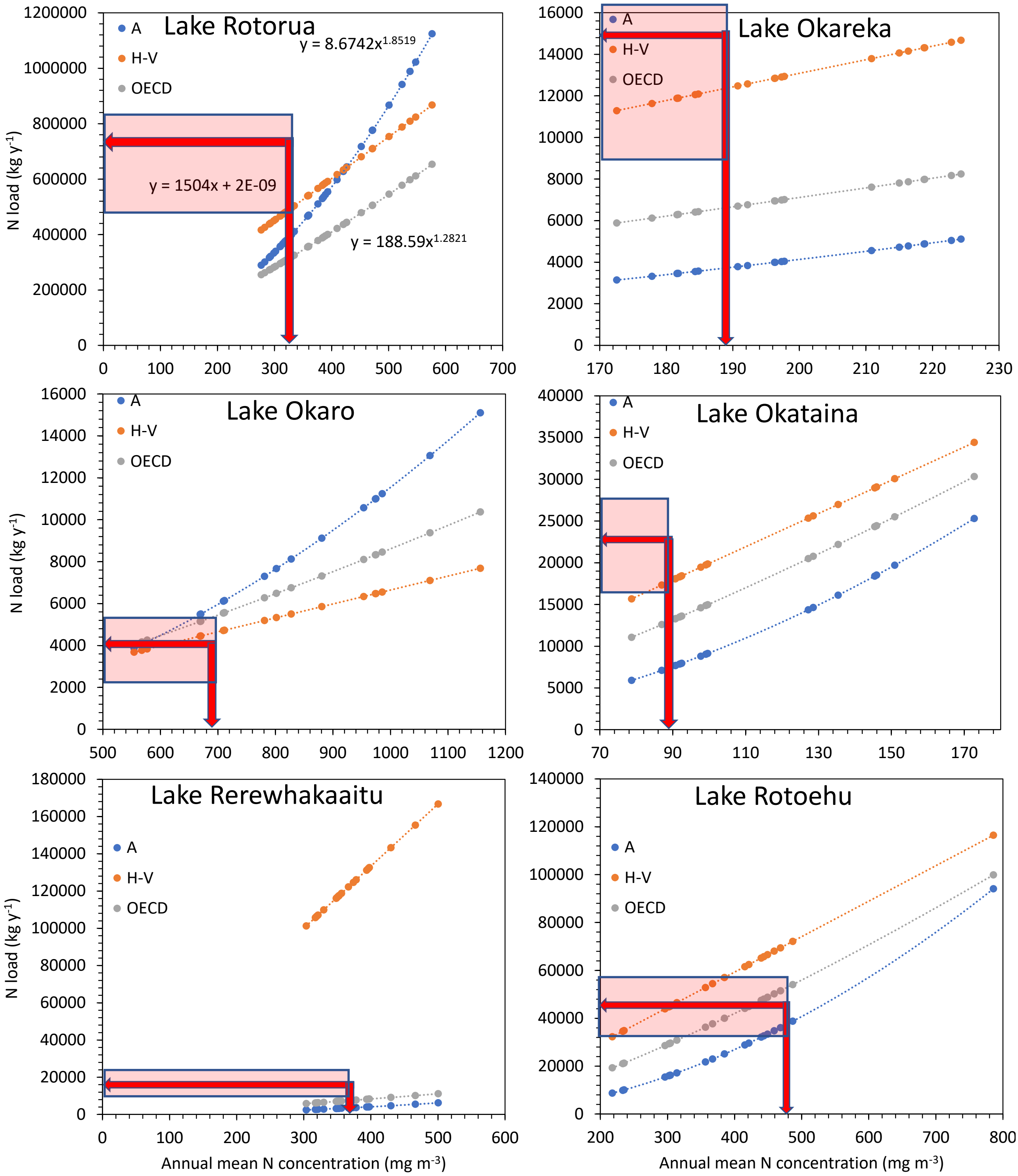


Figure 21: N loads as predicted from the annual mean in-lake N concentrations in 2001-2020 (II) (using inflow flow rates). Models are: A = Abell et al. (2019), H-V = Harrison et al. (2009) and Vollenweider (1976) combination, OECD = OECD (1982). The difference with Figure 17 is in two models (A and OECD) where inflow and outflow rates are not assumed equal, as opposed to Figure 17. Model H-V is unchanged. Two regressions are power fits (models A and OECD, the OECD model is almost linear) and the remaining model (H-V) is linear. n = 40 for each model for lake Rotorua, n = 20 or less for other lakes. Regression equations only shown for Lake Rotorua. All R2's are 1. The red arrows indicate the estimated load on the y-axis with red transparent blocks indicating its range, and the average of the annual mean concentrations in 2016-2020 on the x-axis.

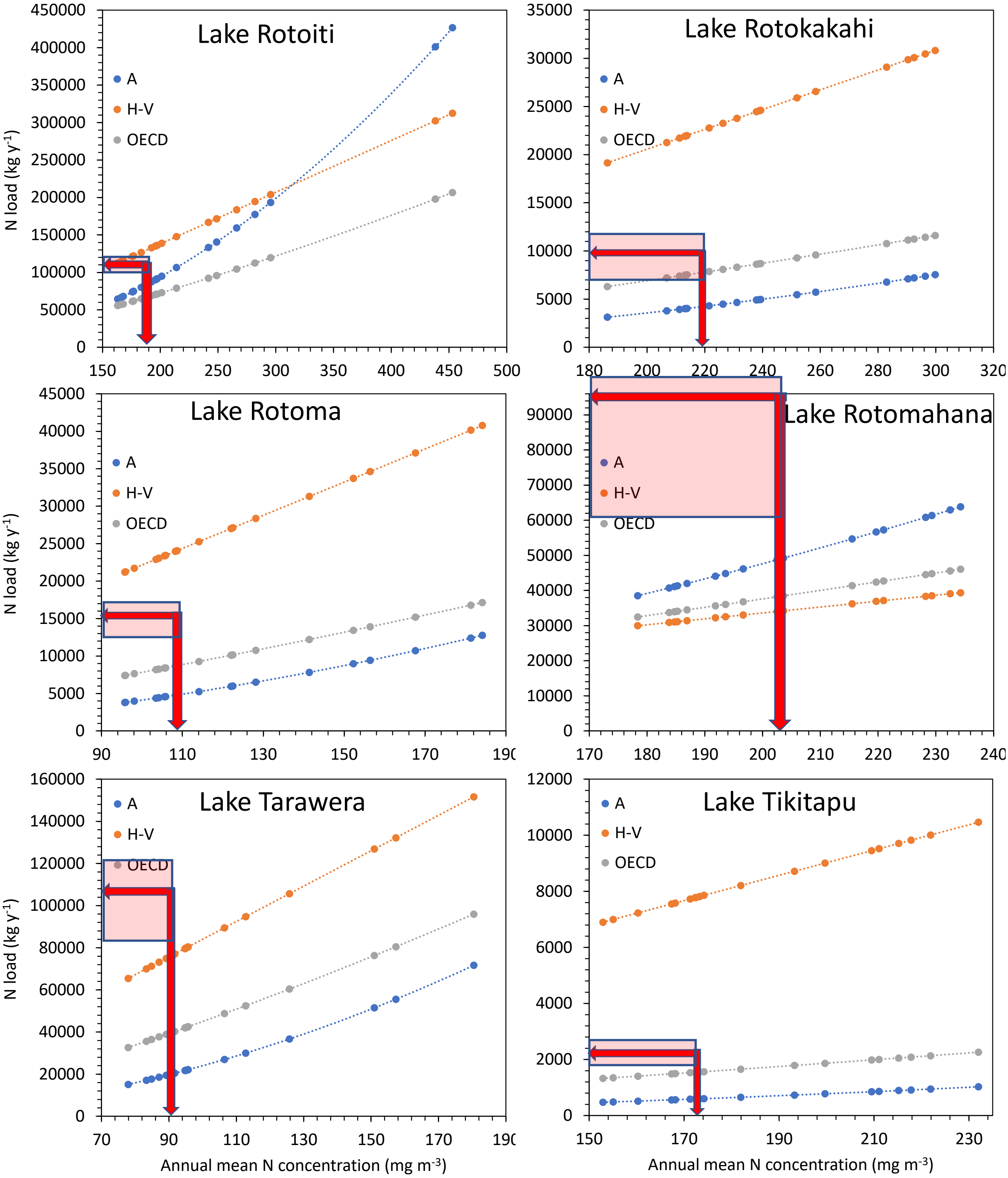


Figure 21: **Continued.**

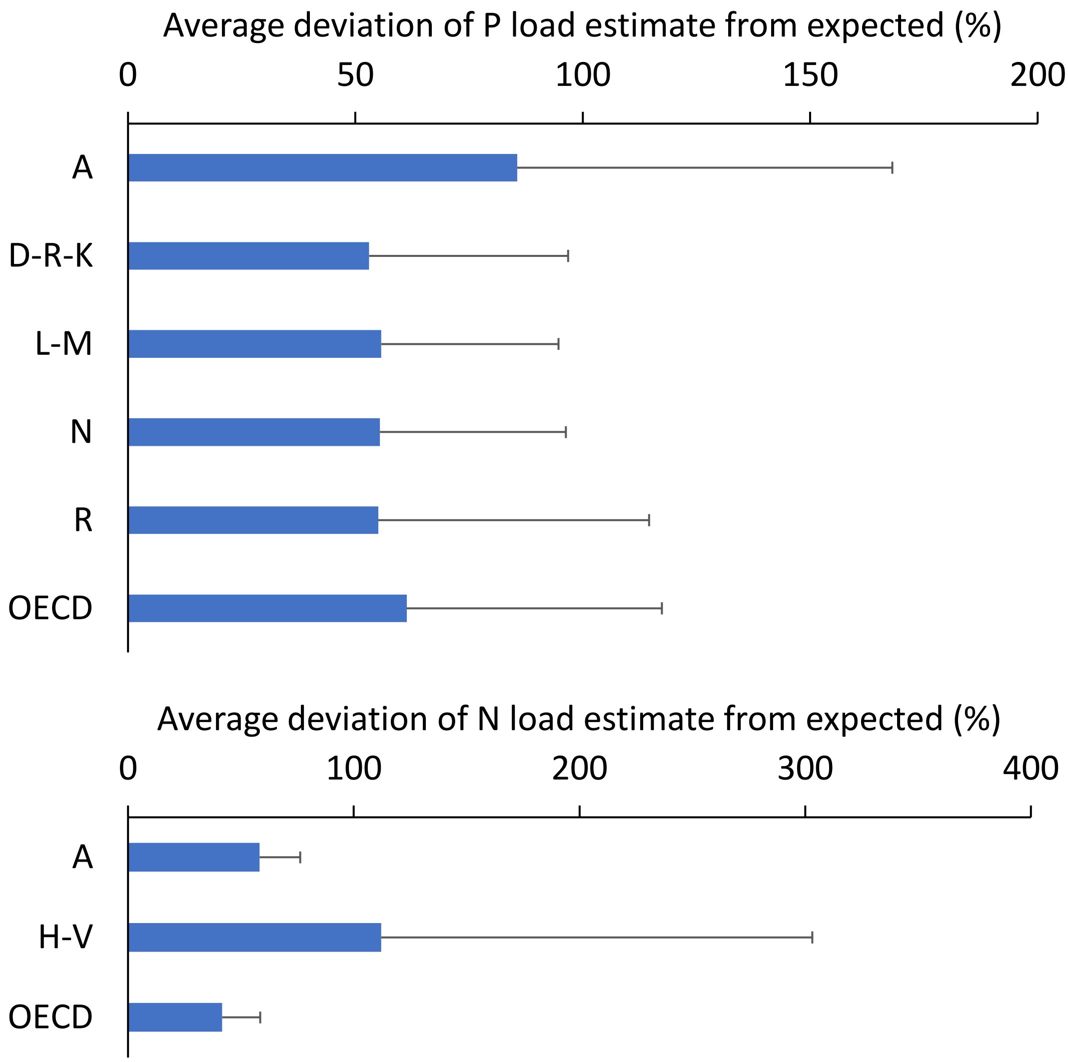


Figure 22: The average absolute deviations (%) of P and N load estimates from expected for each model (II) (using inflow flow rates). Based on the averages of N and P concentrations in 2016-2020, with expected loads taken from McBride et al. (2021). Error bars are standard deviations (n = 12 lakes). Abbreviations for models as in Figure 21. The difference with Figure 19 is in five models (A, L-M and OECD for P, and A and OECD for N) where inflow and outflow rates are not assumed equal, as opposed to Figure 19. The remaining models are unchanged.

3.5.1 Reference state model

In addition to “current state” nutrient mass balance models (those in section 2.1) Abell et al. (2019) also developed “reference state” models. The latter was derived from relatively pristine lakes with low loads and low lake nutrient concentrations. Abell et al. (2019) applied the reference state models to their full lake data set, showing that, as expected, reference state nutrient concentrations as derived from the reference state models are lower than current state concentrations derived from current state equations. Therefore, in lakes where lake nutrient concentrations increased from their reference state to their current state the nutrient loads have increased.

The shape of the reference state equations for P and N was the same as of the current state equations (section 2.1), they only differed in their parameters. Four of the Rotorua Te Arawa lakes (Rotoma, Tarawera, Rotoiti, Okataina) were included in the data set to train model reference conditions for P and six Rotorua Te Arawa lakes (Rotoma, Tarawera, Rotoiti, Okataina, Okareka, Tikitapu) were included in the data set to train model reference conditions for N. Therefore, it was suggested that the reference state equations should be used for these lakes instead of the current state equations (personal communication Deniz Ozkundakci). A comparison follows of the results between the current state and reference models for these subsets of lakes.

Reference state equations produce in general lower P and N loads for the same in-lake concentrations, or, vice versa, reference state equations result in higher in-lake nutrient concentrations for the same loads. For any particular lake the difference between current state and reference models increases with increasing loads and lake concentrations (Figure 23). In all cases but one (P in Lake Rotoiti) the results using the reference state equations diverge more from expected than results using the current state equations. Using the reference state equations for P, the agreement with the measured load was worse in three of the four lakes, compared with using the current state equations, and using the reference state equations for N the agreement with expected was worse in all six lakes. Phosphorus in Lake Rotoiti was the lone exception because there the current state model overestimated the load (as opposed to underestimation in most lakes), and because the reference state model produces, as mentioned, lower P loads for the same in-lake concentrations the result was less overestimation of the load.

Therefore, the divergence from expected results, already relatively high for the Abell et al. (2019) model (see Figure 22), would be higher if reference state models would be used for the relevant lakes. Comparisons of predicted N and P loads derived with the current state equations and with the reference state equations are shown in Figure 23, with Lake Tarawera as an example.

Comparing the current state equation for TP with the reference state equation shows that TP lake concentrations predicted with the reference equation are greater than predicted by the current state equation for lakes where $T_W < 21.16$ years. None of the Rotorua Te Arawa lakes reach such a high residence time. Therefore, under all tested conditions (annual mean P concentrations for each year of 2001-2020, Figures 16-21) the inverted reference equation produces loads lower than the current state equation does.

TN lake concentrations predicted with the reference equation are equal to those predicted by the current state equation for lakes where $[TN]_{in} = 426.22 * z_{max}^{-0.30}$ mg m^{-3}. On the other hand, when $[TN]_{in} < 426.22 * z_{max}^{-0.30}$ mg m^{-3}, for the same load the N concentration is lower according to the reference state equation than according to the current state equation. TN lake concentrations predicted with the reference state equation are higher than predicted by the current state equation for lakes where $[TN]_{in} > 426.22 * z_{max}^{-0.30}$ mg m^{-3}, for the same TN load.

$[TN]_{in}$ was almost always more than $426.22 * z_{max}^{-0.30}$ mg m^{-3}, across the six reference state lakes in the 20-year period of 2001-2020 (n = about 120). Therefore, in most cases, using the reference state equation results in lower N loads for the same lake concentrations. Or, vice versa for the same N loads reference state equations result in higher N lake concentrations.

The reference state models are not intended to be used in this way with current loads and current inflow concentrations. They are intended to be used with estimates of loads based on TN and TP concentrations in lake inflows corresponding to a reference state derived by McDowell et al. (2013), producing reference in-lake nutrient concentrations accounting for climate, topography and geology. McDowell et al. (2013) modelled the relationship between stream nutrient concentrations and the proportion of intensively-farmed land in a catchment. Based on these models, McDowell et al. (2013) calculated reference concentrations that corresponded to concentrations under a scenario of no intensively-farmed land (Abell et al. 2019). Therefore, the reference state equations do not reflect the current state, including in lakes that were used to train the reference state models. Nevertheless, it does seem curious that, when applying the same loads, the reference state equations usually produce a substantially higher estimate of in-lake concentrations than the current state equations, even in the relatively pristine lakes that were used to train the reference state model. This was probably not intentional by the authors and may not have been understood. These lakes have little pasture in their catchments (for instance, 17% in Lake Tarawera, 2% in Lake Tikitapu, Scholes and Hamill 2016). The reference state equations are intended to be used with loads that exclude the nutrient yields from such pasture. Nevertheless, with the low pasture cover and using actual loads in the model these lakes would be expected to be close to their reference state.

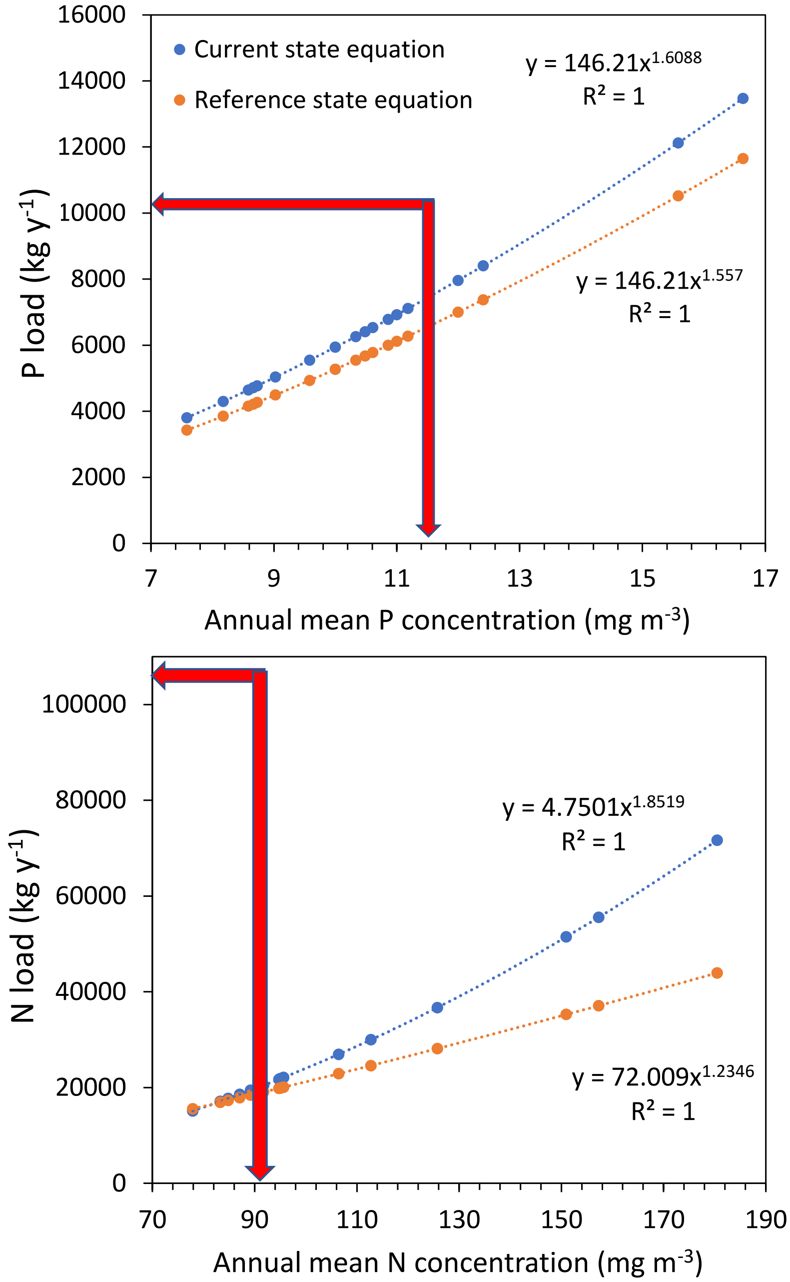


Figure 23: Comparison of reference state models with current state models. Models of Abell et al. (2019). Both panels are for Lake Tarawera, left panel for P and on the right for N. In all cases, including in the other lakes used to train the reference models, the reference state equation generally predicts lower N and P loads for the same lake concentrations, and, vice versa, higher N and P lake concentrations for the same loads as the current state equations.

3.6 Change in N:P ratios from external load to in-lake concentrations

The average load TN:TP ratio (by mass, not molar) was 9 and the average lake TN:TP ratio was 13. There was little correlation between average TN:TP ratios in the lakes and in the loads (R^2 = 0.03). Only lakes Tarawera, Rotoiti, Rotoehu and Okataina showed a decrease in N relative to P (as shown by TN:TP ratios decreasing from inflow loads to in-lake concentrations, Figure 24). This may suggest preferential retention of N compared with P in these four lakes., that is, relatively more loss of N (e.g., by denitrification and by burial) than P (by burial).

Worldwide almost 90% of lakes show preferential retention of P, with the greater retention of P over N leading to elevated TN:TP ratios in lake outflows (Wu et al. 2022, Verburg et al. 2013). Therefore, four out of 12 lakes (33%) with preferential retention of N is a relatively high proportion.

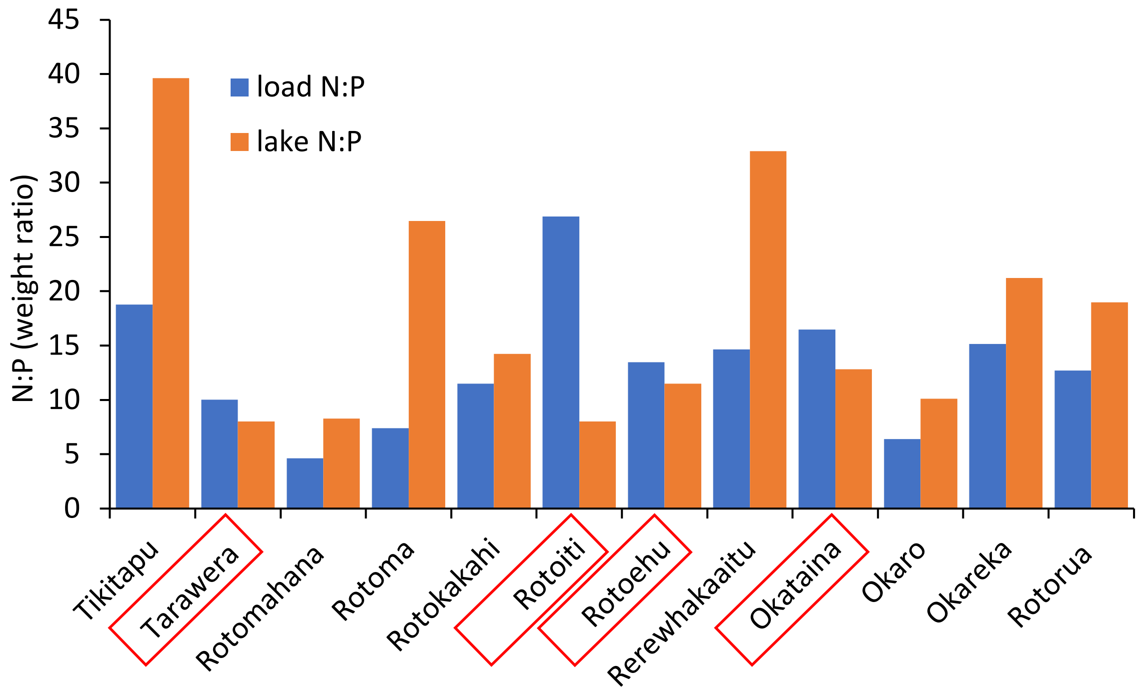


Figure 24: Comparison of N:P ratios between loads and lake concentrations. Lakes where the N : P ratio decreased from inflow to in-lake concentrations indicated by red rectangles.

3.7 Relationships between external loads and lake concentrations

Areal external N loads were highest in Lake Okaro, followed by Rotomahana and Rotorua (Figure 25). Areal external P loads were highest in Lake Rotomahana, followed by Okaro and Rotorua. Volumetric N and P loads were both highest in Lake Okaro, followed by Rotorua and Rotoehu (Figure 25).

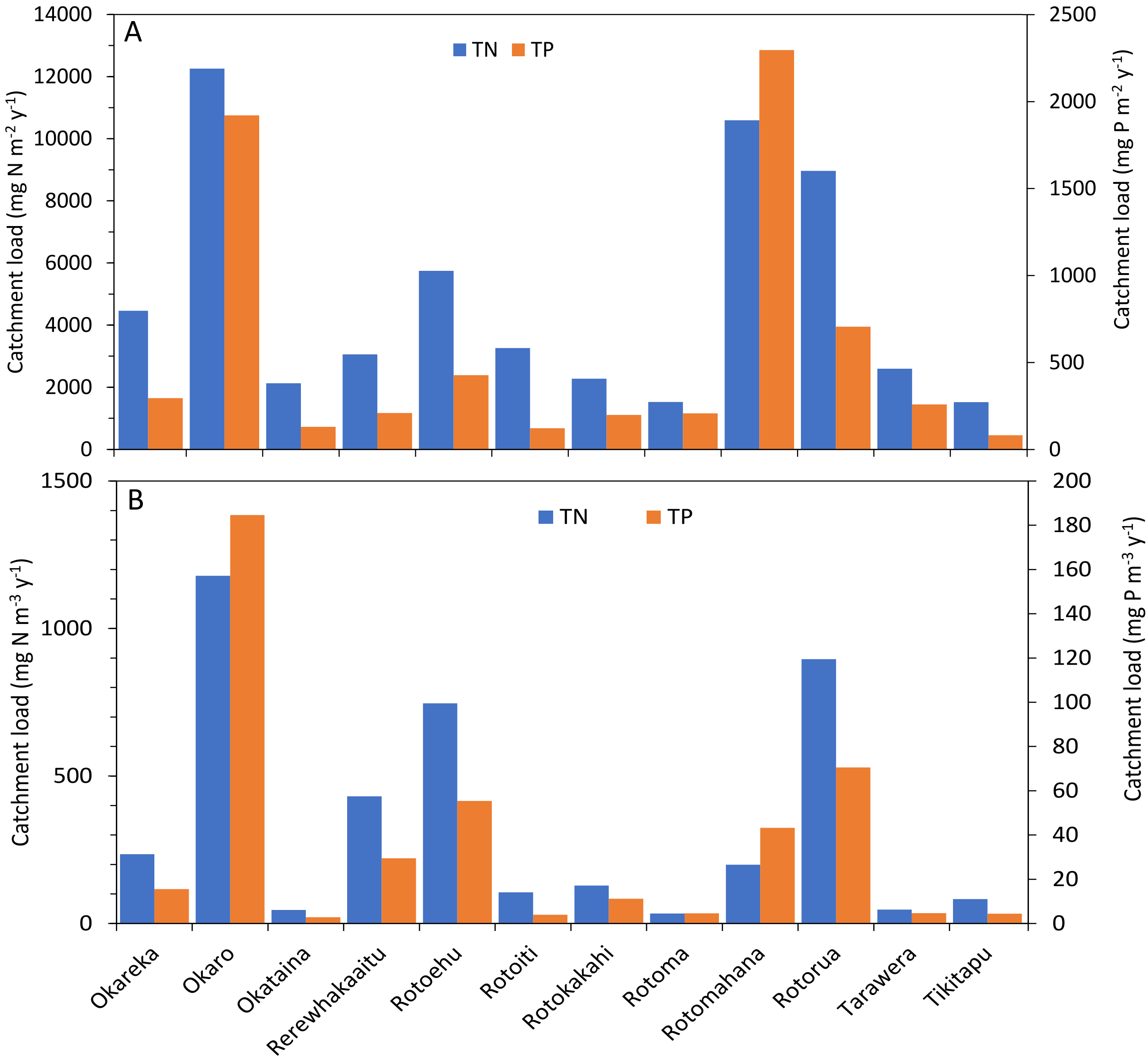


Figure 25: **Catchment nutrient loads.** A. Areal loads (per m^{-2}). B. Volumetric loads (per m^{-3}).

Average lake P concentrations fitted markedly better with total load divided by lake volume (R^2 = 0.83, Figure 26) than with areal load (R^2 = 0.46), or with the flushing-corrected formulations $[TP]_{in}/(1-\sqrt{T_W})$ as used in OECD 1982 (R^2 = 0.48), or $(L_P/q_s)/(1-\sqrt{T_W})$ as used in Vollenweider 1976 (R^2 = 0.53, note that in the latter, L_P/q_s, is equal to the total load divided by the outflow rate). These are linear correlations, power relations (as used by OECD 1982) fitted worse. A power fit of average lake P concentrations against $[TP]_{in}/(1-\sqrt{T_W})$ produced the equation y = $2.11X^{0.58}$ with a low correlation (R^2 = 0.39), compared with $1.55X^{0.82}$ in OECD 1982 (R^2 = 0.86, note this is the OECD equation in methods section 2.1). It is not suggested to apply this revised Vollenweider equation, modified to fit Rotorua Te Arawa lakes, to predict P load or lake concentrations in New Zealand lakes, because it is derived from too few lakes and the R^2 is low. Removing three lakes with lake P concentrations most above the fit (Rotoiti, Rotoehu, Okaro) did not do much to improve the fit (R^2 = 0.55). Note that almost all lakes are below the 1:1 line around which the lakes are expected to cluster (OECD 1982). Internal loading could explain data above the 1:1 line but not below.

Either retention is underestimated (section 3.12) or inaccuracies in the input data are responsible for data substantially below the 1:1 line.

Average lake N concentrations fitted better with total load divided by lake volume (R^2 = 0.86, Figure 26) than with areal load (R^2 = 0.50), with $[TN]_{in}/(1-\sqrt{T_W})$ as in OECD 1982 (R^2 = 0.57), or with $(L_N/q_s)/(1-\sqrt{T_W})$ (R^2 = 0.64, in the latter L_N/q_s is equal to the total load divided by the outflow rate). These are linear correlations, power relations (as used by OECD 1982) fitted slightly better. A power fit of average lake N concentrations against $[TN]_{in}/(1-\sqrt{T_W})$ produced the equation $y = 1.86X^{0.83}$ (R^2 = 0.78), compared with $5.34X^{0.78}$ in OECD 1982, with similar correlation (R^2 = 0.85, note this is the OECD equation in methods section 2.1). Note that almost all lakes are below the 1:1 line around which the lakes are expected to cluster (OECD 1982).

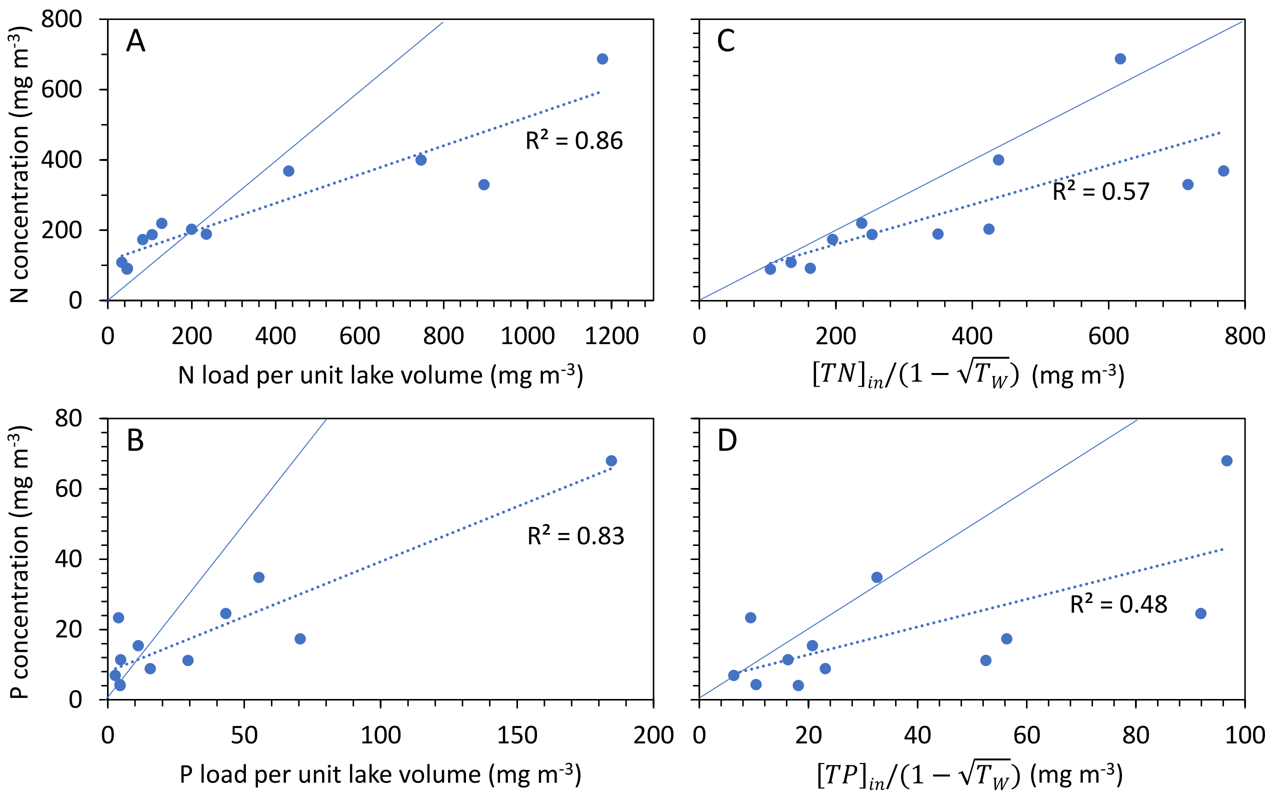


Figure 26: Comparing predictors of lake nutrient concentrations. (A, B) Lake nutrient concentrations against load per unit lake volume. (C, D) Lake nutrient concentrations against estimators of lake nutrient concentrations as used by OECD (1982) and others, i.e., flushing corrected annual average inflow concentrations. Inflow nutrient concentrations were estimated as total load divided by the estimated total inflow rate. R^2 values of linear regressions are included. Interrupted lines are linear regressions. The straight continuous lines are 1:1. The lake farthest away from the 1:1 line to the right in B) is Okaro. The lakes farthest to the right in D) are Rotomahana and Okaro.

3.8 Seasonality in water quality

Because one of the lake attributes is maximum chlorophyll *a*, it is of interest to examine the maxima, and in particular when the maxima occur and what drives seasonality. The monthly means of nutrients, water clarity, chlorophyll *a*, and TN:TP ratios and the month with maximum concentrations, are shown in Figures 27-32.

TP was highest in winter in nine of the 12 lakes, the exceptions being lakes Rotorua, Rotoehu and Rerewhakaaitu. These three lakes mix frequently down to the bottom, agreeing with their stratification potential being lowest of the 12 lakes (Figure 33), according to the index of Osgood (1988). The stratification potential index was defined by Osgood (1988) by the ratio of mean depth (m) to the square root of lake area (km^2) with the latter being considered indicative of fetch (Davies-Colley 1988). This ratio describes the transition along the polymictic (low values) to monomictic (high values –stratifying throughout summer) continuum. In the Rotorua Te Arawa lakes mean TP, TN and chlorophyll *a* in summer (January – March) generally decreased with increasing stratification potential, as Osgood (1988) had shown for a large number of lakes. However, exceptions were lakes Okaro and Rotomahana which had both relatively high mean summer nutrient and chlorophyll *a* concentrations and high stratification potential.

Only four lakes have their chlorophyll peaks in the summer: lakes Rerewhakaaitu, Rotoehu, Rotoiti and Rotorua. Apart from Lake Rotoiti these lakes have relatively shallow mean depths (note there may have been a change in seasonality in Lake Rotoiti after the construction of the diversion wall [Hamilton et al. 2010]– this was not explored). All other lakes are deeper. The average of the mean depths of the four lakes is 13.9 m and the average for the lakes with winter peaks (all others except Okaro where chlorophyll *a* peaks in spring) is 36.5 m. The lake stratification potential ($z/A^{0.5}$) of Osgood (1988) agrees even better than mean depth with the month of maximum chlorophyll *a* (R^2 = 0.90, $p < 0.000005$) (Figure 34).

In the majority of lakes which have wintertime chlorophyll maxima the phytoplankton is unlikely to be dominated by cyanobacteria during the biomass peak, because cyanobacteria are not adapted to the full mixing and cooler conditions that occur during winter. If cyanobacteria are dominant at any time in these lakes it is more likely to be during the summer and autumn season when total phytoplankton biomass is lower. However, information of seasonality of phytoplankton species distributions is not available.

Lakes where the chlorophyll *a* maximum occurs in summer tend to have higher maximum algal biomass. Phytoplankton growth during summer in these lakes is less likely to be P limited than in those with a winter peak in biomass (see below). The average log transformed chlorophyll *a* in the month with highest chlorophyll (see Figure 32), after excluding Okaro and Rotomahana (see above), correlated significantly and negatively with month of highest chlorophyll *a* (R^2 = 0.67, $p < 0.005$) and with Osgood's stratification potential (R^2 = 0.57, $p < 0.05$).

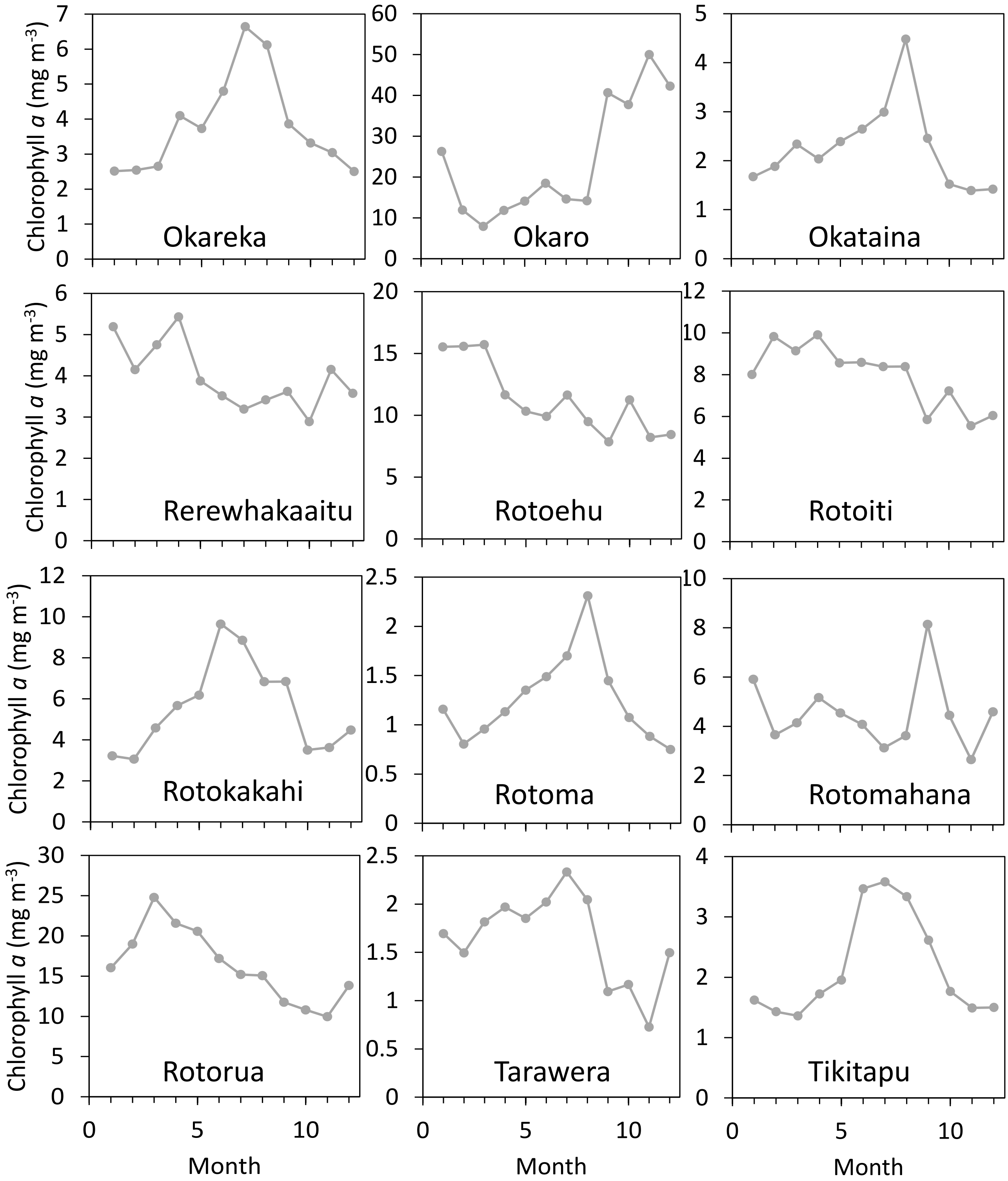


Figure 27: **Monthly mean chlorophyll *a*.** Averaged for 2001-2020, when data available. Sites: Rotorua 2, Rotoiti 3, Rotomahana 2, Tarawera 5, Rotoehu 3, site 1 for the remainder.

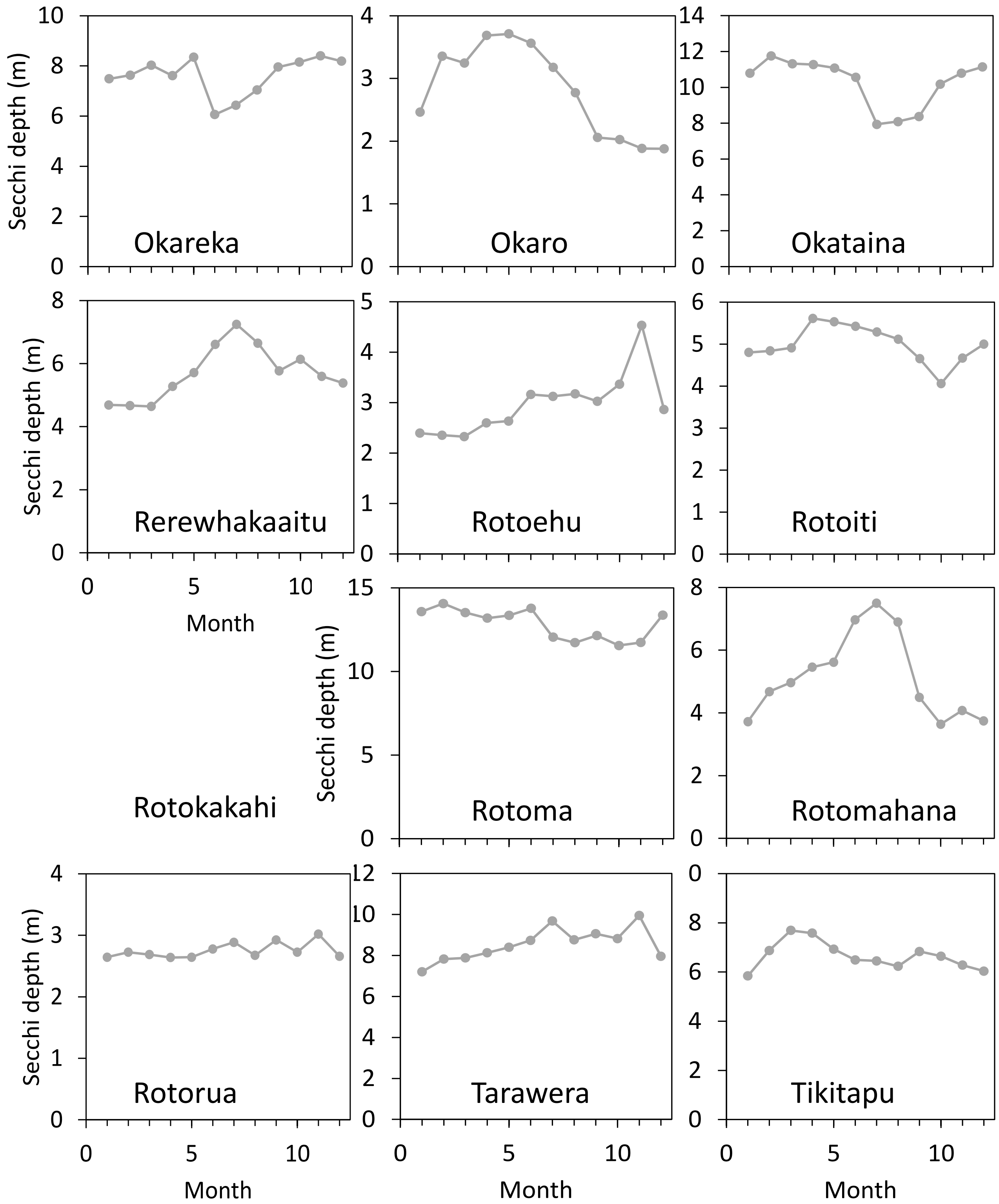


Figure 28: Monthly mean Secchi depth, averaged for 2001-2020. Secchi depth is not recorded in Lake Rotokakahi.

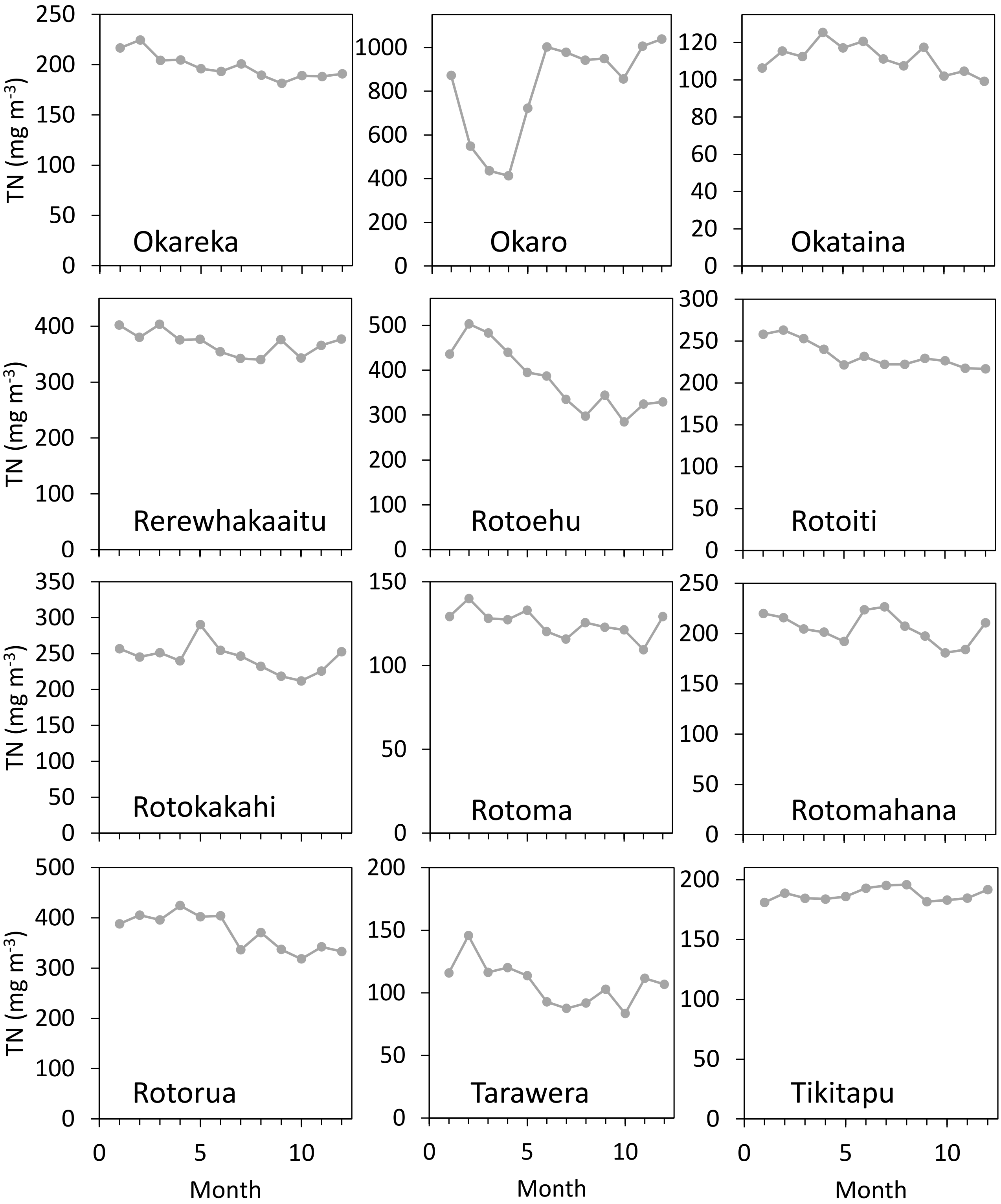


Figure 29: **Monthly mean TN, averaged for 2001-2020.**

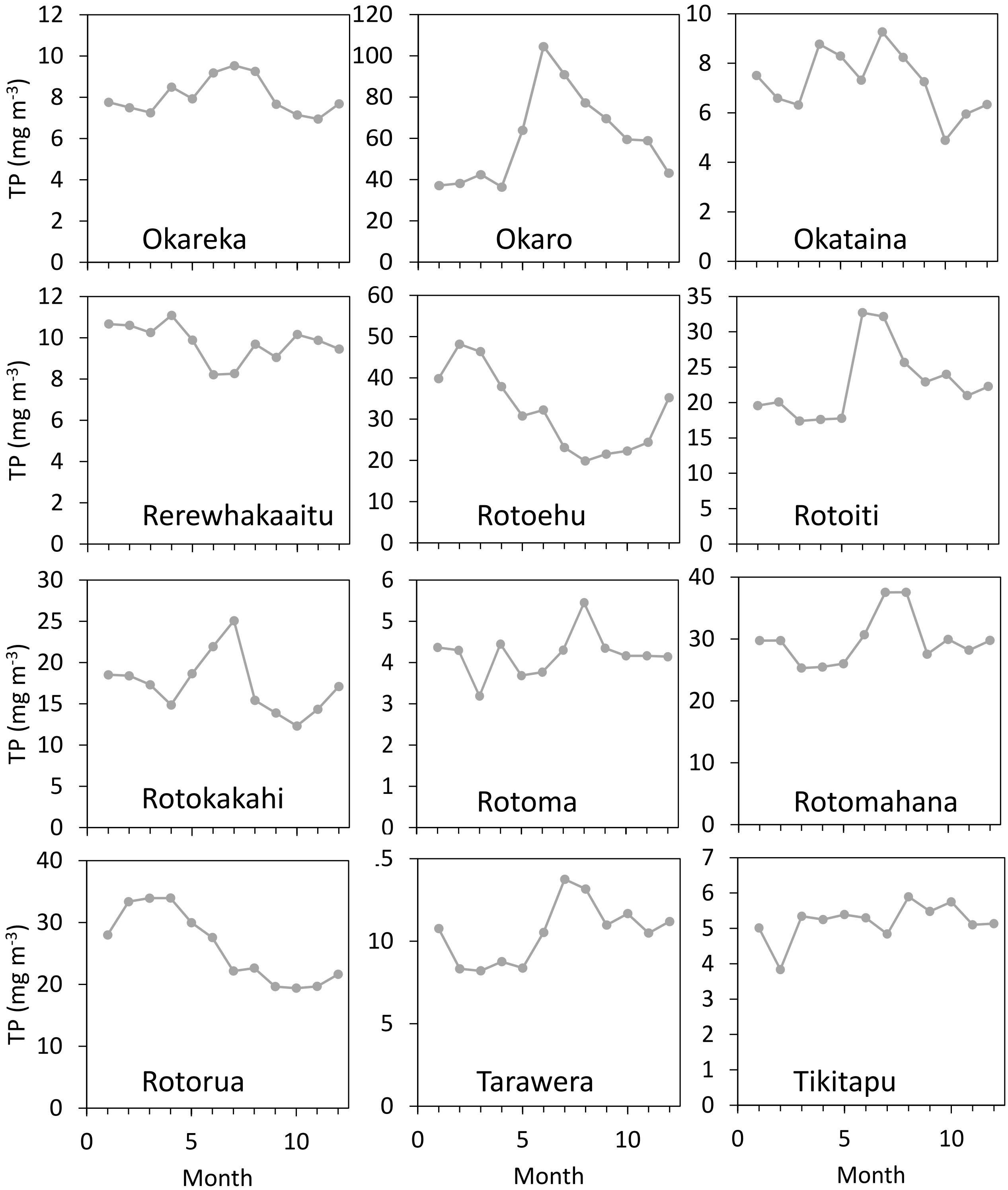


Figure 30: **Monthly mean TP, averaged for 2001-2020.**

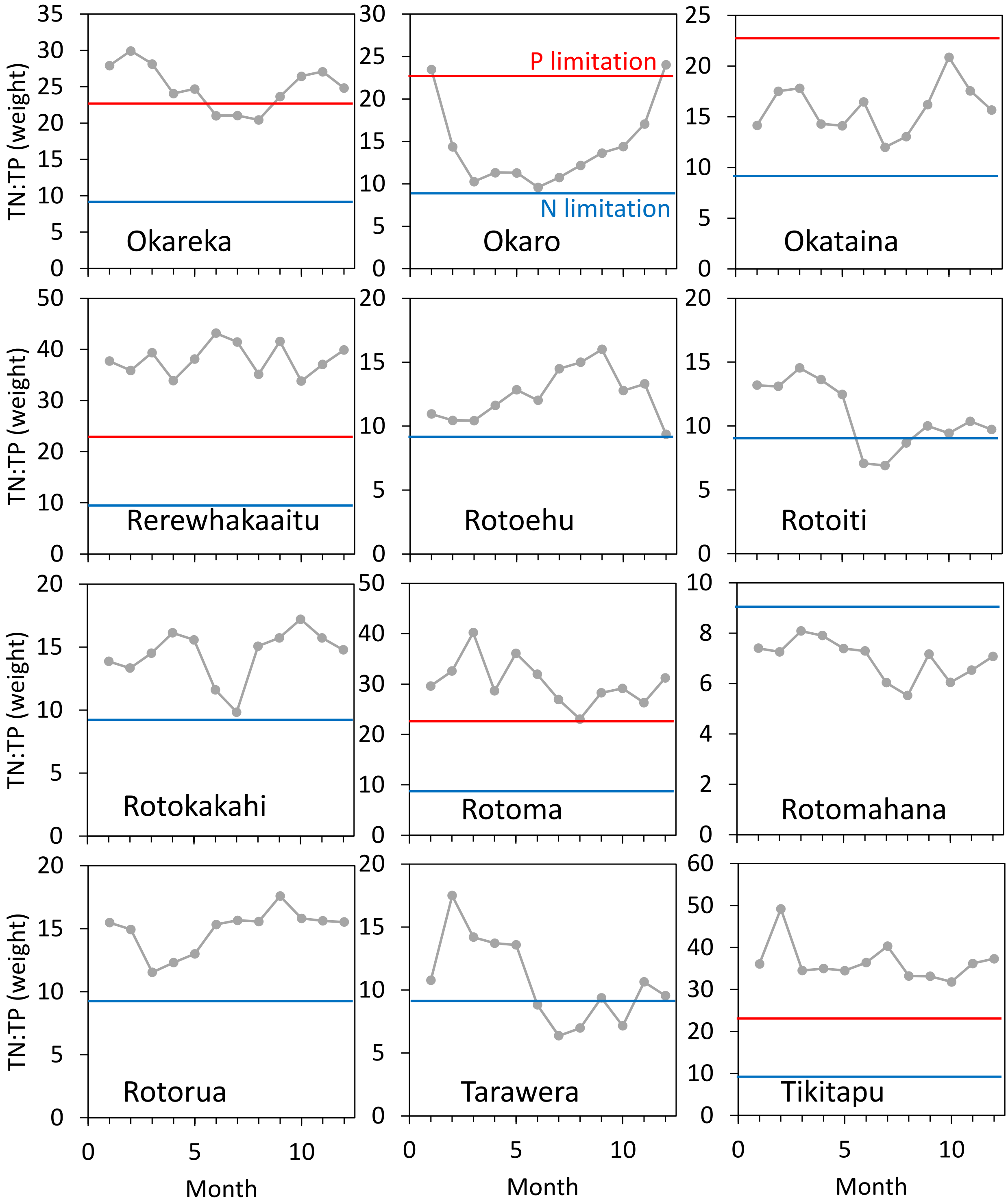


Figure 31: Ratios of monthly mean TN:TP (2001-2020 averages). Red lines indicate the limit above which P limitation of phytoplankton growth is likely and blue lines indicate the limit below which N limitation is likely.

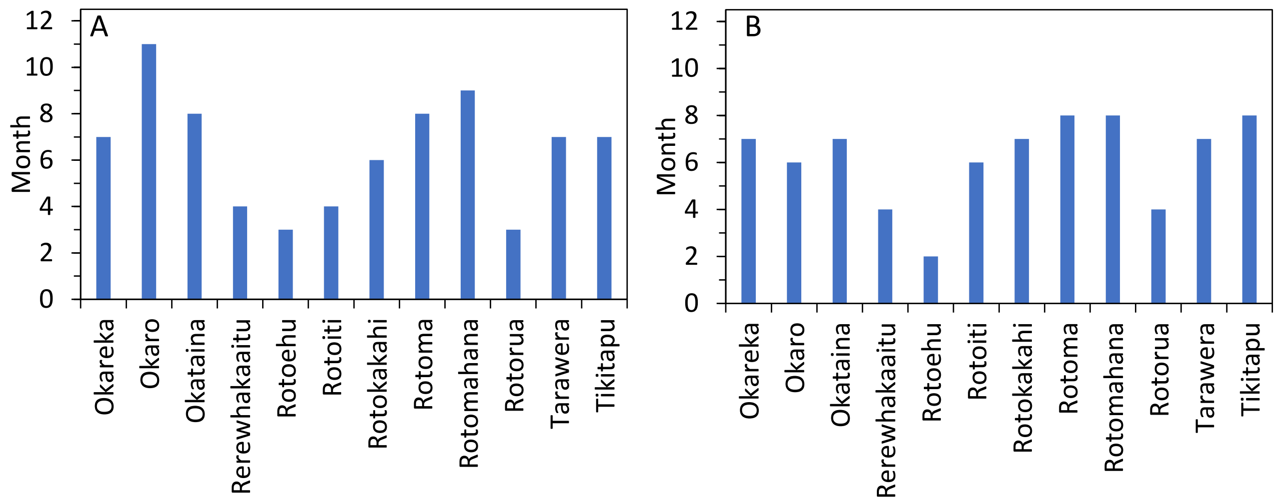


Figure 32: Month of highest chlorophyll *a* and TP in each lake. Averages for 2001-2020. A. chlorophyll *a*. B. TP.

Average TN:TP weight-based ratios varied from 6.9 (Rotomahana) to 37.9 (Rerewhakaaitu, Figure 31). The average TN:TP ratio was between 8.5 and 15 in lakes with average chlorophyll *a* >5 mg m^{-3}. Higher ratios only occurred in low chlorophyll lakes (Okareka, Rotoma, Tikitapu, Rerewhakaaitu). In these four low chlorophyll lakes the annual average TN:TP ratios were >22.6, the limit above which P limitation is likely (Guildford and Hecky 2000), while only in Lake Rotomahana was the TN:TP ratio on average low enough (<9.0) to suggest N limitation may be dominant (and in Rotoiti and Tarawera in the wintertime, Figure 31). When 9.0 < TN:TP < 22.6 other indicators and nutrient uptake assays must be examined to determine whether N or P limits phytoplankton growth (Guildford et al. 2022).

Some lakes showed large seasonal swings in the TN:TP ratio over the year (Okaro, Tarawera, Figure 31), while other lakes showed no seasonality (Rerewhakaaitu). In nine lakes the TN:TP ratio was lower in winter (July-September) than in summer (January-March). In these nine lakes TN:TP in winter was on average 77% of the summer average (range 53-97%). This supports the findings of Guildford et al. (2022) of P limitation of phytoplankton growth generally being dominant during summer in the Rotorua Te Arawa lakes, although the TN:TP ratio is a less good indicator of phytoplankton growth limitation than the particulate nutrient ratios and other indicators used by Guildford et al. (2022) (see also chlorophyll:TN and chlorophyll:TP ratios in Appendix A). The TN:TP ratio was lower in summer than in winter in three lakes: Rerewhakaaitu, Rotoehu and Rotorua. These are the three lakes where TP and chlorophyll was highest in summer as opposed to the remaining lakes where it was highest in winter, and the three lakes with lowest stratification potential (Figure 33). Therefore, these three shallow lakes likely mix occasionally throughout summer. In these three lakes the annual average TN:TP ratio in winter was on average 26% higher than the summer average (range 5-43%). In the nine remaining lakes, which are deeper and tend to stratify more, phosphorus accumulates in the hypolimnion during summer, which is mixed into the surface layer during winter, reducing the surface layer TN:TP ratio and relieving P limitation, resulting in a phytoplankton biomass peak during winter.

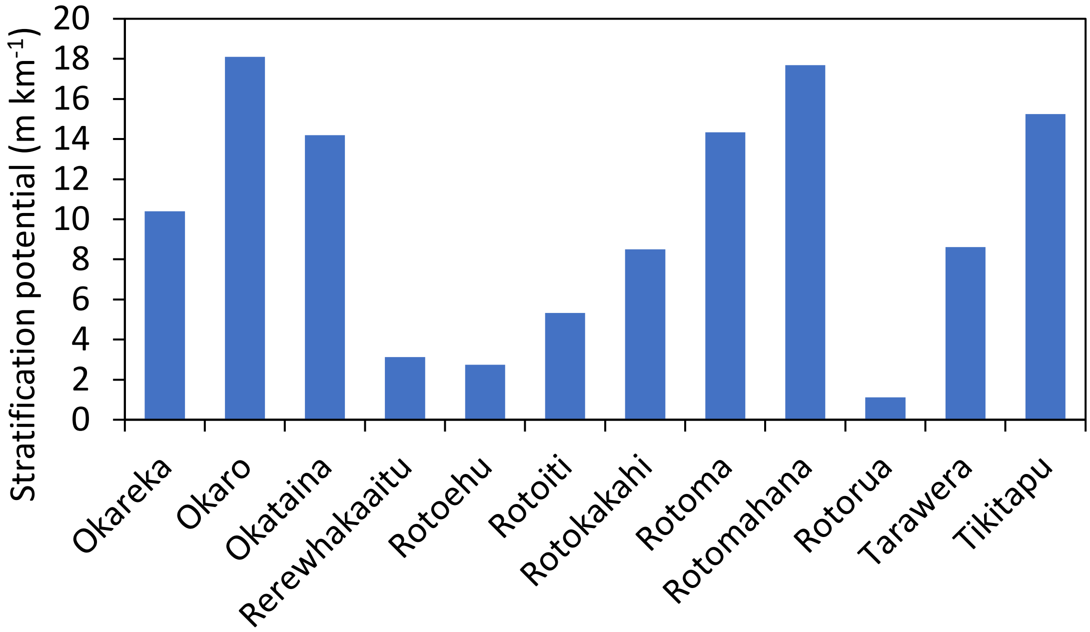


Figure 33: The stratification potential for each lake. As defined by Osgood (1988).

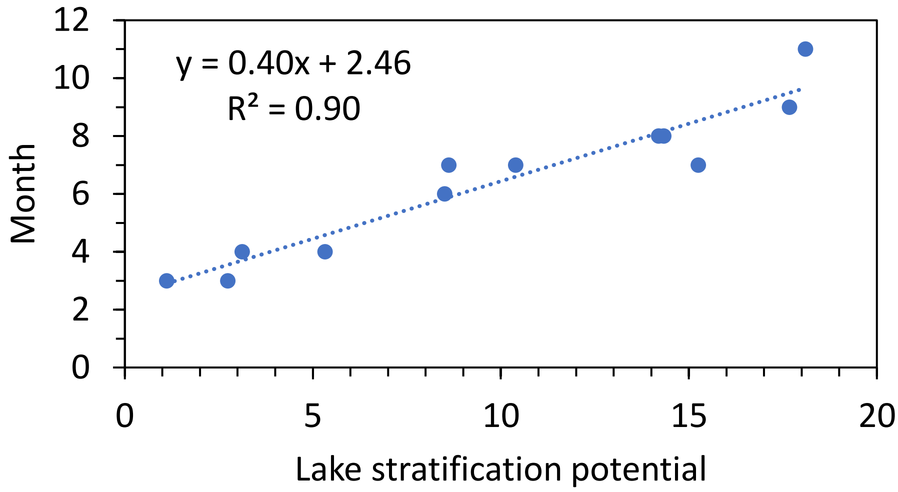


Figure 34: The average month of maximum chlorophyll *a* against the lake stratification potential (z $A^{-0.5}$). Data of 2001-2020. Each data point represents one lake.

3.9 Bottom water oxygen

Apart from sediment contents bottom water oxygen is an important driver of internal loading in most lakes. Anoxic bottom water usually enhances phosphorus release from the sediment although other factors play a role as well (Hupfer and Lewandowski 2008) such as temperature, sediment disturbance, and the amounts of labile P and of iron (Fe) and other elements in the sediments. Even in oligotrophic and fully oxygenated lakes such as Lake Taupo phosphorus builds up in the hypolimnion during summer (Verburg and Albert 2019) by decomposition of settling detritus and perhaps by slow release from the sediment.

Bottom water becomes anoxic during late summer to autumn in monomictic lakes Okareka, Okaro, Okataina and Rotoiti (Figure 35). Lake Tikitapu can become anoxic at the deepest point. Bottom water anoxia in stratifying lakes is typically most extensive in autumn. By June bottom water DO typically increases again as a result of reduced stratification allowing increased vertical mixing. The low oxygen in late summer/autumn and the potential for internal P loading is important for monomictic lakes. Phosphorus accumulates in the hypolimnion during summer in these lakes and is mixed into the trophic zone during the winter overturn. This explains why chlorophyll concentrations are highest in winter in most of the Rotorua Te Arawa lakes. Total phosphorus and chlorophyll concentrations were highest in winter in nine of the 12 lakes, excepting lakes Rotorua, Rotoehu and Rerewhakaaitu (previous section). In the latter three lakes oxygen in April-May was highest among the 12 lakes.

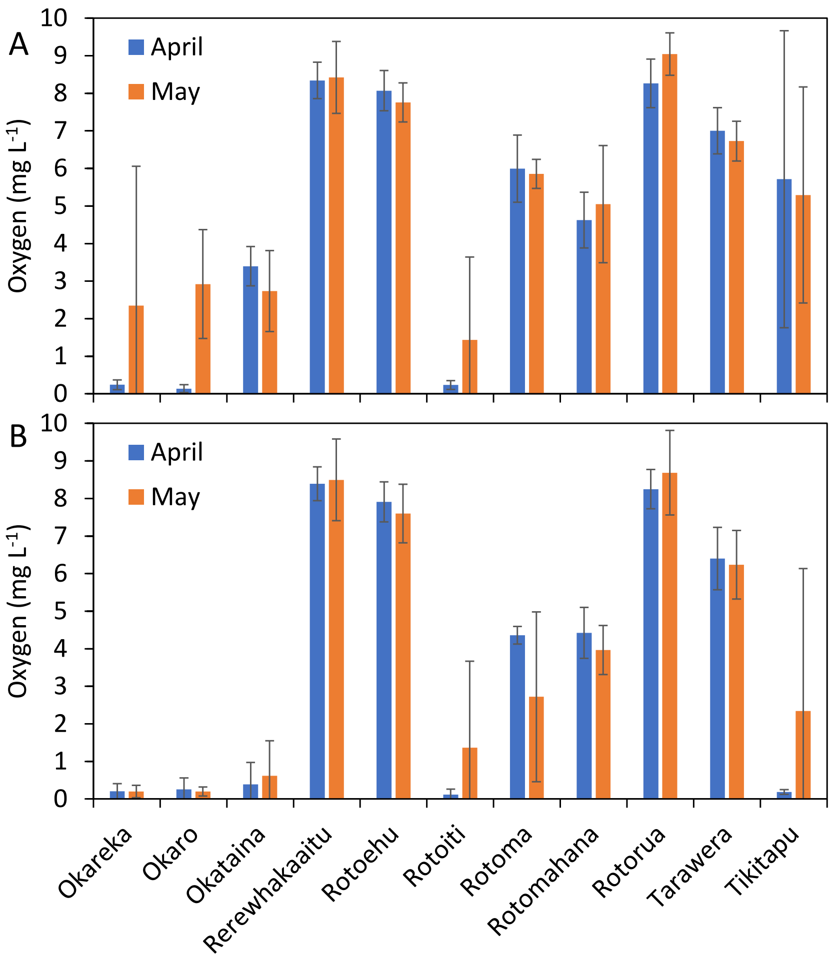


Figure 35: Average deep water DO in each lake in April and May in 2016-2020. A. At the mean depths (Table 4). B. At near maximum depths (deepest depth where sampling frequency was similar to shallower depths). Depths were: Okareka 29 m, Okaro 15 m, Okataina 62 m, Rerewhakaaitu 10 m, Rotoehu 10 m, Rotoiti 98 m, Rotoma 69 m, Rotomahana 109 m, Rotorua 21 m, Tarawera 80 m, Tikitapu 24 m. Error bars are standard deviations. Note oxygen profiles are not recorded in Lake Rotokakahi.

The monthly sampling (low frequency) and limiting this analysis to April-May may miss anoxic events in shallow polymictic lakes. For instance, Lake Rotorua is polymictic and during short periods of stratification the bottom water can rapidly deoxygenate and become anoxic (Guilford et al. 2022). In polymictic lakes this is more likely to happen during summer (December – March) than in autumn. In shallow lakes Rerewhakaaitu, Rotoehu and Rotorua bottom water DO was lower in December - March than in April – May and was lowest in February (on average about 2-3 mg L^{-1} at near maximum depths). Annual minima are shown in Figure 36. However, as mentioned the monthly sampling may have missed more severe events of lower oxygen in shallow polymictic lakes.

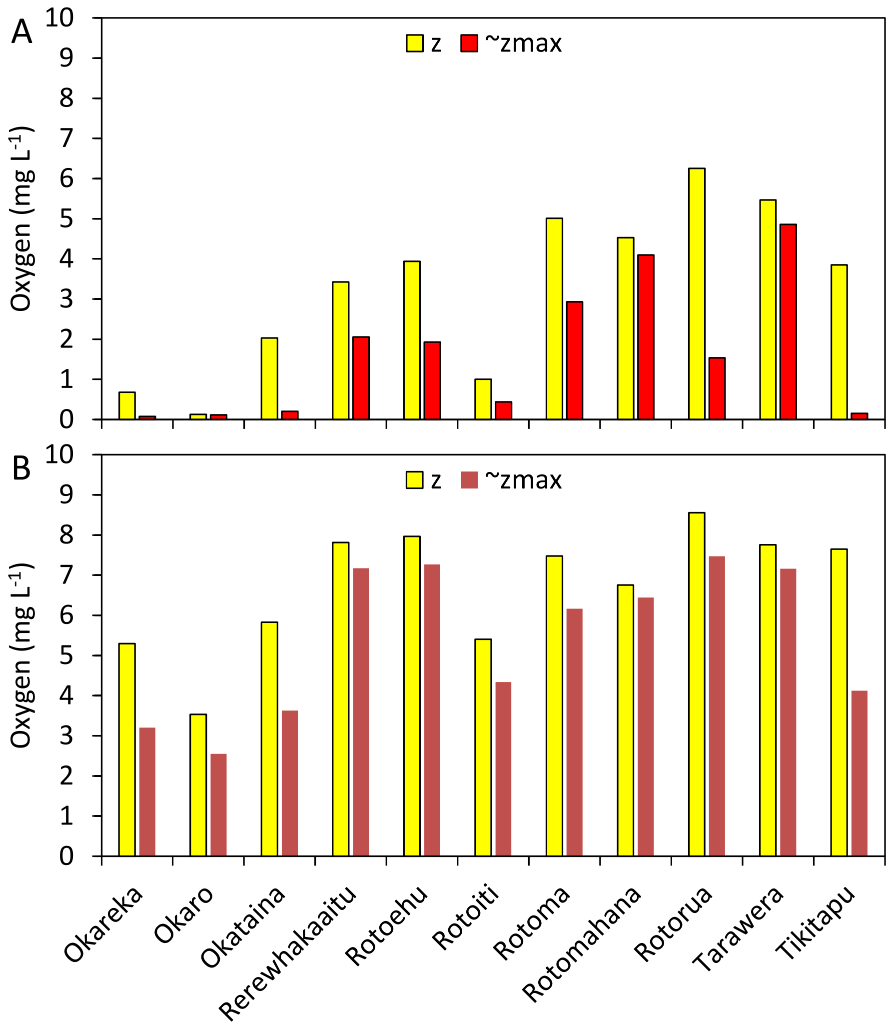


Figure 36: Minimum and average deep water DO in each lake. Data of 2016-2020. A. Averages of the minima in the years 2016-2020 at the mean depths (z) and at near maximum depths (~zmax, depths as in Figure 35). B. Averages of all data in 2016-2020 at the same depths. Oxygen profiles are not recorded in Lake Rotokakahi.

3.10 Gross internal P loading

The gross internal P load is P release from the sediments and can be measured, for instance by using benthic chamber incubations (Burger et al. 2007). The gross internal load does not account for sediment recycling, i.e., it is the total internal loading rate before deducting sedimentation rates, and therefore is a measure of internal loading that is expected to be related to the average standing crop of phytoplankton. The gross internal load drives phytoplankton growth and can be responsible for bloom formation.

Although a literature review carried out by Nurnberg (1984) found on average zero P release in oxic sediment core tubes, P and N may be released to the water column in lakes with an oxic sediment-water interface (White et al. 1980, Vincent et al. 1981, Sondergaard et al. 2001), which in the case of P release may be controlled by iron and sulphur concentrations in the sediment (Sondergaard et al. 2001, Gächter and Müller 2003).

Van der Molen and Boers (1994, cited in: Noordhuis et al. 2020) derived a formula to estimate gross internal P loads from the weight ratios of the total concentrations of P and Fe in the sediments, from an analysis of 47 shallow lakes. Applying the formula to data for the BOP lakes suggests that it may work as well in deeper lakes. Figure 37 shows estimates of gross internal P loading using the method of Van der Molen and Boers (1994) with sediment constituents from Trolle et al. (2008).

The estimated internal P load per unit area correlated positively with the amount of total P in the sediment (R^2 = 0.40) but this was largely driven by Lake Rotoiti having the highest amount of P in the sediment (3485 mg P kg^{-1} dry weight) and the highest internal load (R^2 = 0.30 without Lake Rotoiti). There was almost no correlation between the gross internal P load per unit area and annual mean chlorophyll concentrations in the lakes (R^2 = 0.00). However, there was a good correlation between the gross internal P load per unit area divided by mean depth, and annual mean chlorophyll (R^2 = 0.62). This was despite there being little correlation between the estimates of external and gross internal P loads both per unit area and after dividing by mean depth z ($R^2 < 0.01$ and R^2 = 0.14, respectively). Dividing the gross internal load by the average depth of the water column makes sense because the P released from the sediment is mixed throughout the water column before it can benefit phytoplankton growth, while this is less the case for the external load. The good relationship with annual mean chlorophyll provides support for the estimates of gross internal loads. However, relationships of gross internal P loads per unit area divided by mean depth showed less correlation with lake TP concentrations (R^2 = 0.38, and the correlation was only R^2 = 0.10 for gross internal P loads per unit area).

The fit of annual mean chlorophyll with the external P load per unit area (R^2 = 0.27) improved by dividing this load by the mean depth (R^2 = 0.65), just like it did for the gross internal P load. The correlation with annual mean chlorophyll improved further with the total P load (external + gross internal) per unit area divided by mean depth (R^2 = 0.76, Figure 38).

The estimate for the gross internal P load in Lake Rotorua (4.1 g m^{-2} y^{-1}) based on the sediment constituent analysis agreed with average observed internal P loads in Lake Rotorua at three sites which were 2.9, 3.3 and 16 g m^{-2} y^{-1} (Burger et al. 2007). The lower two values of Burger et al. (site depths 7 and 14 m, respectively) may be more representative for the spatial average gross internal load in the lake because the third site was at 20 m, well below the average lake depth (10 m). The average of the results of Burger et al. (2007), after weighting for bottom area (depth ranges 0-10.5 m, 10.5-17 m and 17-45 m) was 5.62 g m^{-2} y^{-1}.

Burger et al. (2008) found an internal load of 2.9 g m^{-2} y^{-1} in Lake Rotorua by modelling. White et al. (1978) reported an internal load in Lake Rotorua of 7.3-14.6 g P m^{-2} y^{-1} in 1975-1976. Rutherford et al. (1996) found the lower bound of White et al.'s (1978) estimate to agree with their dynamic model which included separate terms for sediment P release and for the net P sedimentation rate.

In a literature review of 49 shallow lakes Van der Molen (1994) found a median gross internal load of 3 g P m^{-2} y^{-1}. Gross internal loads, as determined in situ by measuring release rates from the sediment, can be comparatively huge (in the order of hundreds of times the external load, Ekholm et al. 1997), but most of the gross internal load cycles back into the sediment (Burger et al. 2007).

Burger et al. (2008) found gross internal N and P loads in Lake Rotorua calculated with the model DYRESM–CAEDYM that were 62% and 88%, respectively, of the annual average total nutrient load (external + gross internal). Burger's et al. (2008) results suggested that the internal P load in Lake Rotorua is on average seven times higher than the external P load. The estimated proportion of gross internal P load of the total load was 85% according to the methods and data used here (Figure 37C), similar to the estimate of Burger et al. (2008).

Across the 11 lakes with oxygen data, the gross internal P loads per unit area increased with decreasing annual minimum oxygen at mean depths (R^2 = 0.15), and with decreasing oxygen at mean depths in April (R^2 = 0.27) and May (R^2 = 0.29), as expected. However, none of these weak correlations were statistically significant. The weak relationships are not surprising because the estimates of gross internal loads were based on sediment contents, and were not based on observations of actual P release from the sediment, which is likely to be higher under anoxic conditions than when the water sediment interface is oxygenated. In other words, our gross internal loads should be considered as potential loads, given appropriate redox and other conditions.

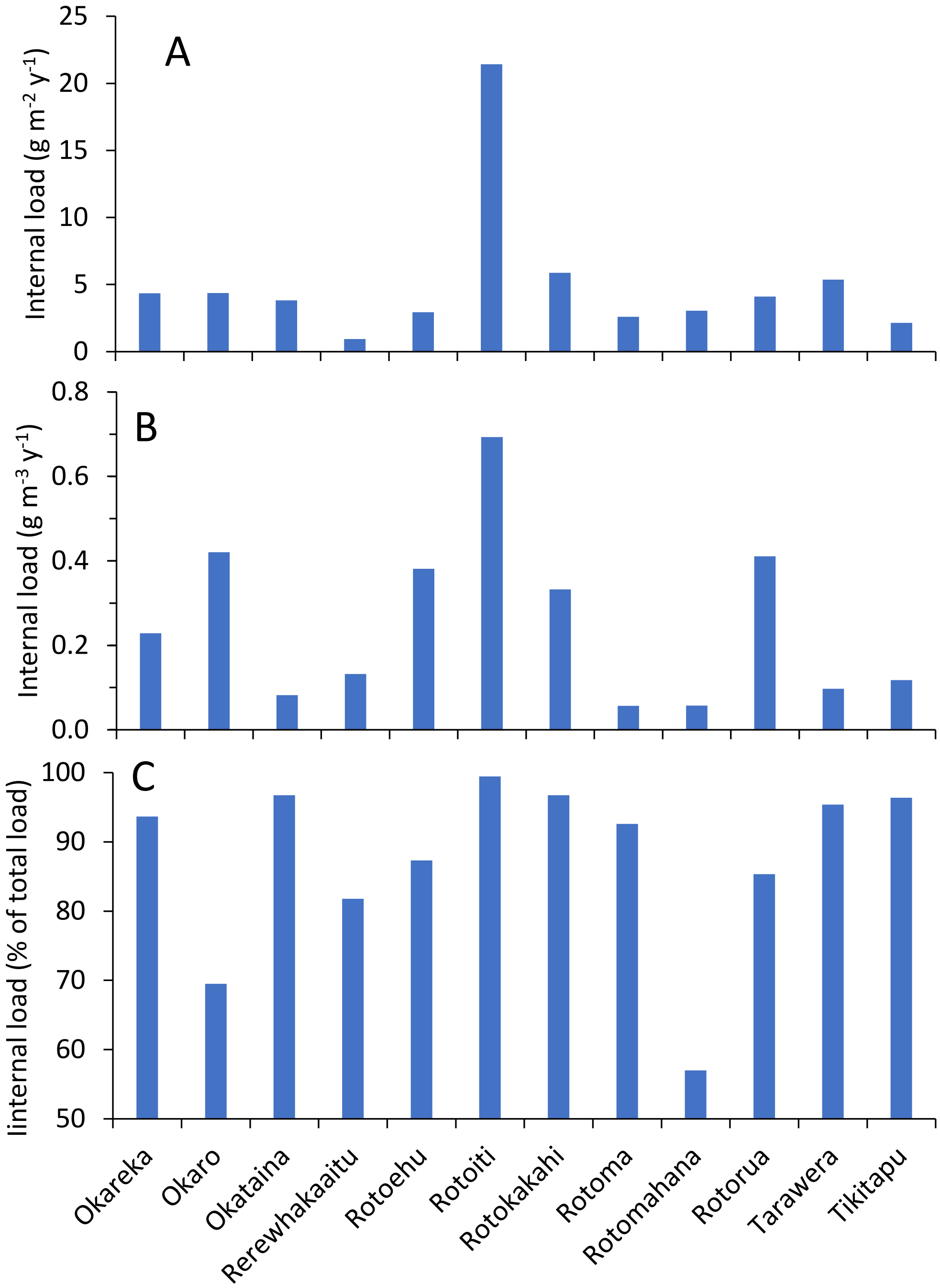


Figure 37: **Gross internal loads.** A. Gross internal P load per unit area as estimated from the ratio of total P and Fe in the sediments (sediment data used for this analysis were taken from Trolle et al. 2008). B. Gross internal P load per unit area divided by mean depth. C. Gross internal P load as a proportion of the total load (external + internal).

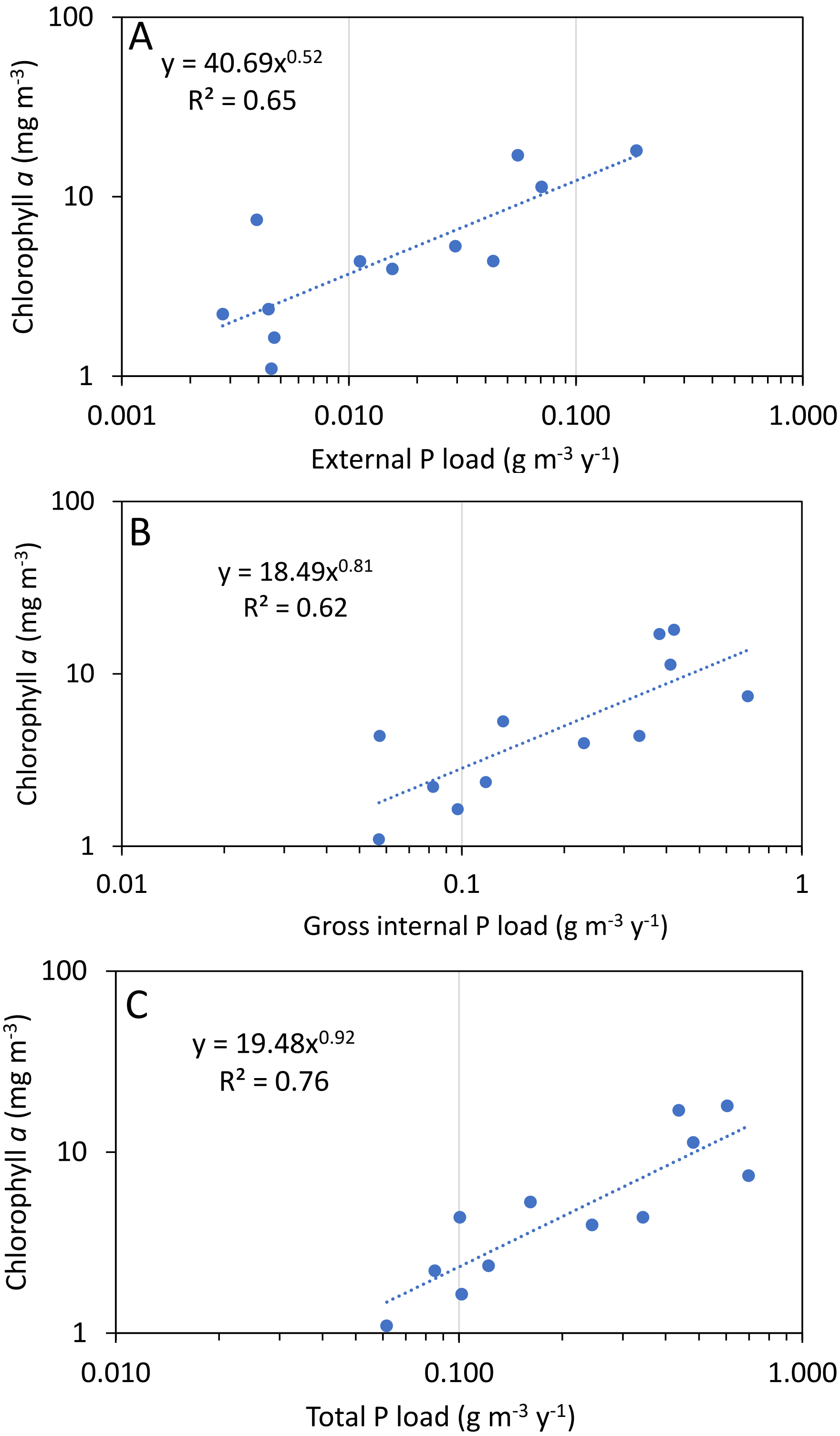


Figure 38: Relationships of annual mean chlorophyll a with the external and internal P loads. Comparing relationships of annual mean chlorophyll *a* with the external P load (A), the gross internal load (B), and the total load (C), all per unit area divided by mean depth.

3.11 Denitrification

A large part of the nitrogen load to lakes is usually removed by denitrification. The average N retention found in lakes is 34% and on average about two thirds of N retention in lakes is accounted for by denitrification and the remainder by uptake by aquatic plants and sedimentation resulting in permanent burial (Saunders and Kalff 2001) but these proportions are highly variable.

Rates of denitrification and loss of N and P by burial in the sediment are often not well known for individual lakes. Measuring denitrification rates directly, and estimating nutrient accumulation rates in the sediment from sediment cores, is time consuming and often involves high uncertainty. Instead, ratios of N:P in different compartments of the nutrient budgets of lakes can be used to estimate denitrification and burial rates of N and P (see Section 2.4 for the methods).

The nutrient mass balance suggested negative denitrification occurred in three lakes (Ōkaro, Rerewhakaaitu and Rotomahana) as a result of TN_{net}:TP_{net} < TN_{sed}:TP_{sed} (Figure 39A). However, TN_{net}:TP_{net}, the ratio of N and P removed from the lake water (but not via the outflow), cannot be less than TN_{sed}:TP_{sed} because all P that is lost (Table 7) is buried in the sediment while at most 100% of the N lost is buried in the sediment. And, naturally, denitrification cannot be negative. The negative denitrification result suggests that at least some of the input data to the nutrient mass balance models are inadequate for these three lakes, although also the mean sediment data of total N and P of Trolle et al. (2008), or more precisely their N:P ratios (it is the ratio that is used in the equation), may not be representative for these three lakes. Without proper input data it is not possible to achieve acceptable nutrient mass balance models and this likely explains part of the error illustrated by Figures 19 and 22.

The highest rates of denitrification per unit area occurred in Lake Rotorua (3.0 g N m^{-2} y^{-1}) and Lake Rotoiti (2.3 g N m^{-2} y^{-1}) and the lowest in Lake Tikitapu (0.27 g N m^{-2} y^{-1}) and Lake Rotoma (0.01 g N m^{-2} y^{-1}), while denitrification as a proportion of the N inputs ranged from 0.6% to 71% (average 40%), not considering lakes with negative results. The average for all 12 lakes was 30% of the N inputs when assuming denitrification to be zero in those lakes where the mass balance suggested negative rates. This is closer to the average found by Saunders and Kalff (2001) of about two thirds of 34% retained.

There was virtually no correlation between various measures of deep water oxygen (Figures 35 and 36) and denitrification rates. There were no strong correlations between average chlorophyll concentrations and denitrification rates (either as areal rate or proportion of the inputs), which was unexpected (Bruesewitz et al. 2011). Perhaps the lakes in the Bay of Plenty region differ too little in trophic state to resolve differences in denitrification rates. However, the lowest denitrification rates did occur in the least productive lakes (Rotoma and Tikitapu). The conclusion that the low correlation of average chlorophyll concentrations with denitrification rates per unit area might be because of a limited range in trophic state among the lakes was supported by the improvement produced (R^2 = 0.93) by adding two lakes at the extreme ends of the trophic scale, oligotrophic Lake Taupo and hypereutrophic Lake Horowhenua. This correlation was mostly driven by the very high denitrification and chlorophyll in Lake Horowhenua.

As the proportion lost by denitrification increases, the ratio of N:P in the lake decreased relative to volume weighted average in the inflows (Figure 39B). When denitrification exceeds about 50% of the external N load the TN:TP ratio in the lake becomes less than that in the loads, suggesting a larger proportion of N is lost than of P from the surface waters in the lake, and accordingly the N:P ratio decreases relative to the loads.

In lakes where denitrification is high enough to reduce the N:P ratio relative to the loads N limitation of phytoplankton growth is more likely to occur. Denitrification was not related to the N:P ratios in the lakes (R^2 = 0.03), only to the change from the ratios in the inflows to ratios in the lakes.

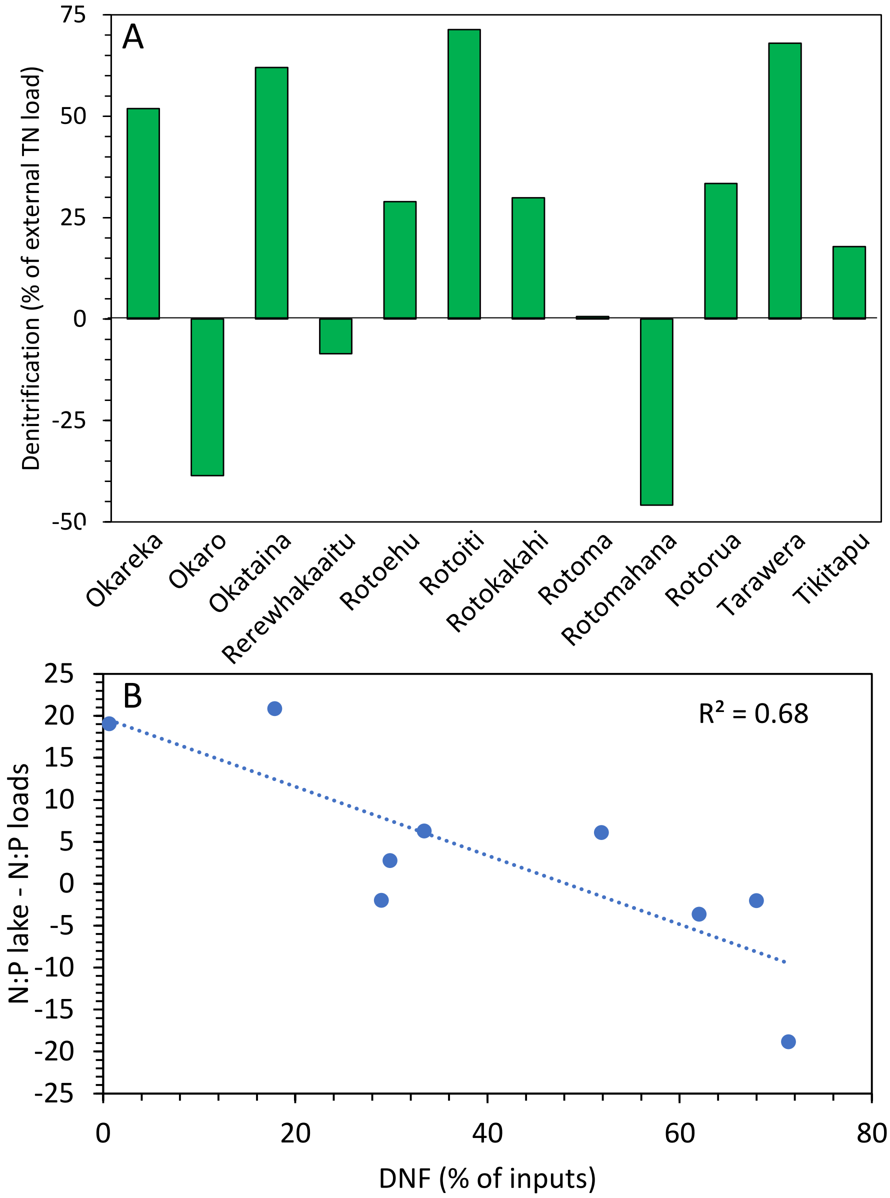


Figure 39: Denitrification. A. presented as a percentage of the external nitrogen loads. B. The change in TN:TP ratios (weight) from the catchment loads to the lake concentrations, with negative values when the ratio is lower in the lake than in the loads (excluding three lakes with negative results for denitrification shown in A).

There was little correlation between average TN:TP ratios in the lakes and the ratios in the loads (R^2 = 0.03) and between TN:TP in the loads and the change in TN:TP from the loads to the lakes (R^2 = 0.14). There was a strong correlation between TN:TP in the lakes and the change in TN:TP from the loads to the lakes (R^2 = 0.72, Figure 40).

Lakes Tarawera, Rotoiti, Rotoehu and Okataina, the four lakes where the TN:TP ratio was lower in the lake than in the loads (Figure 24), have the lowest lake TN:TP ratios (8, 8, 11 and 13 respectively), apart from Lake Rotomahana (8, up from 5 in the load) and Lake Okaro (10, up from 6 in the load) where the lake TN:TP ratios were low as well. In these lakes the TN:TP ratio may be sufficiently low to enhance N limitation (Figure 40).

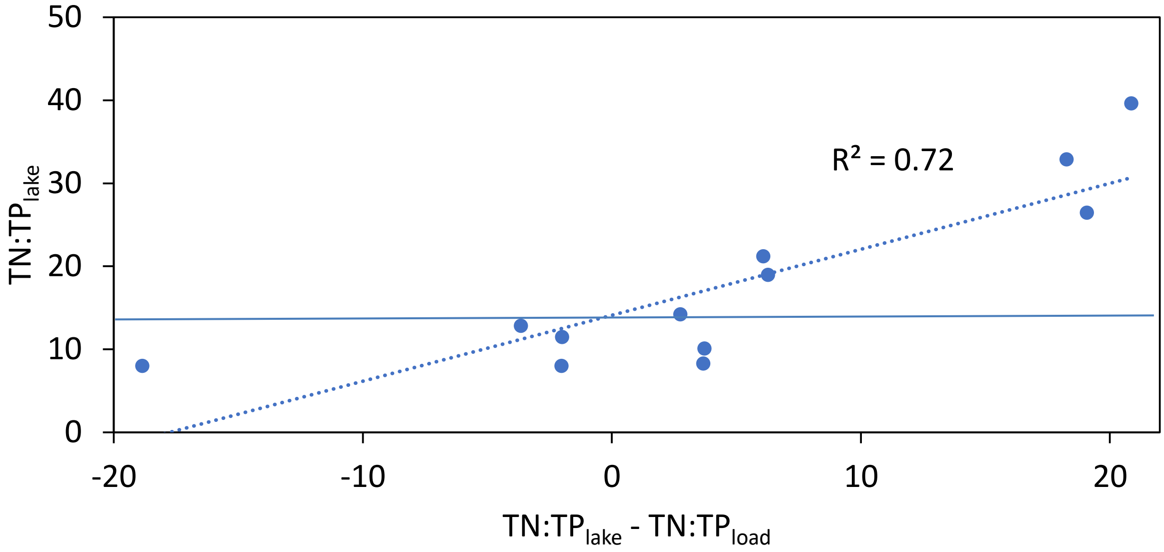


Figure 40: TN:TP ratios in each lake compared with the change in TN:TP ratios from the catchment inflows. Ratios are weight based. A linear fit is included. An exponential curve does not improve the fit.

3.12 Nutrient retention

Nutrient retention is the proportion of the external load that does not leave the lake via the outflow (sediment burial for P, sediment burial + denitrification for N). In none of the Rotorua Te Arawa lakes did the export of N or P through the outlet exceed the external inputs. Therefore, there was a net retention of TN and TP in all 12 lakes, indicating that the average fluxes of sedimentation and permanent burial of P and N in the sediment (and denitrification in the case of N) are greater than the average fluxes of P and N release from the sediment.

From data presented by Burger et al. (2007, their Figure 6), it is clear that no more than about half of the TP and TN entering Lake Rotorua from external sources leaves the lake through the outlet. Therefore, their data support net retention in the lake for both TN and TP, confirming the average sedimentation flux (plus loss of N by denitrification) is higher than the release of P and N from the sediment. An even greater proportion of TN (56%) compared with TP (50%) remained in Lake Rotorua or was lost by denitrification according to data in Burger et al. (2007). These proportions in Burger et al. (2007) were lower then found here, 78% for N and a higher proportion for P (85%). This difference is likely explained by the TN and TP loads in Burger et al. (2007) being only 75% and 62%, respectively, of the loads for Lake Rotorua in McBride et al. (2021). Lower load estimates would result in lower retention estimates after subtracting the nutrients lost through the outlet from the loads.

Mean depth (z) was not strongly correlated with nutrient retention of N (R^2 = 0.31) or P (R^2 = 0.02). Correlations of retention with q_S and T_W were similar to that with z for both nutrients.

Observed (i.e., estimated from the data) retention of P was less than predicted by each of the 3 models (Nurnberg 1984, Vollenweider 1976 and Kirchner and Dillon 1975) only in lakes Rotoehu and Rotoiti (Figures 41 and 42). In Lake Okataina only R predicted by Vollenweider (1976) exceeded the observed retention. This is consistent with these three lakes (Rotoehu, Rotoiti, Okataina) being among the four lakes where TN:TP ratios decreased from average inflow to in-lake concentrations (Figure 24, section 3.6), suggesting proportionally less retention of P than of N and higher than predicted lake concentrations of P. The ratio of the $TN:TP_{lake}$ to $TN:TP_{load}$ (i.e., the ratio of these ratios) increased with increasing retention of P (R^2 = 0.49, $p < 0.02$). In other words, the TN:TP ratio increased significantly from inflow to the lake with increasing P retention, as expected. In addition, the observed retention of P being less than predicted in lakes Rotoehu and Rotoiti is consistent with all mass balance P models overestimating the P load in these two lakes from the lake P concentrations (Figure 16), even though all P models usually underestimated the loads in the other lakes (7 out of 12 lakes).

On the other hand, Lake Rotoiti being a large outlier in Figure 41 with very low retention for its estimated residence time might be the result of underestimated outflow rates. It is possible that its outflow was underestimated when taking into account the effect of the diversion wall in Lake Rotoiti (McBride et al. 2021), and therefore its residence time might be overestimated.

The models can overestimate the external P load for a given lake P concentration when internal loading occurs, because the models cannot distinguish between internal and external load sources of lake nutrient concentrations.

The mass balance methods to estimate internal loading of Nurnberg (1984) and Verburg et al. (2018), suggest internal P loading to occur only in lakes with less than predicted retention of P, therefore in lakes Rotoehu and Rotoiti. In principle the gross internal load can be found from the difference between modelled total load and the known external load, and the net internal load can be estimated from the gross internal load using equations in Verburg et al. (2018) and Nurnberg (2009). However, internal P loading indicated by the mass balance only in lakes Rotoehu and Rotoiti contrasts with internal loading of P known to occur in several more of the remaining lakes, for instance in Lake Rotorua (Burger et al. 2007) and Lake Okaro (Ozkundakci et al. 2011). But if the external P loads were overestimated for all or most lakes by McBride et al. (2021) or if the predicted retention was too low in each lake (see below for discussion) then improved estimates for these two numbers would result in internal loading indicated in more lakes.

On the other hand, the gross internal load per unit volume as predicted from the sediment contents was highest in these four lakes (Figure 37B), providing support for this method to estimate the gross internal load. In all four lakes bottom water becomes anoxic over summer.

Also, the ratio gross internal P load: external P load increased the more negative the difference between observed and predicted P retention was (R^2 = 0.72, $p < 0.0005$, taking for predicted R the average of the three methods to predict P retention). Even after excluding Rotoiti and Rotoehu the correlation is still strong (R^2 = 0.59, $p < 0.01$). These are not interdependent estimates because the gross internal load was not estimated from the mass balance. Therefore, this correlation provides additional support for the method to estimate the gross internal load.

Regarding the method to use P retention to enable estimation of internal P loading, it is noteworthy that only the P retention predicted by Vollenweider (1976) exceeded the observed retention in Lake Okataina, that average P retention was similar between the methods of Nurnberg (1984) and Vollenweider (1976), 0.65 and 0.67 respectively, and that the average for the five lakes with potential internal P loading (Rotoehu, Rotoiti, Okataina, Rotorua, Okaro) was virtually the same between the two methods (0.60 for both). This is surprising in view of the fact that Nurnberg (1984) on purpose excluded lakes with suspected internal loading, to enable the method to estimate internal loading in lakes with suspected internal loading, while Vollenweider (1976) did not.

Observed retention of N was less than predicted by the model of Harrison et al. (2009) in seven lakes: Okaro, Rerewhakaaitu, Rotoehu, Rotoiti, Rotokakahi, Rotoma and Tikitapu. This may in part be a result of the choice of the coefficient in the equation. A smaller coefficient would result in lower predicted retention. However, a different coefficient in an equation of N retention depending on q_S would not be able to much improve the fit with observed retention (Figure 41B). Possibly because of variability in denitrification rates, the in-lake retention of N is generally less well predicted than that of P and there is less literature on the subject than on the prediction of P retention.

Observed N retention was higher than predicted in five lakes (Okareka, Okataina, Rotomahana, Rotorua and Tarawera). Average retention of N was 0.70 while the average predicted R was 0.73.

The maximum proportion of P retained in the 12 lakes was 0.92. The theoretical range of predicted R is from 0 to 1 in the formulations of Vollenweider (1976) and Dillon and Kirchner (1975) but the formulation of Nurnberg (1984) has an artificial and unrealistic maximum of 0.83 (Verburg et al. 2018). There were six lakes with observed R > 0.83.

The mean observed P retention was 0.71 while the mean predicted retention was 0.66, ranging from 0.65 to 0.67 between the three methods. The observed P retention was higher than predicted by each of the three R models in most lakes, which is consistent with the P models usually underestimating the P loads (in seven out of 12 lakes, Figure 16). In all seven lakes where all P models underestimated the P loads the observed P retention was above the predicted R value (for all three models of R). Because the lake P concentrations are the most reliable of the various variables affecting the mass balance, it may be that P external loads were instead overestimated by McBride et al. (2021) in those seven lakes, resulting in lower estimates of observed retention.

On the other hand, the formulations to predict annual mean P retention may be inadequate. They were based on empirical work in North American and European lakes, mostly at higher latitudes than the Rotorua Te Arawa lakes. These were temperate lakes that generally freeze over during winter, or at least did before the impact of climate change became more noticeable (lakes in the Netherlands, for instance, used to freeze over every year but do no longer). Dillon and Molot (1996) realized that annual mean P retention must be higher on average in lakes in warmer climatic zones than those lakes on which the empirical relationships were based when they found that P retention was much lower in winter, during ice cover, than during the rest of the year. The lakes in the Bay of Plenty are relatively warm and never freeze over. Therefore, it may be that the empirical relationships found in temperate North American and European lakes are less appropriate in most New Zealand lakes and result in underestimated P retention. The only model based on New Zealand lakes (Abell et al. 2019) did not estimate P or N retention.

It would be possible to fit predicted R to New Zealand lakes in such a way that it ranges from 0 to 1 (as in the formulations of Vollenweider 1976 and Dillon and Kirchner 1975), for instance by adapting predicted R of Vollenweider (1976) by inserting coefficients to multiply with $\sqrt{T_W}$ both in the denominator and enumerator. However, this should be fitted to a sufficiently large number of lakes, with reliable input data (in particular: loads, lake concentrations, and outflow rates).

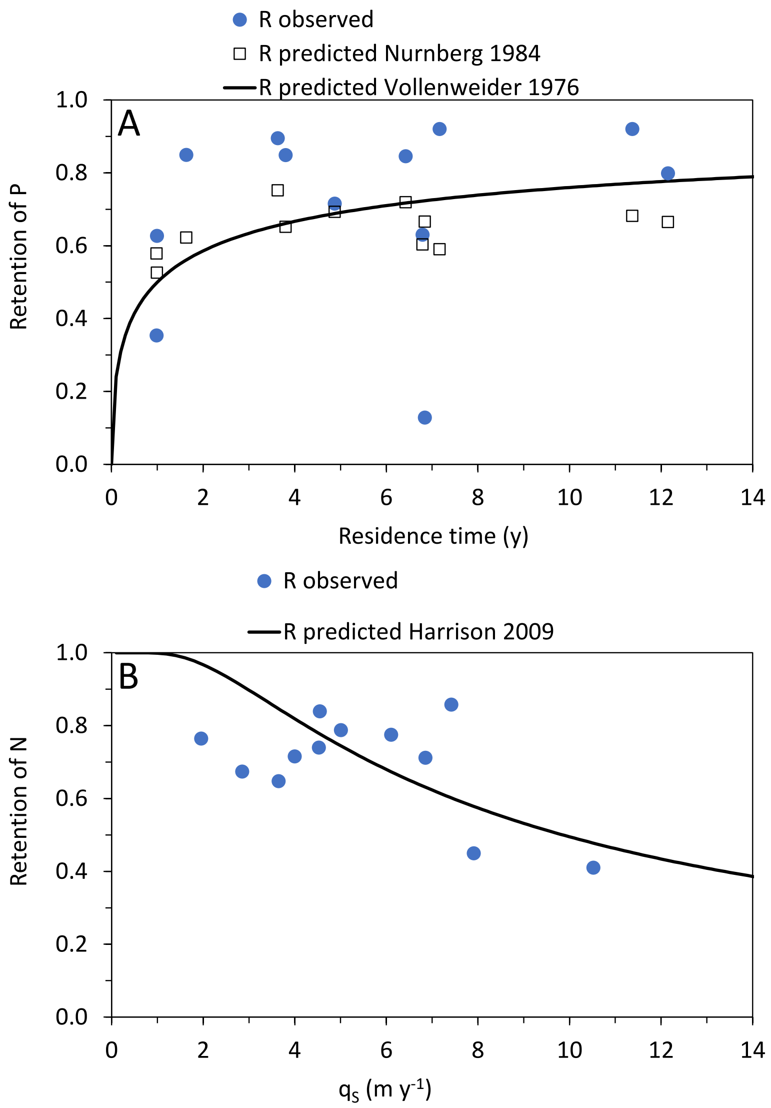


Figure 41: Nutrient retention (R) in each lake, observed and predicted (I). R is the proportion of the external load that does not leave the lake via the outflow (sediment burial for P, sediment burial + denitrification for N). A) Retention of P. R for P as predicted by Nurnberg (1984) is based on q_S and therefore does not relate well to the x-axis which is for T_W on which the prediction of P retention of Vollenweider (1976) is based. Retention of P as predicted by Kirchner and Dillon (1975) was very similar to that predicted by Nurnberg (1984) and is not shown (but see Figure 42). B) Retention of N. Retention of N as predicted by Harrison et al. (2009) depends on q_S.

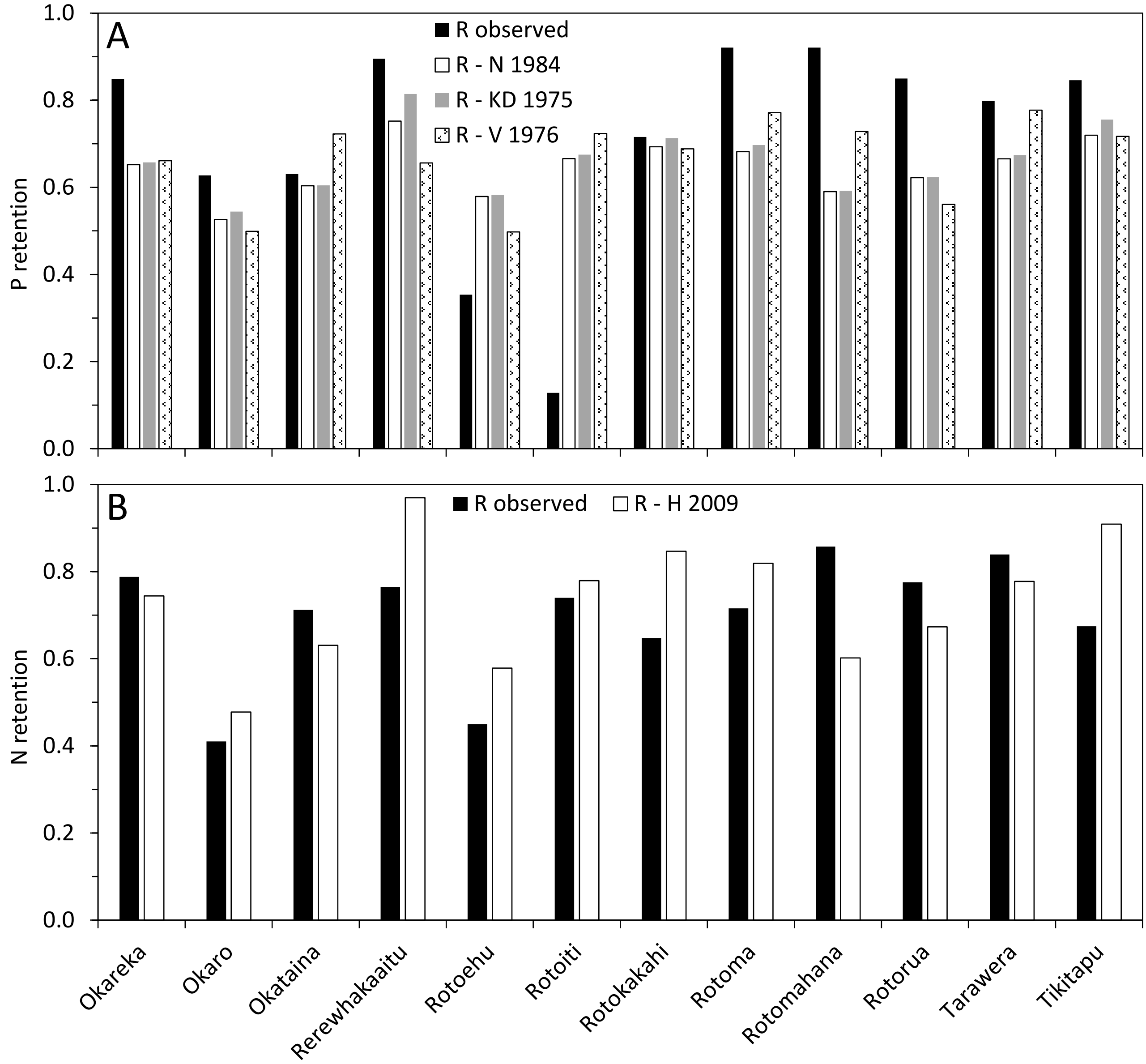


Figure 42: Nutrient retention (R) in each lake, observed and predicted (II). R observed is the observed retention as the proportion of the nutrient load following from the mass balance. A) P retention. R - N 1984 is retention predicted by Nurnberg (1984) based on q_S, R - KD 1975 is retention predicted by Kirchner and Dillon (1975) based on q_S, and R - V 1976 is retention predicted by Vollenweider (1976) based on T_W. B) N retention. R - H 2009 is retention predicted by Harrison et al. (2009) based on q_S.

3.13 Nutrient burial in the lake sediment

Burial in the sediment of P is by definition equal to the retention of P. This is only not true where a portion of the P load exits the lake other than by burial or through the outflow, for instance in the case of harvesting of P containing macrophytes. In Lake Rotoehu, harvesting of hornwort may remove 18% of the annual P load (and 15% of the N load, Verburg et al. 2018). Therefore, P burial in Lake Rotoehu as a proportion of the load estimated from the mass balance, although already low in Figure 43A (35%), may be lower by 18% after accounting for harvesting, suggesting 17% burial.

The smallest P burial (as a proportion of the external load, Figure 43) occurred in Lake Rotoiti (13%) followed by Lake Rotoehu (35%). The largest P burial occurred in lakes Rotomahana and Rotoma (both 92%) followed by Lake Rerewhakaaitu (90%).

In contrast to P retention, N retention is the combination of N burial in the sediment plus the loss of fixed nitrogen by denitrification (apart from losses by harvesting in lakes where that occurs). Therefore, TN burial as estimated from the mass balance is equal to TN retained - denitrification. The smallest N burial (as a proportion of the external load) occurred in Lake Rotoiti (3%) followed by Lake Okataina (9%). The largest N burial occurred in Lake Rotoma (71%, excluding lakes where denitrification was negative - Rotomahana, Rerewhakaaitu and Okaro - because as a result the N burial estimates in these lakes were doubtful). The proportions of P and N burial were not strongly correlated with z, T_W or z/T_W (= q_S).

Total nitrogen retained (as a percent of the external N load), and the contributions due to denitrification and sediment burial as estimated by mass balance are in Figure 43B. On average 70% of N inputs were retained, 40% by denitrification and 30% by sediment burial (excluding the lakes with negative denitrification estimates, when assuming denitrification zero in these lakes the averages were 30% and 40%, respectively, note the average of total retained = burial + denitrification is the same whether excluding or including lakes with negative denitrification).

These proportions are higher than the averages found by Saunders and Kalff (2001). Saunders & Kalff (2001) found a much lower average retention of nitrogen in lakes of 34%, although this may be in part explained by more lakes with short residence times in their data set. Saunders & Kalff (2001) found that on average about two-thirds of N retention in lakes was accounted for by denitrification, which agrees with our findings for the Rotorua Te Arawa lakes, and the remainder by uptake by aquatic plants and sedimentation resulting in permanent burial (Saunders and Kalff 2001), but that the proportions were highly variable.

Denitrification was large and negative in Lake Rotomahana explaining its unrealistic estimate of burial over 100% of the load (Figure 43C). Denitrification was also negative in lakes Okaro and Rerewhakaaitu but proportionally less, therefore the burial estimate in those lakes did not exceed 100% of the load. N burial in Figure 43C is artificially high in lakes where calculated denitrification was negative, because burial was estimated as retention minus denitrification. N burial cannot be higher than retention and cannot be higher than 100%. In other words, N burial in Lake Rotomahana in Figure 43C (132%) in reality cannot be higher than the total retention at 86% (Figure 43B).

Unfortunately, there was little agreement between P burial in the sediment estimated from the mass balance (Figure 43) and P burial estimated by multiplying P content in the sediment (mg P kg^{-1} dry weight) in Trolle et al. (2008) by the net sedimentation rate (kg m^2 y^{-1}) in Trolle et al. (2008). Correlations were almost non-existent (R^2 = 0.07) and even negative for burial in mg P m^2 y^{-1} and for the proportions of P burial relative to the load.

Burial estimates for N agreed better than for P between the mass balance approach and burial estimates following from data in Trolle et al. (2008). For N burial, some of estimates calculated from data in Trolle et al. (2008) data are too high (three lakes > 100% of the external loads), but less extreme than was the case for P (Figure 43A). However, also for N, the correlation was negligible and negative between N burial from the mass balance and N burial calculated from data in Trolle et al. (2008), as proportions of the loads.

The cause for the disagreement between P burial in the sediment as estimated from the mass balance and P burial estimated from data in Trolle et al. (2008) lies mostly in the latter. Trolle et al. (2008) analysed 2 or 3 cores in most lakes, and 8 in Lake Rotorua. Possibly either the net sedimentation rates or the N and P content in the sediment (or both) were not representative for lake-wide average conditions. Most of the estimates of P burial following from data in Trolle et al. (2008) are too high (Figure 43A). In eight of the 12 lakes P burial was over 100% of the external loads (range, 130-574%, average 262% of loads). Even the overall average is 186% of the loads. More than 100% burial is not possible: at 100% no nutrients would be leaving the lake through the outlet. The average was 71% for the estimates from the mass balances, which is in line with expectation from literature (Vollenweider 1976). Therefore, with the data available the mass balance is the best method to estimate lake wide average nutrient burial in the sediment. If the sediment nutrient contents of Trolle et al. (2008) are not representative for lake wide average conditions, that does not necessarily affect the estimation of denitrification and gross internal loading with the methods in this report because these methods did not use the actual concentrations in the sediment but rather their ratios which may be more reliable.

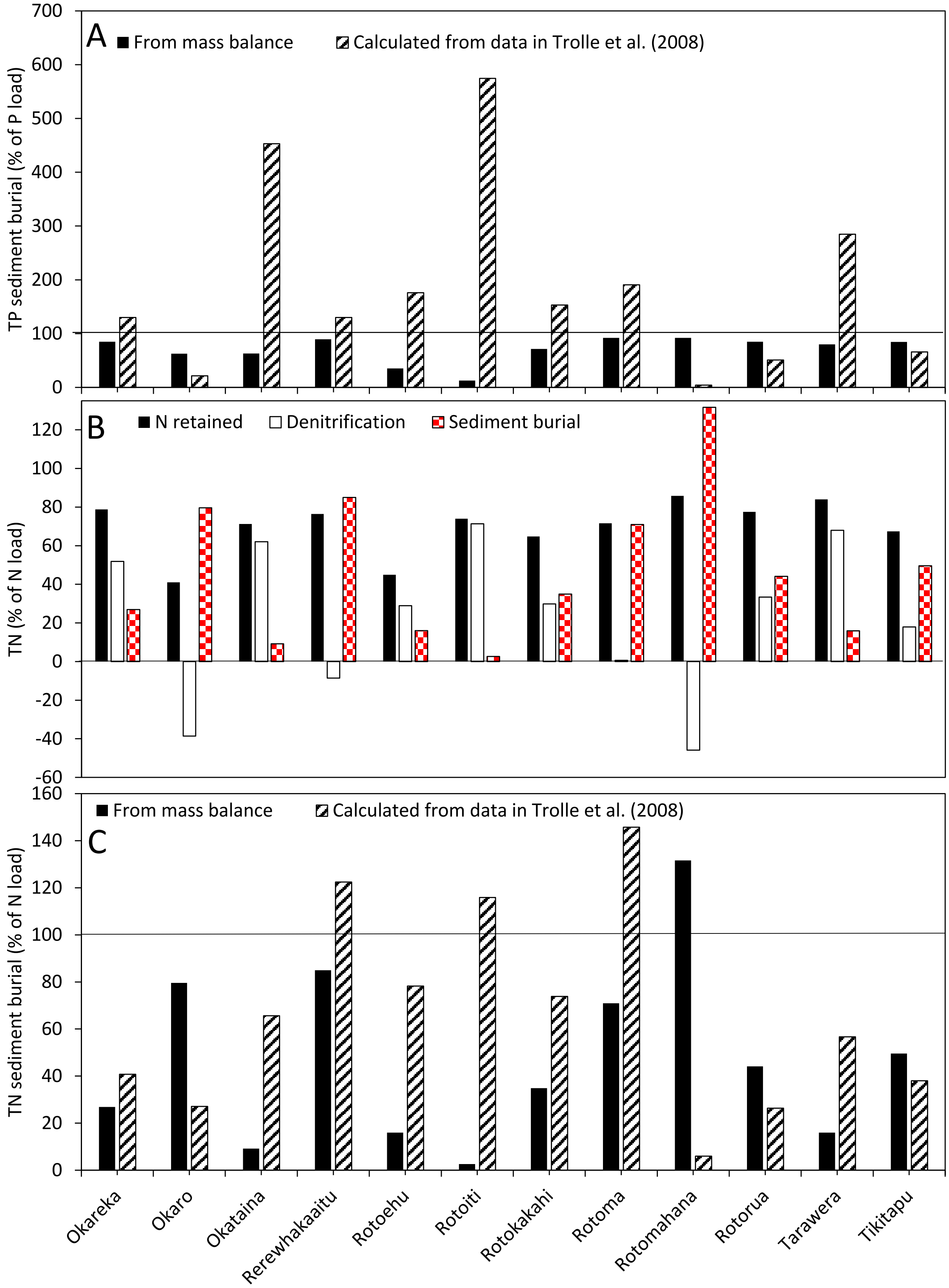


Figure 43: Burial of P and N as a proportion of the loads. Estimated from the mass balance and compared with burial calculated from data in Trolle et al. (2008). In A) and C) the horizontal lines show 100%. In B) N retention is split between sediment burial and denitrification, with burial estimated as retention – denitrification and N retained being the sum.

4 Discussion and Conclusions

4.1 General

There were clear differences between the lakes in processing of nutrients and therefore in the relationships between nutrient loads and in-lake concentrations. There was a remarkable range of TN:TP ratios in the surface layer water among the lakes (7 to 38 for annual averages), which were not related to the nutrient ratios in the loads (R^2 = 0.03). Therefore, the difference between lakes in dominance of N versus P limitation of phytoplankton growth is primarily decided, not by the relative proportions in the loads, but by differences in processes such as sedimentation, burial and release from the sediment, denitrification, and vertical mixing. Seasonality in TN:TP ratios in the deeper lakes can be explained by phosphorus accumulating in the hypolimnion, which is mixed into the surface layer during winter resulting in lower TN:TP ratios, relief of P limitation and phytoplankton biomass peaks during winter.

In nine out of 12 lakes the current P loads estimated from measurements (from McBride et al. 2021) were outside the range of P loads predicted by the mass balance models from measured concentrations. The results show the limitations of the mass balance modelling approach with the data available.

An option to derive the load to achieve TAS as an average ± the variance from the combination of outputs from all models is not appropriate because most or all models either overestimated, or underestimated, the nutrient load, depending on the lake (section 3.5).

The difference between the P loads estimated by the six models (based on 2016-2020 mean water quality) and the estimated current loads of McBride et al. was greater (average 61%, range 53-86%, Figure 22) than the desired reductions in the lake P concentrations (average 17%, range 0 – 65%). Especially in view of the latter argument, and because of the large difference in load estimates between models, we propose that instead of using modelling to find the loads needed to achieve TAS in each lake it is preferable to apply the proportional change from observed water quality to TAS in the same proportion to the catchment nutrient inputs.

4.2 Reasons why models did not perform well

That none of the models fitted well and that the average divergence of the models was greater than the improvement desired in water quality are the main concerns. Only models that do not substantially overestimate or underestimate nutrient concentrations compared with observed concentrations are appropriate. Divergences are generally explained either by error in the input data (which we must assume to be best estimates) or by the model for some reason not being appropriate. Overestimation suggests actual retention is greater than predicted retention. This could make sense, however, for instance in Lake Rotorua, where alum inputs may have enhanced P retention beyond what is expected in a lake with its characteristics (to a lesser extent this may also play a role in Lake Rotoehu although in that lake concentrations were underestimated). Underestimates, on the other hand, do not necessarily suggest data input error but may be explained by processes occurring in the lake, in particular internal loading, driving up observed nutrient concentrations compared with predictions. In this case, actual retention is less than predicted. In such cases, the load required to achieve TAS as incorrectly suggested by the models may even exceed the present external load estimate despite the TAS being lower than the current measured concentration.

The relationships presented by the mass balance lake models describe the average statistical behaviour of a large population of lakes, rather than the specific behaviour of a single lake (OECD 1982). When applied to the Rotorua Te Arawa lakes, the performance of these models was in general poor, in that predicted concentrations were substantially higher than observed concentrations in a large proportion of the lakes. No single model predicted P or N accurately enough in all (or even most) lakes to be considered sufficiently reliable for making projections on which to base management targets.

The only model developed using a New Zealand lakes data set (Abell et al. 2019) performed worse than the five other examined models for P, and was also not the best model for N. While the European and North American models may underestimate nutrient retention because of a difference in climate with New Zealand, the national scale New Zealand model of Abell et al. (2019) did not improve on the estimates of nutrient load versus lake concentration relationships of the European and North American models. It may be that the disagreement between estimates of current loads and the Abell et al. model estimates is explained by the fact that the model was developed using P and N loads estimated using the catchment model CLUES, rather than the areal export coefficient method used by McBride et al. (2019) with the latter providing the current load estimates adopted for the present study. Nevertheless, this does not change the fact that the model did not fit well with the current best estimates for the external loads and measured lake concentrations.

On the other hand, the estimated external loads of some of the lakes may be sufficiently inaccurate to prevent obtaining good model fits. The estimation of external loads is complicated especially in lakes that are connected, either by surface water (Tarawera receives inflow from several lakes, and Rotoiti receives inflow from Rotorua) or via ground water.

Model performance may seem impressive on average when fitted to a large number of lakes but models are not expected to fit individual lakes with sufficient accuracy to be useful to derive load reductions that are relatively small compared with the model error. Apart from the fact that the large error for individual lakes is obvious in plots of predicted versus observed lake concentrations, for instance in OECD (1982), the large error for individual lakes was pointed out by reviews of mass balance models, such as Brett and Benjamin (2008) and Ahlgren et al. (1988). Ahlgren et al. (1988) showed that the 95% confidence limits of the OECD (1982) equation for predicted lake P concentration enveloped a range more than twice the predicted values. The best mass balance model tested in Brett and Benjamin (2008) explained 84% of the variability in lake TP concentrations but had a large prediction error for individual lakes. Brett and Benjamin (2008) suggested that the large prediction errors may be due to inadequate lake input data such as the estimates of the external loads, and the large number of factors that influence TP losses to lake sediments (e.g., the amount of iron, calcium and/or aluminium in lake sediments, bioturbation by benthivorous invertebrates or fish, dominance of the zooplankton community by efficient grazers like *Daphnia* and production of faecal pellets, and the form in which phosphorus is supplied to the lake). Therefore, the model error for individual lakes is not random but may have a specific cause for each lake. In support of that point, it is notable that the models in this study agreed in either overestimating or underestimating the loads in most of the lakes.

Although it is not expected that mass balance-predicted loads would fit observed/estimated loads exactly (nor predicted concentrations to fit exactly the observed concentrations) because the models were based on a large number of lakes with the data set containing unexplained variability, even so the magnitude of the differences between predicted and observed values are a concern.

An additional explanation for poor model fit and error in mass balance model predictions is in the assumption of steady state conditions. External loads and in-lake concentrations are never truly in equilibrium. Annual mean input data (external loads, lake concentrations, inflow and outflow rates, residence time) change all the time. In addition, it takes time for lake nutrient concentrations to adjust to changes in external loads and flow rates. Therefore, there is always a slight mismatch to be expected between the data of the external loads and the lake concentrations.

The imperfect model fits with observed nutrient loads lake nutrient concentrations may in part be the result of restoration interventions in some of the lakes. For instance, dosing of P-binding agents alum and Aqual P (zeolite) dosing in lakes Rotorua, Rotoehu, and Okaro are likely to have altered the mechanics of P-retention in these lakes and may have prevented models from fitting well, upsetting expected relationships between external loads and lake concentrations. Also, the diversion of much of the natural inflow to Lake Rotoiti to exit through the outlet before mixing with the lake basins, has complicated correctly estimating loads, flows and residence time. The Lake Rotoiti diversion wall has changed the retention time of the lake and diverted a substantial proportion of the load entering from Rotorua. McBride et al. (2021) attempted to take this into account in the estimation of the nutrient load and the flow, but the effect of the wall may have been estimated imperfectly.

Average outflow rates were estimated by CLUES modelling by McBride et al. (2021), using methods based on Woods et al. (2006, except for Lake Rotoiti, see above), and not by direct measurements. However, the hydrology of the Rotorua Te Arawa lakes, with large proportions of incoming and outgoing water via groundwater, makes these lakes extra challenging to model. Quite a few of the lakes have no continuous surface outflow and/or inflow and even those that do can have substantial groundwater throughflow. Therefore, important parameters in the models are hard to estimate. Flow rates may be under or overestimated affecting the residence time. If there are substantial inaccuracies in any of the input data (external loads, flows, residence time) then models cannot be expected to find a close a fit between predicted and observed lake concentrations.

Inaccuracy in input data does not change the fact that the models cannot sufficiently reproduce our best estimates; whether that is because of model error or error in the input data that the models use, or that their output is compared with, is irrelevant. Either way this means we cannot be certain that the models can be relied on to produce reasonably accurate estimates of the load reduction required to achieve TAS, at least not in units of kg y^{-1}.

It would be possible to adapt a model by fitting model parameters to the available data, however, fitting a model to the estimated model input data will do little good if the input data are inaccurate or are affected by management interventions. The resulting predictions would not be realistic. Moreover, while a new model based on the Rotorua Te Arawa lakes may fit relatively well (compared with the existing models used in this report) with the data used to create the model, typically when using that model on new data the fit will be less good. Building new models for small groups of lakes such as the Rotorua Te Arawa lakes, and without improvement of the input data, is not recommended.

Lastly, strong geothermal influences occur in several lakes, including proportionally large contributions to P loading in Lakes Rotomahana and Tarawera. Estimated geothermal P inputs were included in the estimates of external loads of McBride et al. (2021). However, the substantial uncertainty in the geothermal P inputs adds uncertainty to the estimates of external loading.

In summary, potential reasons for why the models did not perform well were:

1. Mass balance models typically do not perform well for individual lakes. This may well be the main reason.
2. The tested models may not be appropriate for the Rotorua Te Arawa lakes, perhaps because of underestimating nutrient retention in the New Zealand climate (section 3.11). However, the one model derived from New Zealand lake data performed worse on average.
3. Inaccuracies in the model input data, such as the external loads, inflow and outflow rates, residence time.
4. Lake management interventions, such as alum dosing and the diversion wall at the main inflow to Lake Rotoiti. Models are to be used on unmanipulated natural lake systems.
5. The steady state assumption underlying the mass balance models is more or less valid but not fully.
6. Uncertainty in the contributions of geothermal P loading, contributions which are thought to be large in several lakes.

4.3 Alternative approach

In view of the above, the best method to estimate the change in loads needed to achieve TAS is to apply the ratio of desired TAS to present concentration proportionally to the loads. In other words, the nutrient load should reduce by the same percentage as the in-lake concentrations to achieve TAS.

Brett and Benjamin (2008) demonstrated that changing the P loads should always have a directly proportionate impact on lake phosphorus concentrations, provided the type of phosphorus loaded (e.g., dissolved or particulate) does not vary.

Mass balance models expect nutrient retention as a proportion of the load to remain the same when the load is reduced, as nutrient retention depends on residence time or areal water loading rates and not on the loads.

4.4 Load reductions to achieve TAS for each lake attribute

Tables 9-10 show the proportions that nutrient loads need to be reduced by to achieve selected TAS values for annual median TP, TN, chlorophyll *a* and maximum chlorophyll *a*. These were derived from the proportions by which TAS values are lower than current concentrations (2026-2020). Annual median and maximum chlorophyll were also examined as derived from the relationships (Figure 3) with median TP (regressions with TN correlated similarly). The TAS for annual median and maximum chlorophyll were converted to median TP using these relationships and then the required nutrient load reduction estimated. However, the resulting suggested percentage changes were considered too uncertain. These proportions of the loads were highly variable, in some lakes substantially lower than the proportions of the desired change of TAS relative to current concentrations for the chlorophyll attributes, in other lakes substantially higher, and in a few cases the method suggested large reductions where the recent chlorophyll concentrations were less than the TAS values.

This is probably the result of a mismatch between TAS values for nutrients and chlorophyll. Therefore, these results were not used. Tables 11 and 12 show the reduced loads (for current loads see Table 6).

A caveat of external load reduction is the delay between the start of the load reduction and its effect on lake water quality, because of the time it takes for groundwater to reach the lakes, which in some locations can take decades. Naturally, this would affect the proportional approach in the same way as load reduction suggested by a mass balance model.

It is important to realize that if inflow rates to the lakes change, such as is expected in a changing climate, then nutrient concentrations in the inflows will change and hydraulic residence time will change and as a result the concentrations in the lake will change (Brett and Benjamin 2008). If inflow rates decrease then nutrient concentrations in the inflows are likely to increase, and in the lakes as well. Therefore, in a climate of decreasing rainfall a greater external load reduction will be required to meet the TAS.

Table 9: Percent reduction needed to achieve minimum TAS for annual median TP, TN, chlorophyll *a* and maximum chlorophyll *a*. Maximum change = the maximum change needed to achieve requirements for all four attributes, as derived from the required proportional change in the attributes. Non-zero values shaded and bold.

Lake	Median TP	Median TN	Median Chl-*a*	Maximum Chl-*a*	Maximum Change
Okareka	0	0	0	**8**	**8**
Okaro	**39**	0	0	**20**	**39**
Okataina	**3**	0	0	0	**3**
Rerewhakaaitu	**19**	**18**	**18**	**26**	**26**
Rotoehu	**35**	0	**44**	0	**44**
Rotoiti	0	0	**11**	**24**	**24**
Rotokakahi	0	0	**23**	0	**23**
Rotoma	**28**	6	**4**	**50**	**50**
Rotomahana	0	**6**	**19**	**34**	**34**
Rotorua	**15**	**2**	**17**	**41**	**41**
Tarawera	0	0	**31**	**16**	**31**
Tikitapu	**4**	0	**52**	**40**	**52**

Table 10: Percent reduction needed to achieve maximum TAS for annual median TP, TN, chlorophyll *a* and maximum chlorophyll *a*. Maximum change = the maximum change needed to achieve requirements for all four attributes, as derived from the required proportional change in the attributes. Non-zero values shaded and bold.

Lake	Median TP	Median TN	Median Chl-*a*	Maximum Chl-*a*	Maximum Change
Okareka	7	12	37	8	37
Okaro	80	73	80	87	87
Okataina	3	0	0	24	24
Rerewhakaaitu	28	18	59	63	63
Rotoehu	70	35	88	76	88
Rotoiti	55	14	70	50	70
Rotokakahi	34	25	49	32	49
Rotoma	28	6	4	50	50
Rotomahana	58	20	46	53	58
Rotorua	40	7	82	51	82
Tarawera	14	9	31	16	31
Tikitapu	4	6	52	40	52

Table 11: Reduced loads (kg y^{-1}) required to meet minimum TAS for median TP, TN, chlorophyll *a* and maximum chlorophyll *a*. Based on current loads in McBride et al. (2021). For current loads see Table 6. Reduced loads to achieve chlorophyll attributes and the maximum change are given in kg y^{-1} of P. Reduced loads are shaded and bold. Loads that do not require change are not shaded or bold.

Lake	Median TP	Median TN	Median Chl-*a*	Maximum Chl-*a*	Maximum Change
Okareka	985	14912	985	903	903
Okaro	385	4045	634	510	385
Okataina	1345	22821	1385	1385	1345
Rerewhakaaitu	879	12963	886	804	804
Rotoehu	2192	45418	1889	3370	1889
Rotoiti	4088	109876	3639	3093	3093
Rotokakahi	857	9847	661	857	661
Rotoma	1491	14395	1993	1026	1026
Rotomahana	20706	90308	16834	13597	13597
Rotorua	48001	704803	47213	33291	33291
Tarawera	10653	106767	7398	8927	7398
Tikitapu	112	2187	56	69	56

Table 12: Reduced loads (kg y^{-1}) required to meet maximum TAS for median TP, TN, chlorophyll *a* and maximum chlorophyll *a*. Based on current loads in McBride et al. (2021). For current loads see Table 6. Reduced loads to achieve chlorophyll attributes and the maximum change are given in kg y^{-1} of P. Reduced loads are shaded and bold. Loads that do not require change are not shaded or bold.

Lake	Median TP	Median TN	Median Chl-*a*	Maximum Chl-*a*	Maximum Change
Okareka	914	13183	621	903	621
Okaro	128	1074	126	85	85
Okataina	1345	22821	1385	1058	1058
Rerewhakaaitu	781	12963	443	402	402
Rotoehu	996	29591	420	816	420
Rotoiti	1822	94342	1213	2062	1213
Rotokakahi	565	7418	441	587	441
Rotoma	1491	14395	1993	1026	1026
Rotomahana	8792	76049	11223	9712	8792
Rotorua	34286	671241	10492	27742	10492
Tarawera	9177	97690	7398	8927	7398
Tikitapu	112	2063	56	69	56

4.5 Effect of nutrient load reduction on internal nutrient loading

We do not know what the annual lake wide gross internal loads are on average in each of the lakes, except Lake Rotorua. Even for the latter lake, the information dates from before alum dosing started and internal loading in Lake Rotorua may now be lower. We also cannot use the mass balance models to predict how internal loading will change when the external loads are reduced. However, a best guess is that in the long term the gross internal loads are proportional to the external loads. There is evidence that gross internal loads decrease in proportion with decreasing external loads, as discussed below.

A knowledge gap exists regarding rates of internal loading in lakes that are not very eutrophic. Typically, internal loading is studied only in very eutrophic lakes. However, in the vast majority of lakes there will be at least some level of recycling from the sediments. Therefore, internal loading should not be expected to stop entirely when external loads are reduced. As an example, in Lake Rotorua the gross internal P load is 88% (Burger et al. 2008) to 85% (this study) of the total P load. Therefore, the external load is only 12 to 15% of the current total load (external plus gross internal). It is unlikely that reducing the external load could bring the gross internal loading to a complete stop resulting in lake concentrations of less than 15% of the current concentrations. Such a relationship between load and in-lake concentration would also not agree with any of the mass balance models.

The linearity is supported by the mass balance models of OECD (1982). The lakes used to develop the mass balance model of OECD (1982; n = 87) ranged from oligotrophic to hypereutrophic. OECD (1982) did not consider anoxia as a criterion, that is, it did not remove lakes with anoxic bottom water from its data base. Vollenweider (OECD 1982) removed only lakes with the most extreme internal loading, where annual mean lake TP concentration was >$[TP]_{in}$. This would apply in none of the Rotorua Te Arawa lakes (Figure 6, Tables 4 and 6). The OECD (1982) data ranges for annual mean lake TP concentration (3-370 mg m^{-3}), $[TP]_{in}$ (5-1425 mg m^{-3}) and $[TP]_{in}/(1+\sqrt{T_W})$ (2-663 mg m^{-3}) encompass the full ranges among the Rotorua Te Arawa lakes (4-68, 23-338, 6-97 mg m^{-3}, respectively). The maximum annual mean lake TP concentration in the data set of OECD (1982) was more than five times higher than in Lake Okaro, the lake with highest P concentrations in the Bay of Plenty. OECD (1982) areal P loads ranged from 0.02 to 80 g P m^{-2} y^{-1} (average 1.2 g P m^{-2} y^{-1}), while those in the Rotorua Te Arawa lakes ranged from 0.08 to 2.30 g P m^{-2} y^{-1} (average 0.57 g P m^{-2} y^{-1}). Internal loading is likely to have occurred at the high end of these ranges in the OECD data set, which far surpassed trophic state conditions in the Rotorua Te Arawa lakes. Internal loading is implicitly accounted for by the model, with internal loading in equilibrium with external loading (van der Molen and Boers 1994). Nevertheless, the relationship between flushing corrected inflow concentrations of P and lake P concentrations in OECD (1982) was essentially linear, with data clustered around the 1:1 line, and did not show a sudden change for a midrange of loading values above which internal loading would kick in and below which no or substantially less internal loading would occur.

Gross internal loading decreased linearly with decreasing external loads in a data set of van der Molen and Boers (1994, Figure 44), further supporting the case for linearity.

In conclusion, the limited evidence presented here suggests that gross internal load likely changes proportionally to the external load; therefore, it would reduce by a similar proportion as the external load and the lake concentrations. A proportional decrease in the internal load following a reduction of the external load is also consistent with a retention factor that remains constant as a proportion of the external load, as mass balance models suggest. However, gross internal loading may reduce only slowly, and it may take a decade after reduction of the external load before a new equilibrium is reached (van der Molen and Boers 1994, Jeppesen et al. 2005).

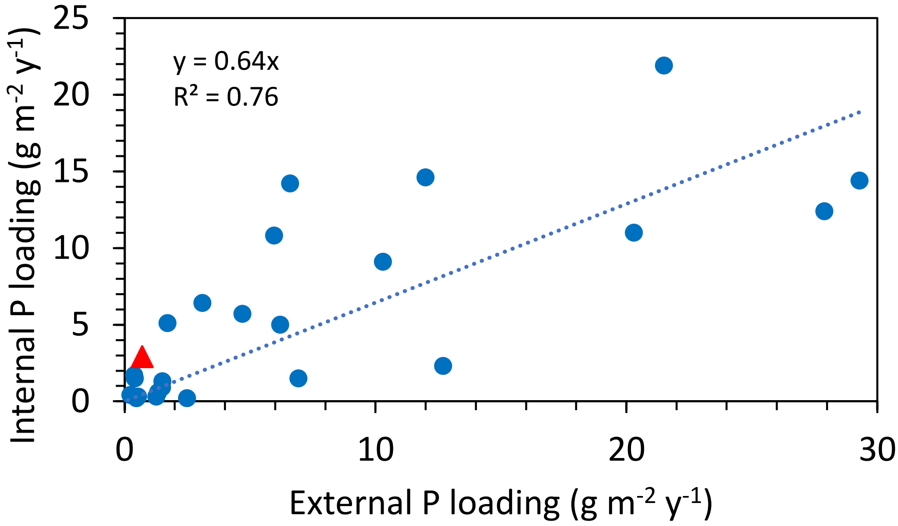


Figure 44: Gross internal loads increasing with external loads. In 24 lakes (blue dots, data from van der Molen and Boers 1994) and Lake Rotorua (red triangle; Burger et al. 2008).

5 Summary of main findings

The mass balance models and/or the available data for input to the models were not able to reproduce water quality observations versus estimated loads in the BOP lakes with reasonable accuracy, i.e., divergence of the model results were on average proportionally greater than the desired improvement in water quality.

The results suggested that the model of Abell et al. (2019) performs the least well when used in combination with the best estimates for current nutrient loads (areal export coefficient method), outflow rates, lake morphometry and lake residence times (McBride et al. 2021). That is surprising because the Abell et al. model is the only model that was developed for New Zealand lakes.

Because the deviations of each of the models from observations were on average greater than the desired reductions in in-lake concentrations, and because of the large difference in load estimates between models, the best approach to determine the change in loads to the lake required to achieve TAS for each variable is to adopt a load reduction proportional to the decrease in each variable relative to present water quality. A linear relation between water quality improvement and load reduction is reasonable because for individual lakes (when variables such as long-term average residence time are in fact taken as constants) the models are all either linear or indistinguishable from linear (excepting the Abell et al. model which fitted least well).

In the long term a water quality improvement is expected proportional to the external load reduction, including in lakes where internal loading may occur. However, in lakes where internal loading is substantial it may take a decade after reduction of the external load before a new equilibrium is reached (van der Molen and Boers 1994, Jeppesen et al. 2005). On the other hand, because of the lag time involved in groundwater reaching the lakes, with lag time varying with location in the catchments of each lake, it may take an unknown number of years as well for changes in land use leading to reductions in the external load to have a noticeable effect on lake water quality.

6 Recommendations

- Review selected values for TAS and the resulting effects on load reduction requirements, particularly around the mismatch of TAS between annual median TN, TP, chlorophyll and maximum chlorophyll.

- Adopt reductions in loadings proportional to the desired improvement in water quality, as presented in Tables 9-12.

Additional investigations:

- Fit observed retention of N and P against T_W and q_S, respectively, for a sufficiently large data base of New Zealand lakes in which no internal loading occurs, to derive relationships appropriate for the New Zealand climate. This can improve predicted retention because in New Zealand lakes typically do not freeze over, enabling higher annual average retention than in higher latitude lakes. This will allow simple mass balance estimates of internal loads from the divergence from expected retention where no internal loading occurs.

- Measure in situ internal loading to compare with internal loading as estimated from the mass balance and from the P/Fe ratio in the sediments.

- Improve where possible upon estimates of model input data as given by McBride et al. (2021). loading to lakes. This relates especially to the N and P loads, outflow rates, and residence time data. A review of the data of lakes Okaro, Rerewhakaaitu, Rotomahana should be focussed on first because the data for these lakes resulted in negative denitrification estimates, indicating problems with the input data used in the mass balance models. In addition, a review of the outflow rate of Lake Rotoiti would be useful in view of its unexpectedly low observed P retention. It is possible that, although the effect of the diversion wall in Lake Rotoiti has been taken into account (McBride et al. 2021), its outflow might be underestimated and therefore its residence time might be overestimated. On the other hand, the low observed P retention may be the result of internal loading.

- Monitor inflow and outflow rates in all lakes and where possible determine groundwater flows through lakes.

- Because monitoring multiple inflows is more difficult than a single outflow, inflow rates can be estimated from the water balance as done in this report. To improve the inflow estimates the requirements are 1) estimates of annual rainfall on each lake to replace the long-term averages used in this report 2) annual mean evaporation estimates for each lake which can be derived from buoy based meteorological data. This will improve the resolution of lake evaporation between lakes because this report uses only a single estimate, from Rotorua, and in addition this estimate is likely to be biased because it is land-based.

- Monitor inflow nutrient concentrations and use this, together with the flow rates, to derive annual nutrient loads with annual resolution, to replace the single values in McBride et al. (2021).

- Collect surface layer water (epilimnion) samples and analyse composition of phytoplankton by recording cell counts and biovolume by species, in each of the lakes, at least once every two months. Knowledge of phytoplankton species distributions, their differences between lakes and change across the year, is important because some species are more desirable than others. This will allow verification when and where cyanobacteria are the dominant group among the phytoplankton. In the nine lakes where phytoplankton biomass peaks in winter the cyanobacteria are not expected to dominate during biomass peaks but this must be verified. Cyanobacteria are more likely to dominate high biomass in the three lakes where phytoplankton biomass peaks in summer. This may be a concern because phytoplankton biomass on average tended to be highest in lakes where it peaks in summer.

7 Acknowledgements

We thank Sanjay Wadhwa for providing the VCSN rainfall data for the lake catchments, James Dare for the lake nutrients concentrations data base and Deniz Ozkundakci for comments.

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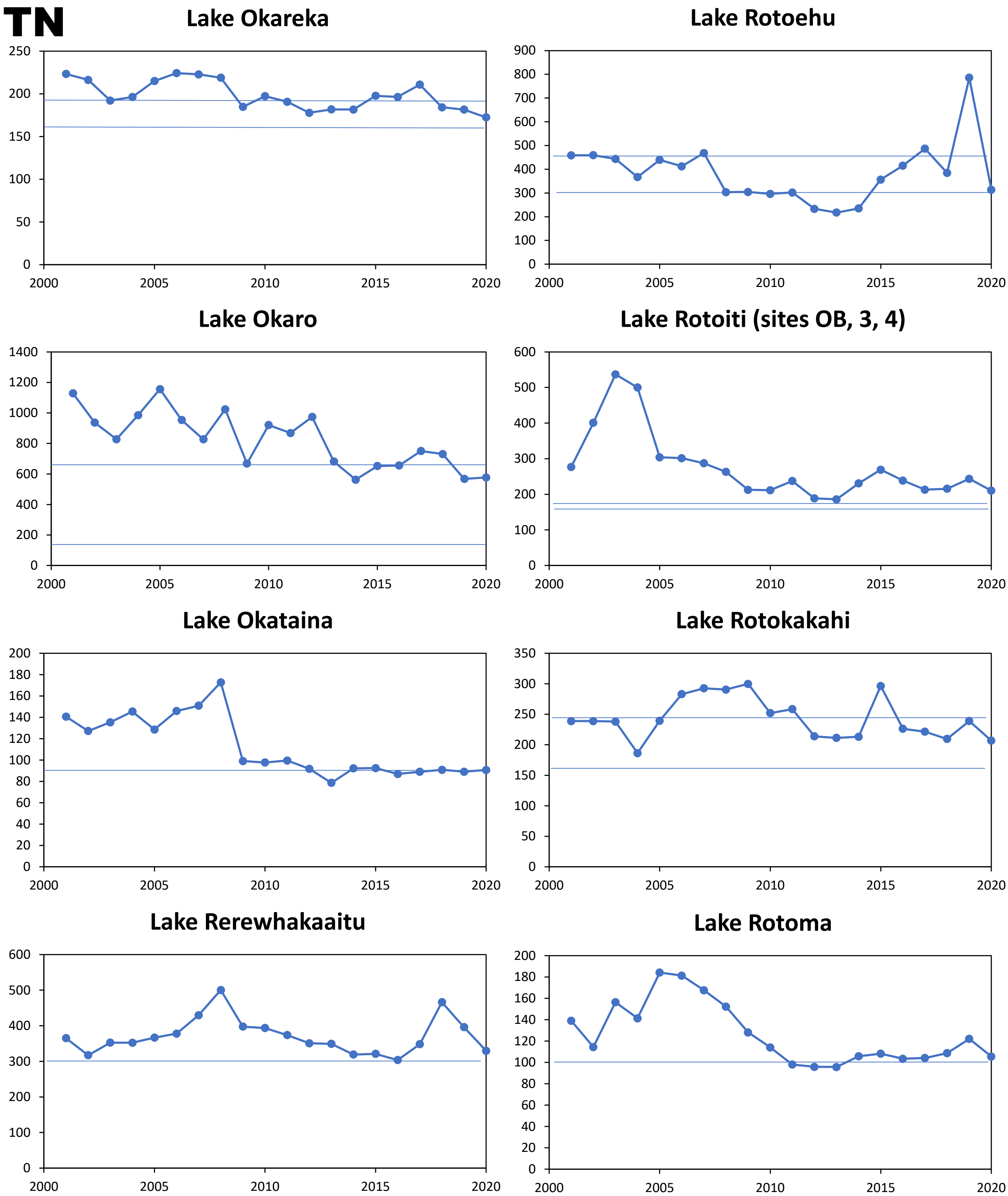
Appendix A Annual mean TN, TP and chlorophyll a concentrations in each lake since 2000

The analysed database includes TP and DRP values from the new silica blocking method (for TP and DRP) with a retrospective adjustment for historical data (carried out by Keith Hamill).

Figures of annual mean TN, TP and chlorophyll *a* concentrations in each lake since 2000 (mg m^{-3}) are on the following pages. In some of the figures the data are averaged across several sites (indicated in the figure). Horizontal lines represent the minimum and maximum TAS.

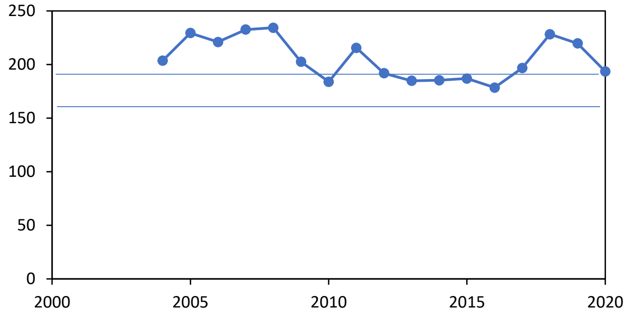
Summary of monitoring data:

- TN decreased in a large number of lakes: Okareka, Okaro, Okataina, Rotoiti, Rotoma, Rotorua, Tarawera, and Tikitapu.
- TN did not show a consistent increase in any lake.
- TP decreased in Okaro (apart from very high values in 2019), Rotomahana, and Rotorua.
- TP increased in Okareka, and Rerewhakaaitu.
- Chlorophyll *a* decreased in Okaro (since 2005), Rotoma, and Rotorua.
- Chlorophyll *a* increased in Rerewhakaaitu, and in no other lake.
- Secchi depth increased in Okaro, Rotoiti, and Rotorua.
- Secchi depth did not show a consistent decrease in any lake.
- TN:TP decreased in Okareka, Okataina, Rerewhakaaitu, Rotoiti, Rotokakahi, Rotoma, and Tarawera.
- TN:TP increased in Rotorua only.
- Chla:TN decreased in Rotorua only.
- Chla:TN increased in Rerewhakaaitu, Tarawera and Rotoiti (since 2005).
- Chla:TP decreased in Okareka (but not since 2005), and Rotoiti (slightly).
- Chla:TP increased in Okataina (since 2004), Rotoehu (since 2005), and Tikitapu (since 2005), No significant overall trend in Lake Rotorua while negative trend in Chla : TN, suggesting chlorophyll *a* concentrations were (primarily) determined by phosphorus concentrations.

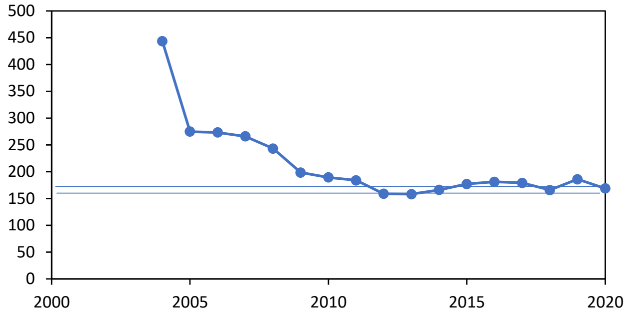

TN
Lake Okareka
0 50 100 150 200 250
2000 2005 2010 2015 2020
Lake Rotoehu
0 100 200 300 400 500 600 700 800 900
2000 2005 2010 2015 2020
Lake Okaro
0 200 400 600 800 1000 1200 1400
2000 2005 2010 2015 2020
Lake Rotoiti (sites OB, 3, 4)
0 100 200 300 400 500 600
2000 2005 2010 2015 2020
Lake Okataina
0 20 40 60 80 100 120 140 160 180 200
2000 2005 2010 2015 2020
Lake Rotokakahi
0 50 100 150 200 250 300 350
2000 2005 2010 2015 2020
Lake Rerewhakaaitu
0 100 200 300 400 500 600
2000 2005 2010 2015 2020
Lake Rotoma
0 20 40 60 80 100 120 140 160 180 200
2000 2005 2010 2015 2020

TN

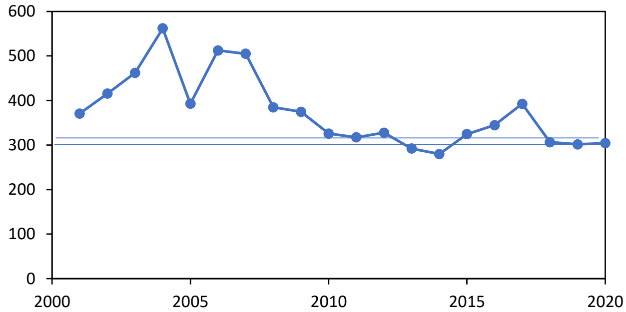
Lake Rotomahana (site 2)



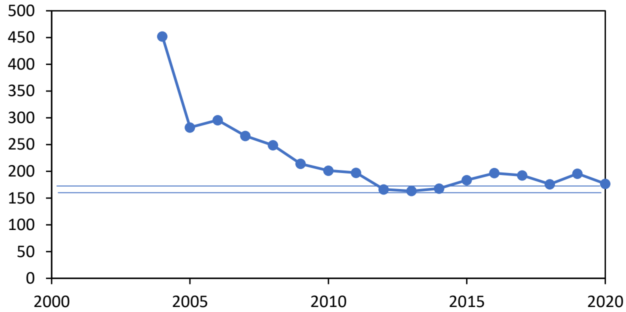
Lake Rotoiti (sites 3, 4)



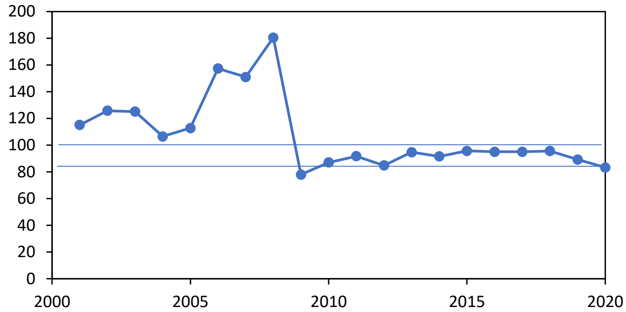
Lake Rotorua (sites 2, 5)



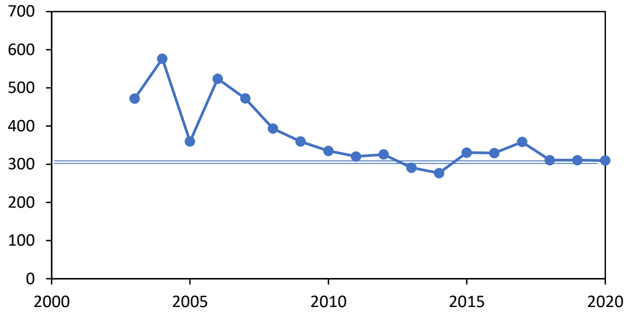
Lake Rotoiti (site 3)



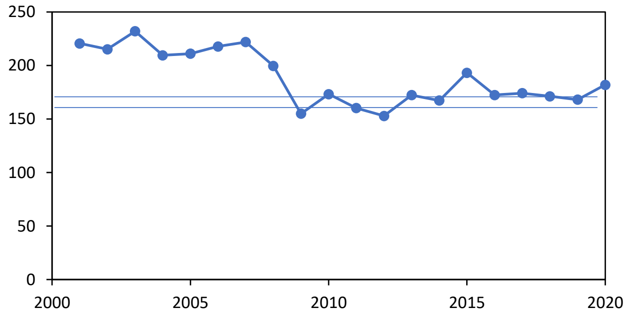
Lake Tarawera



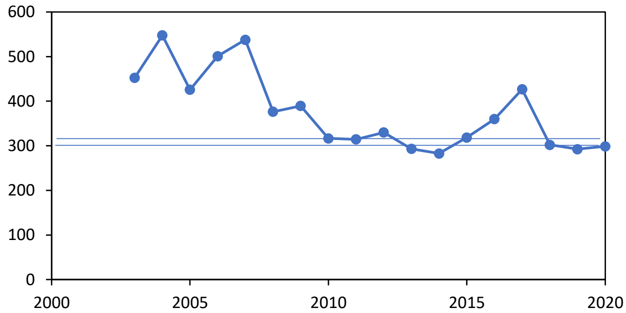
Lake Rotorua (site 2)



Lake Tikitapu

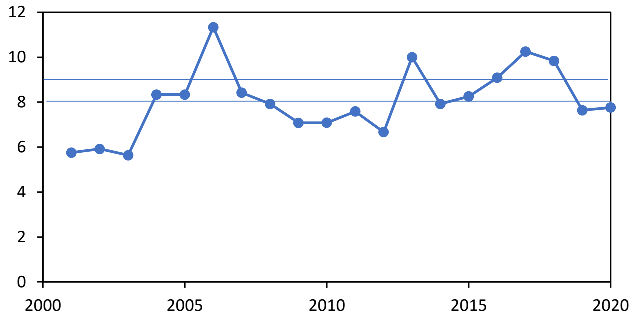


Lake Rotorua (site 5)

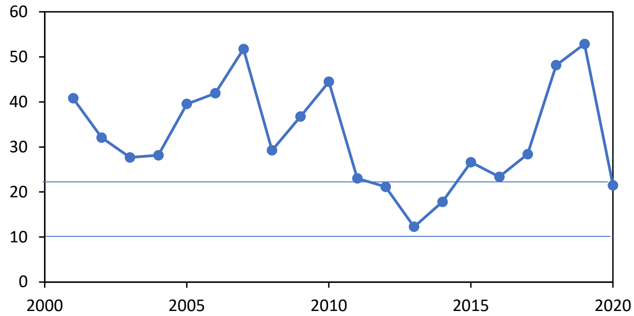


TP

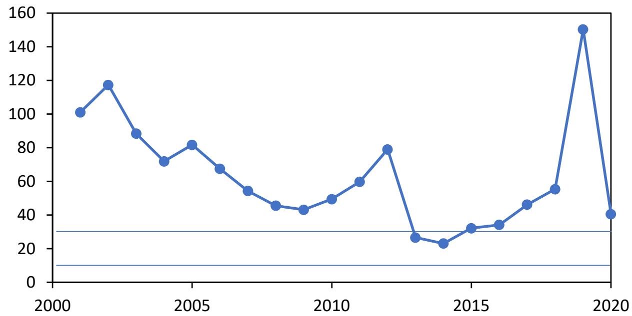
Lake Okareka



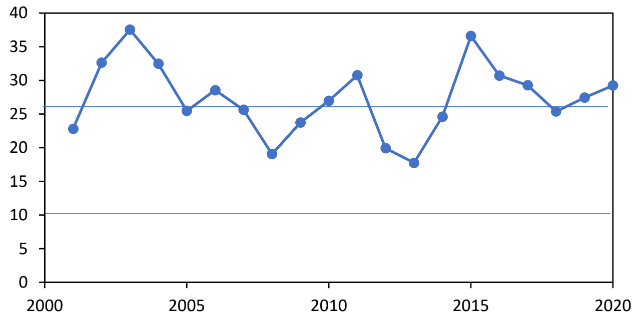
Lake Rotoehu



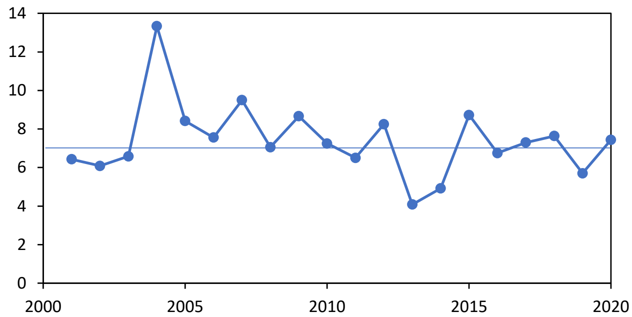
Lake Okaro



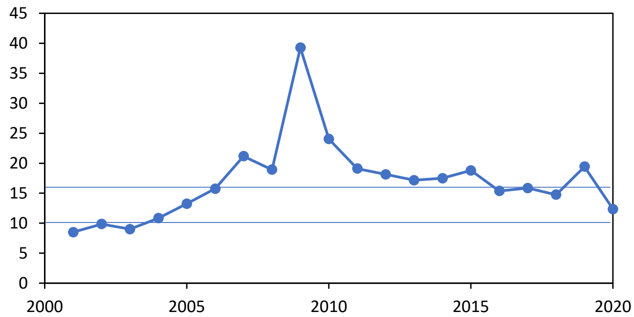
Lake Rotoiti (sites OB, 3, 4)



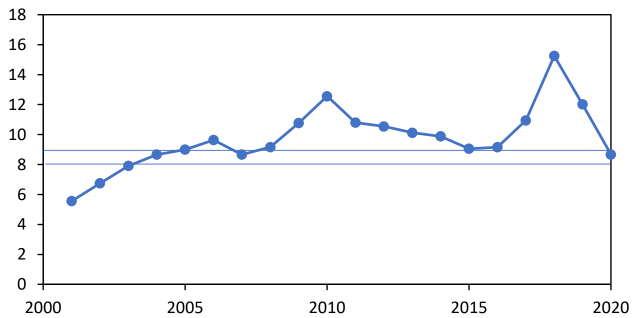
Lake Okataina



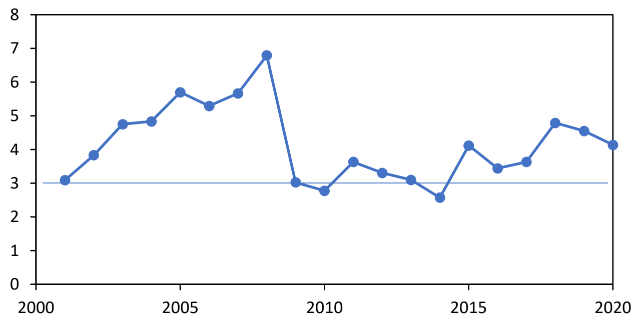
Lake Rotokakahi



Lake Rerewhakaaitu

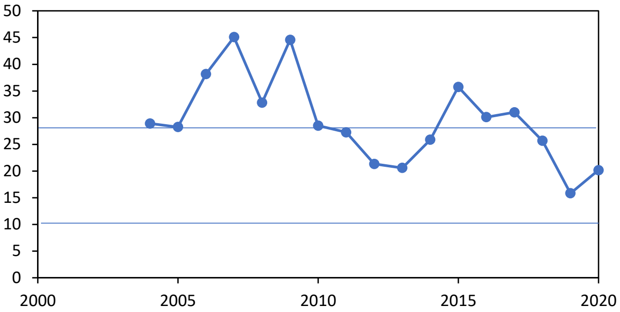


Lake Rotoma

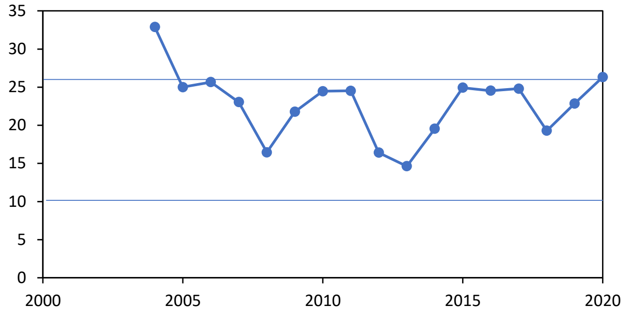


TP

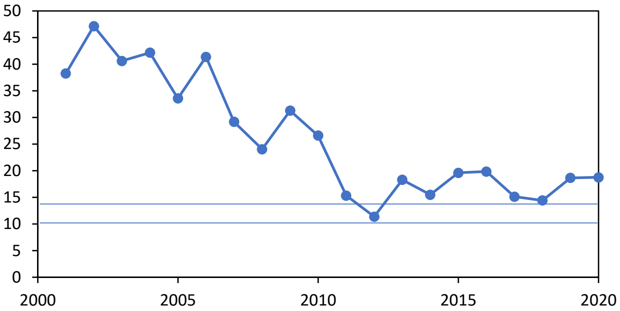
Lake Rotomahana (site 2)



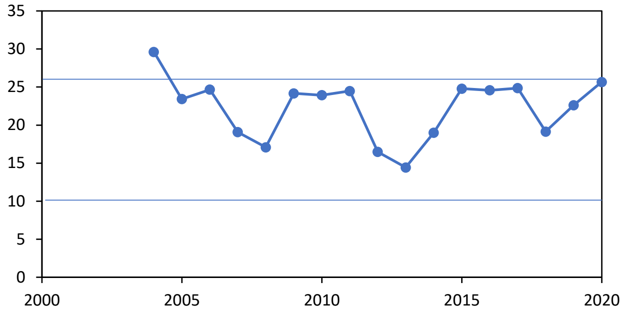
Lake Rotoiti (sites 3, 4)



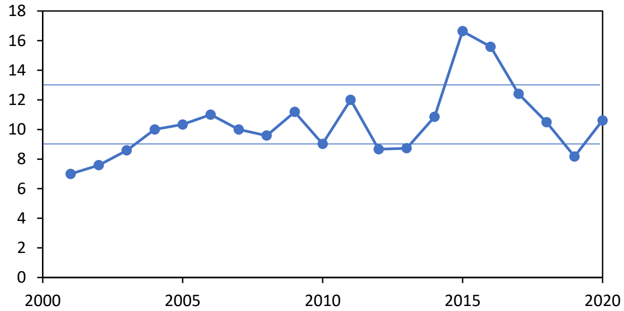
Lake Rotorua (sites 2, 5)



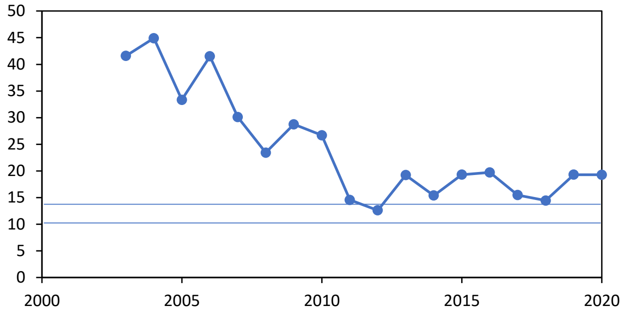
Lake Rotoiti (site 3)



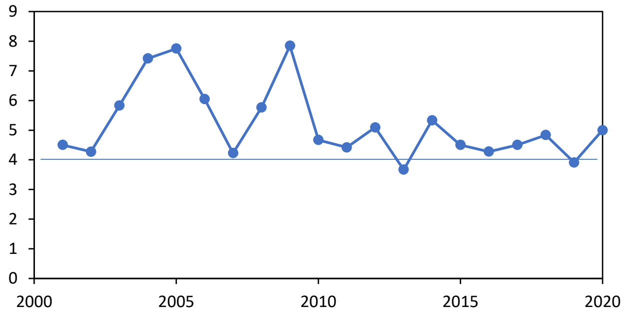
Lake Tarawera



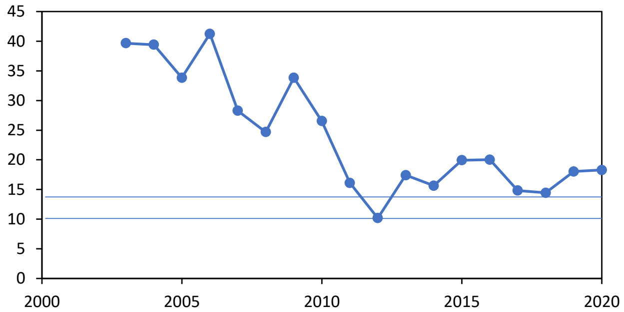
Lake Rotorua (site 2)



Lake Tikitapu

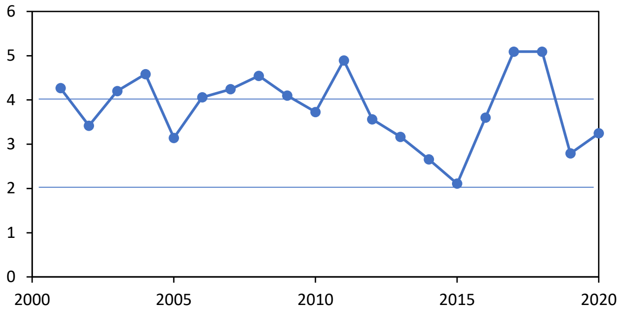


Lake Rotorua (site 5)

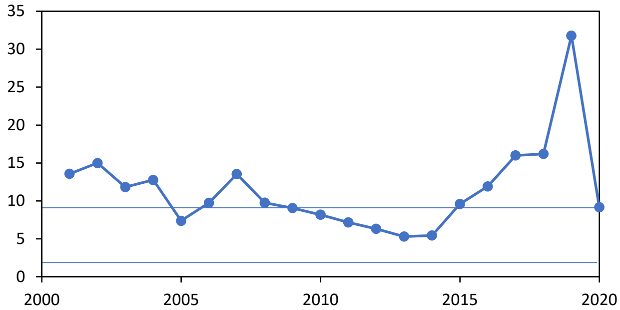


Chl-α

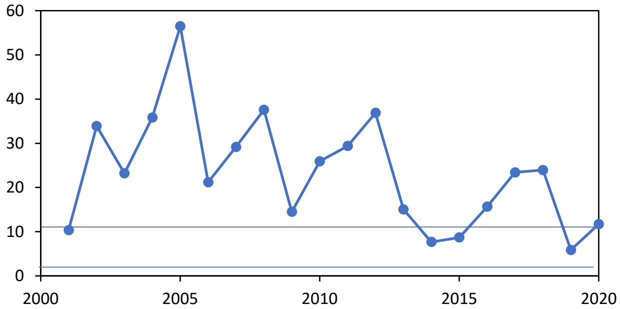
Lake Okareka



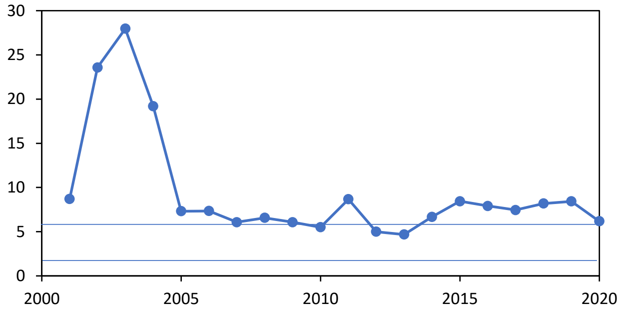
Lake Rotoehu



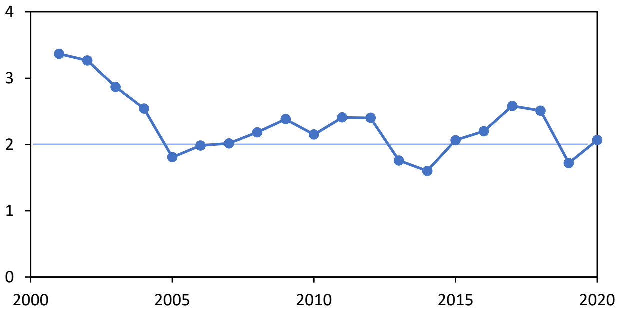
Lake Okaro



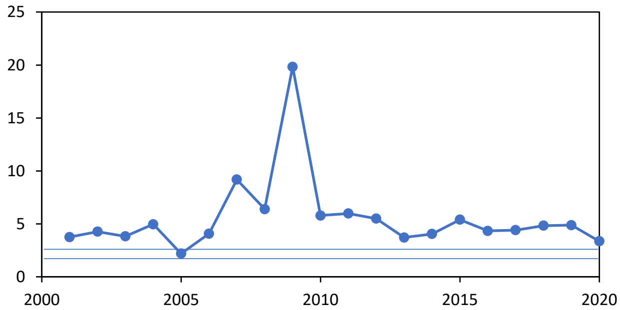
Lake Rotoiti (OB, site 3, 4)



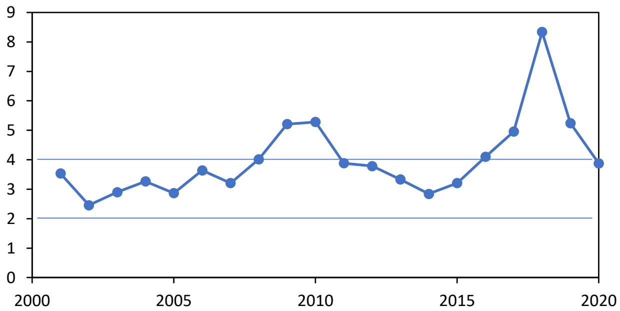
Lake Okataina



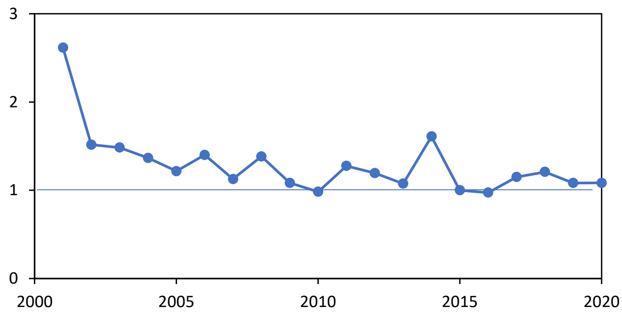
Lake Rotokakahi



Lake Rerewhakaaitu

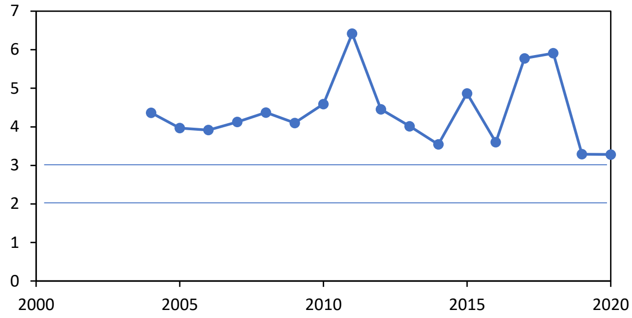


Lake Rotoma

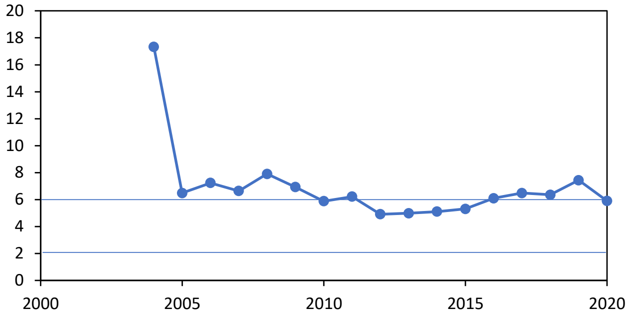


Chl-α

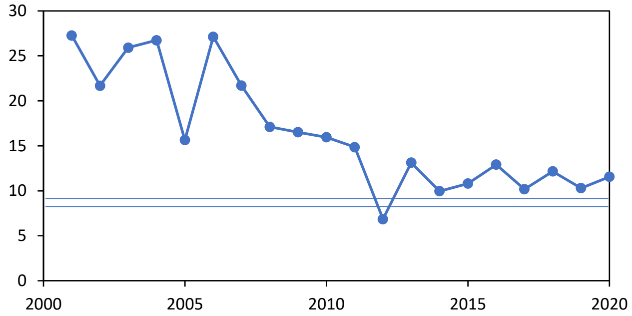
Lake Rotomahana (2)



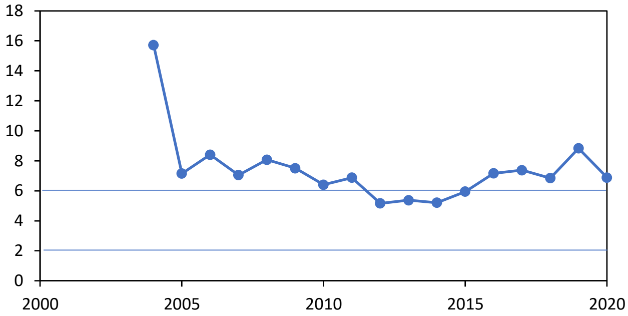
Lake Rotoiti (3, 4)



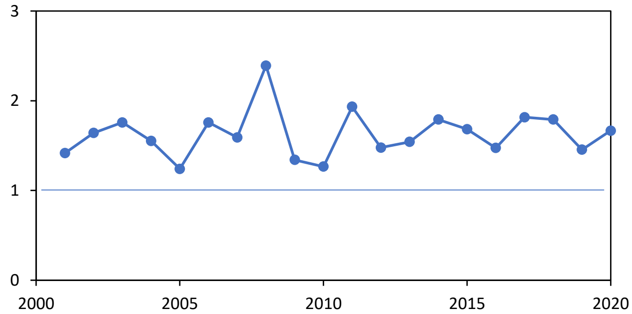
Lake Rotorua (2, 5)



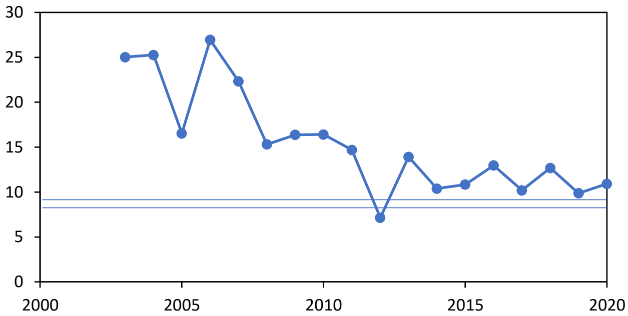
Lake Rotoiti (3)



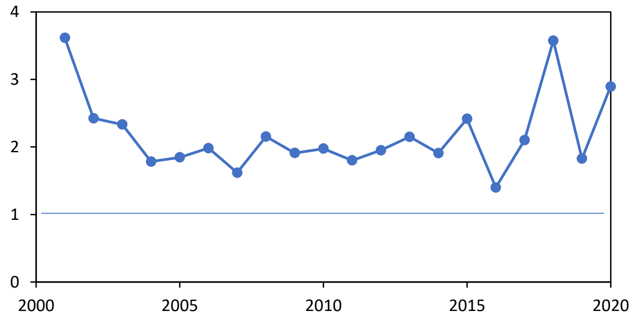
Lake Tarawera



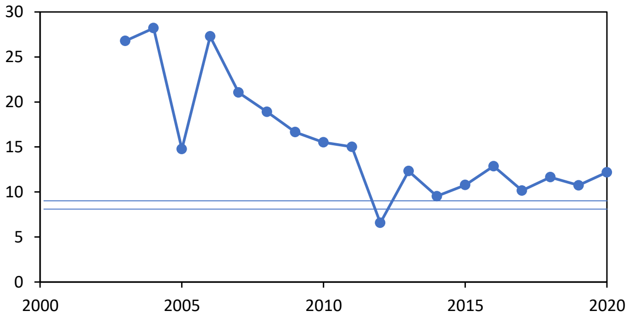
Lake Rotorua (2)



Lake Tikitapu

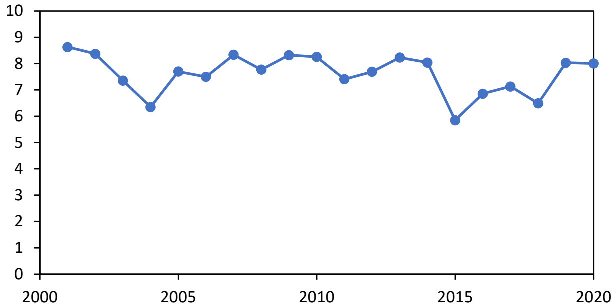


Lake Rotorua (5)

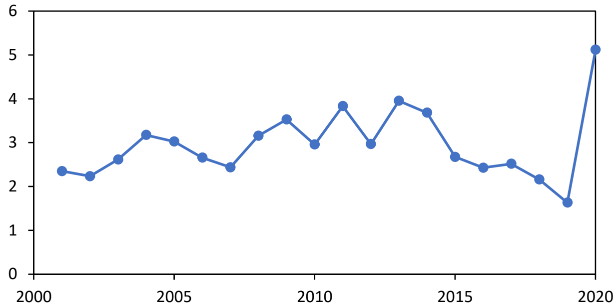


Secchi

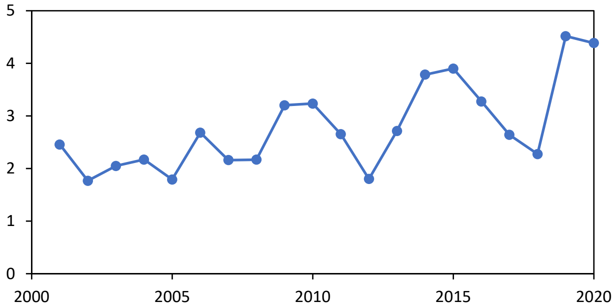
Lake Okareka



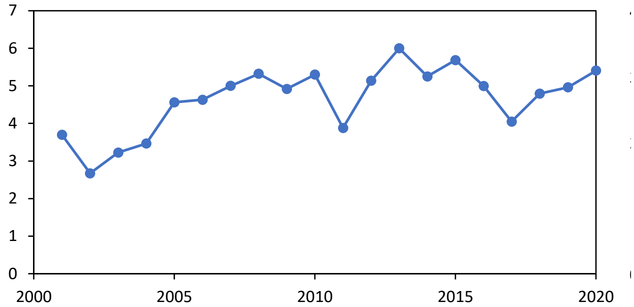
Lake Rotoehu



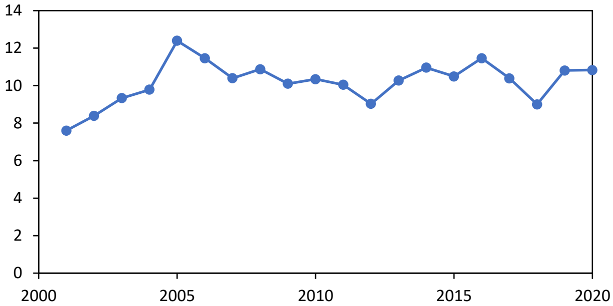
Lake Okaro



Lake Rotoiti (OB, site 3, 4)

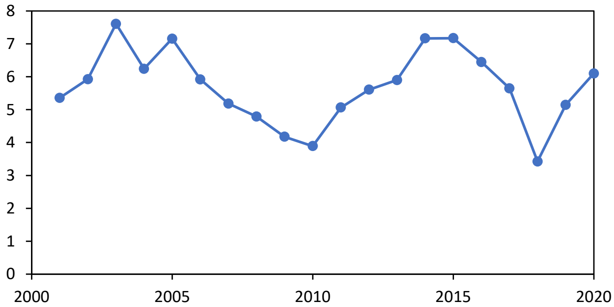


Lake Okataina

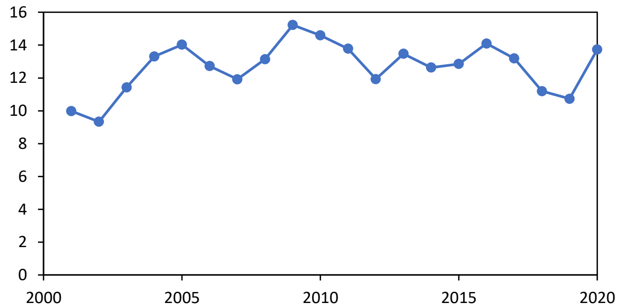


Lake Rotokakahi

Lake Rerewhakaaitu

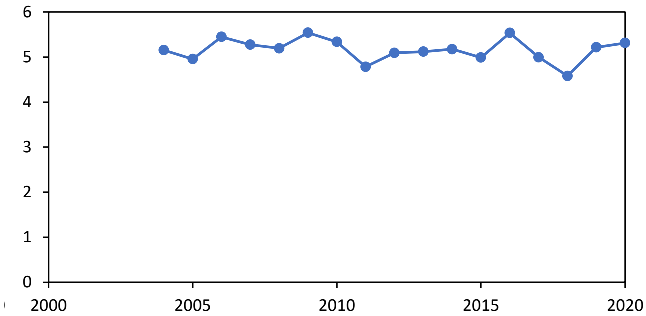


Lake Rotoma

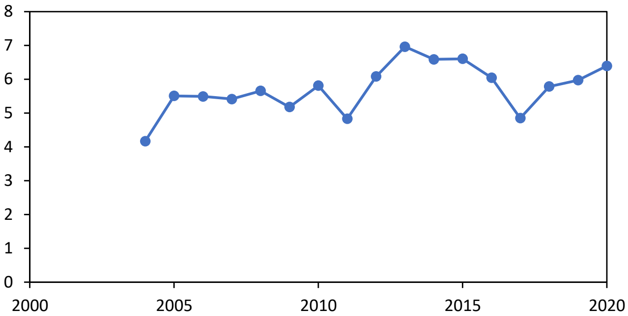


Secchi

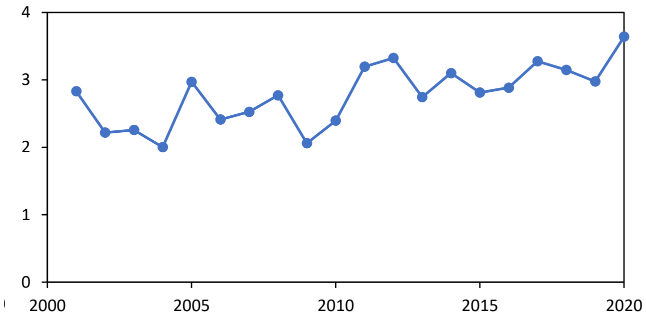
Lake Rotomahana (2)



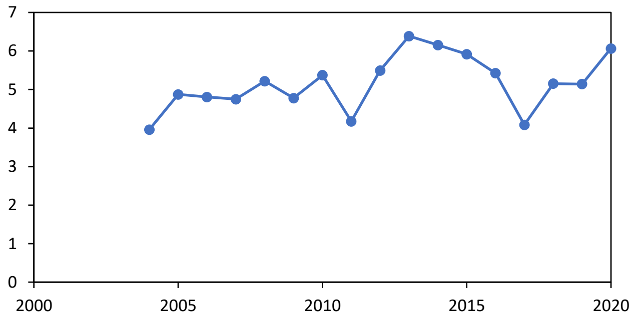
Lake Rotoiti (3, 4)



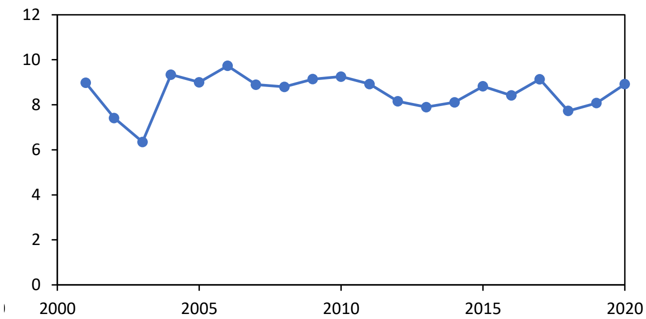
Lake Rotorua (2, 5)



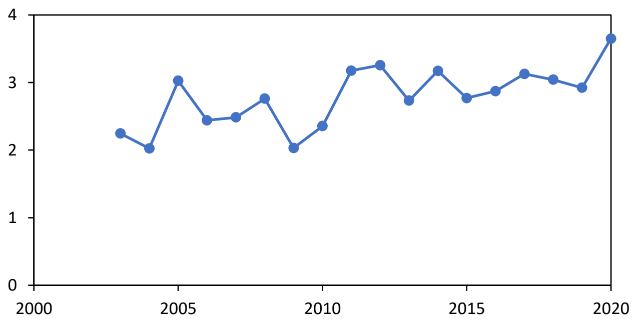
Lake Rotoiti (3)



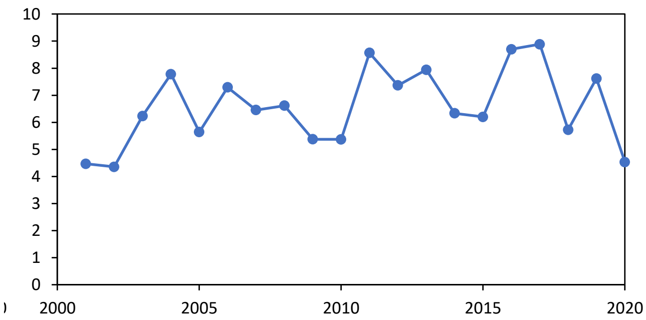
Lake Tarawera



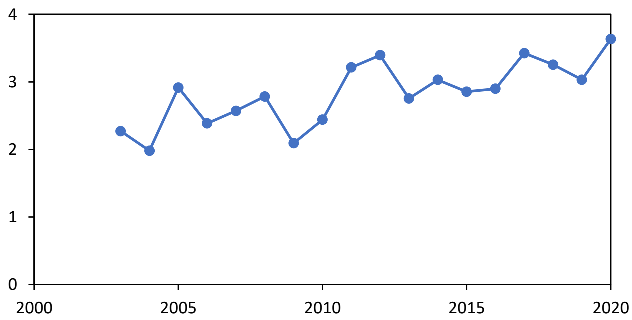
Lake Rotorua (2)



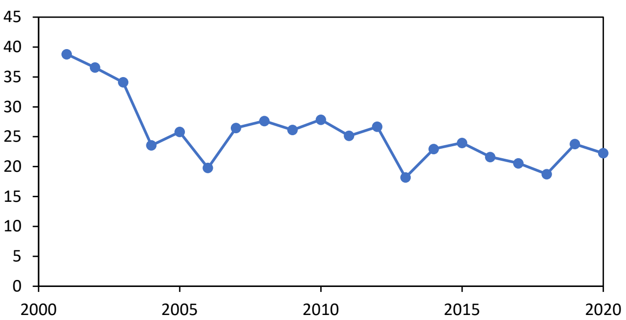
Lake Tikitapu

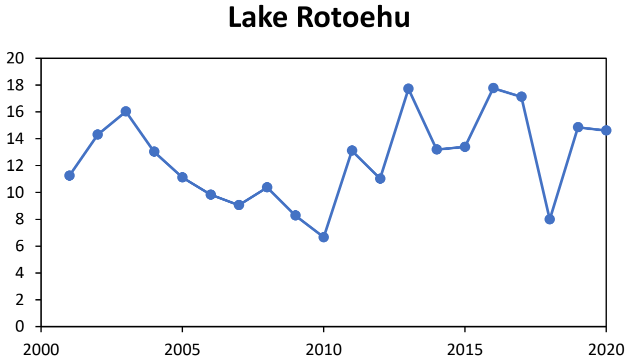


Lake Rotorua (5)



TN:TP


Lake Okareka
45
40
35
30
25
20
15
10
5
0
2000
2005
2010
2015
2020


Lake Rotoehu
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14
12
10
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6
4
2
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2005
2010
2015
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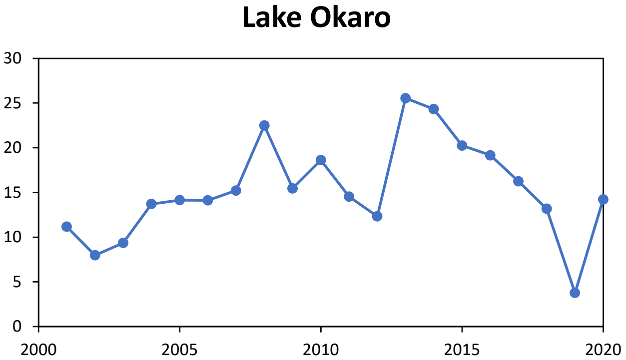
Lake Okaro
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25
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10
5
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2000
2005
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2015
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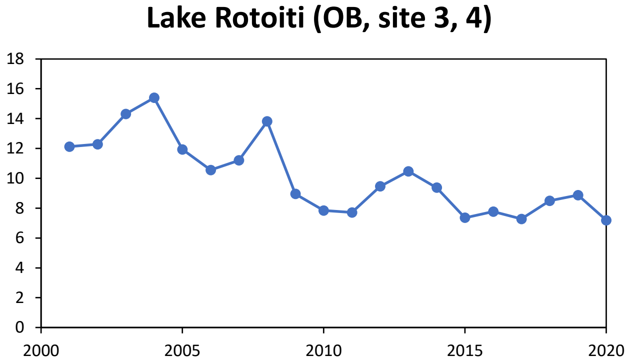
Lake Rotoiti (OB, site 3, 4)
18
16
14
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10
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2000
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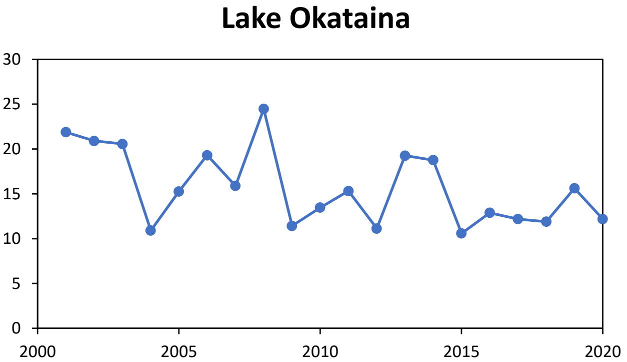
2010
2015
2020


Lake Okataina
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25
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5
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2005
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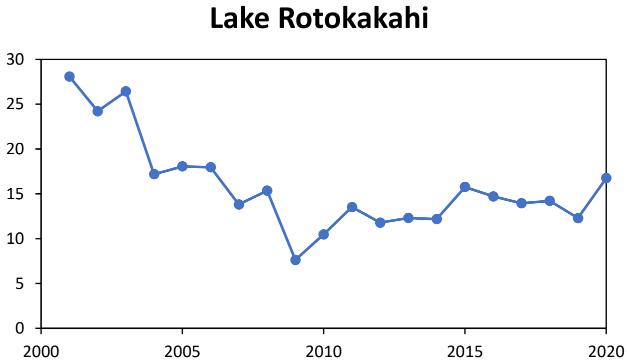
2020


Lake Rotokakahi
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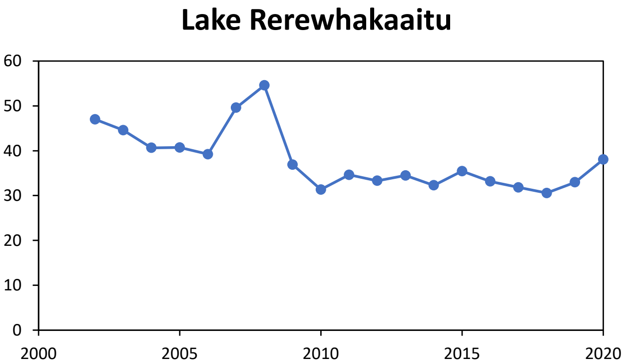
Lake Rerewhakaaitu
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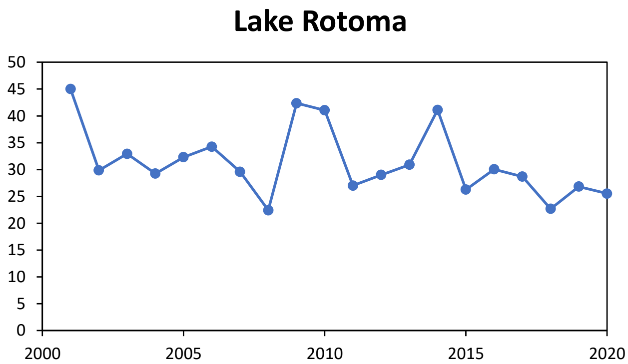
Lake Rotoma
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45
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0
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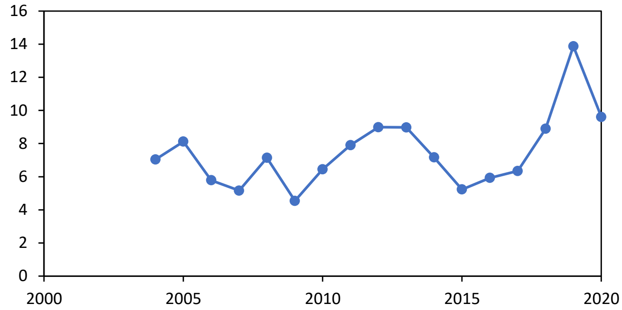
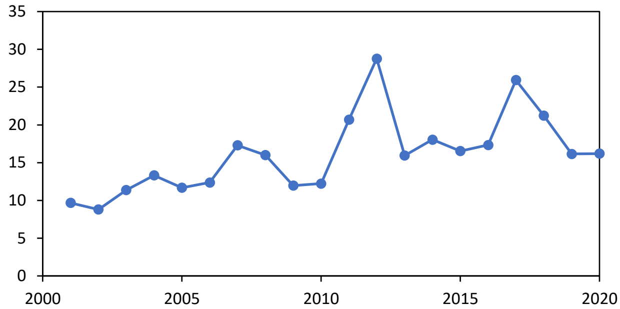
2005
2010
2015
2020

TN:TP Lake Rotomahana (2)

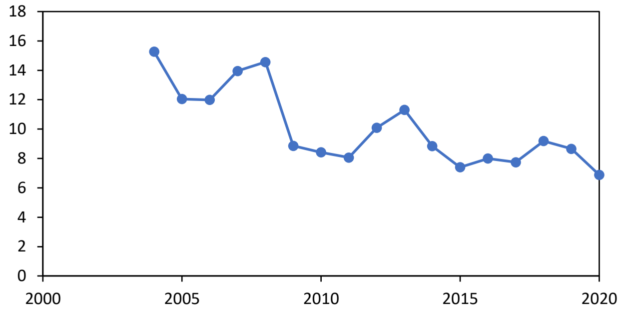


Lake Rotoiti (3, 4)

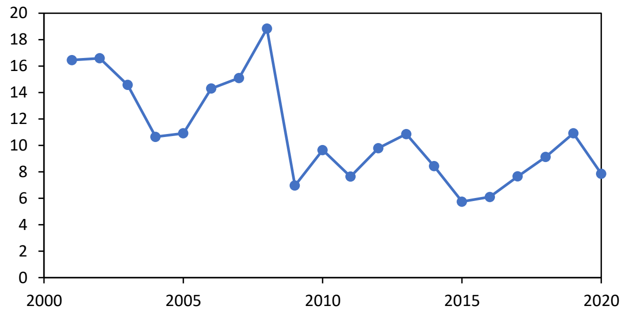
Lake Rotorua (2, 5)



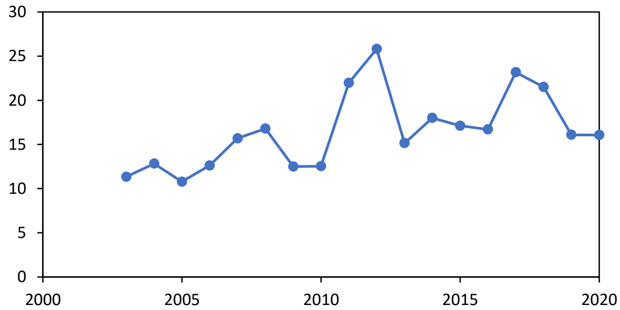
Lake Rotoiti (3)



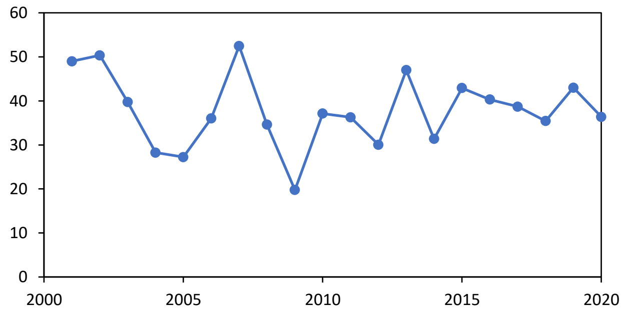
Lake Tarawera



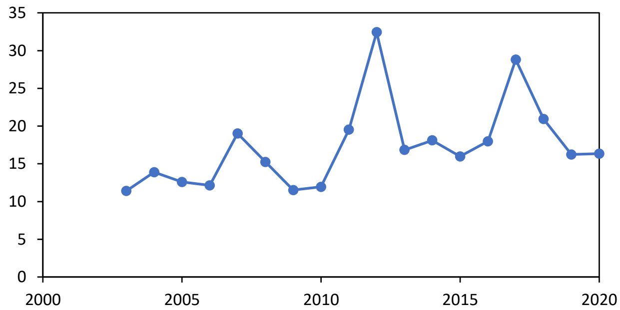
Lake Rotorua (2)



Lake Tikitapu

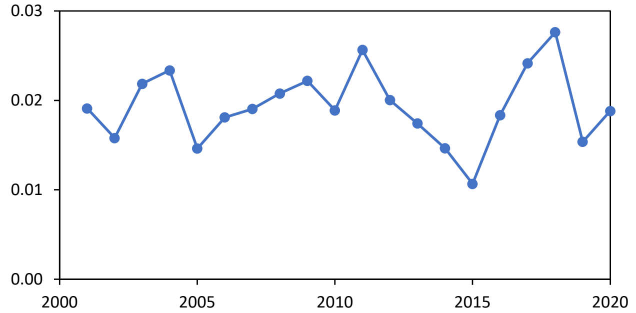


Lake Rotorua (5)

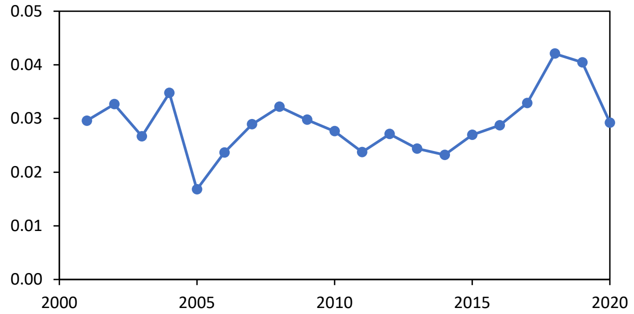


Chla:TN

Lake Okareka

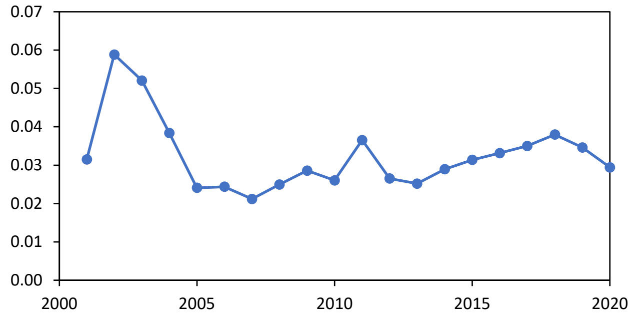


Lake Rotoehu

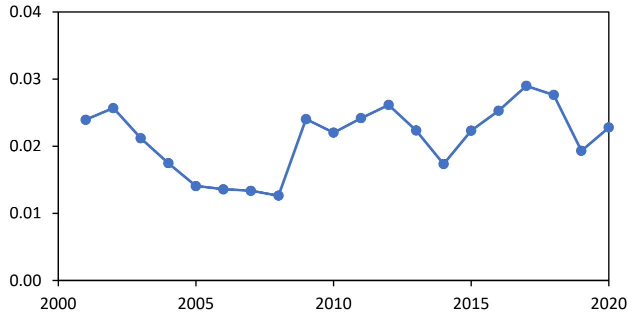


Lake Okaro

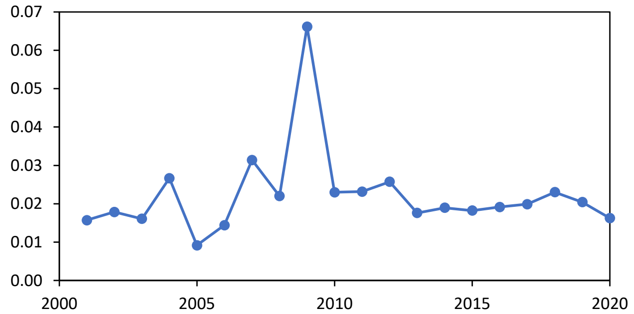
Lake Rotoiti (OB, site 3, 4)



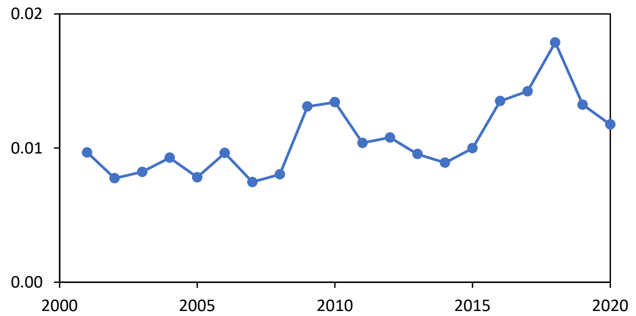
Lake Okataina



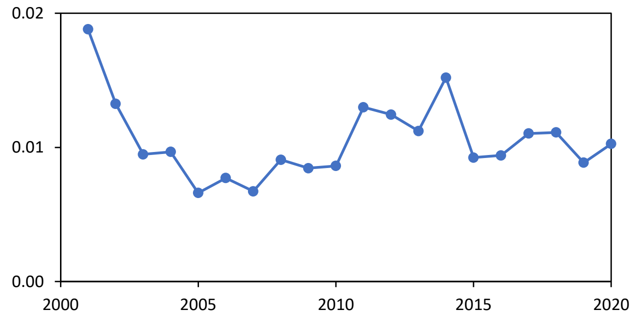
Lake Rotokakahi

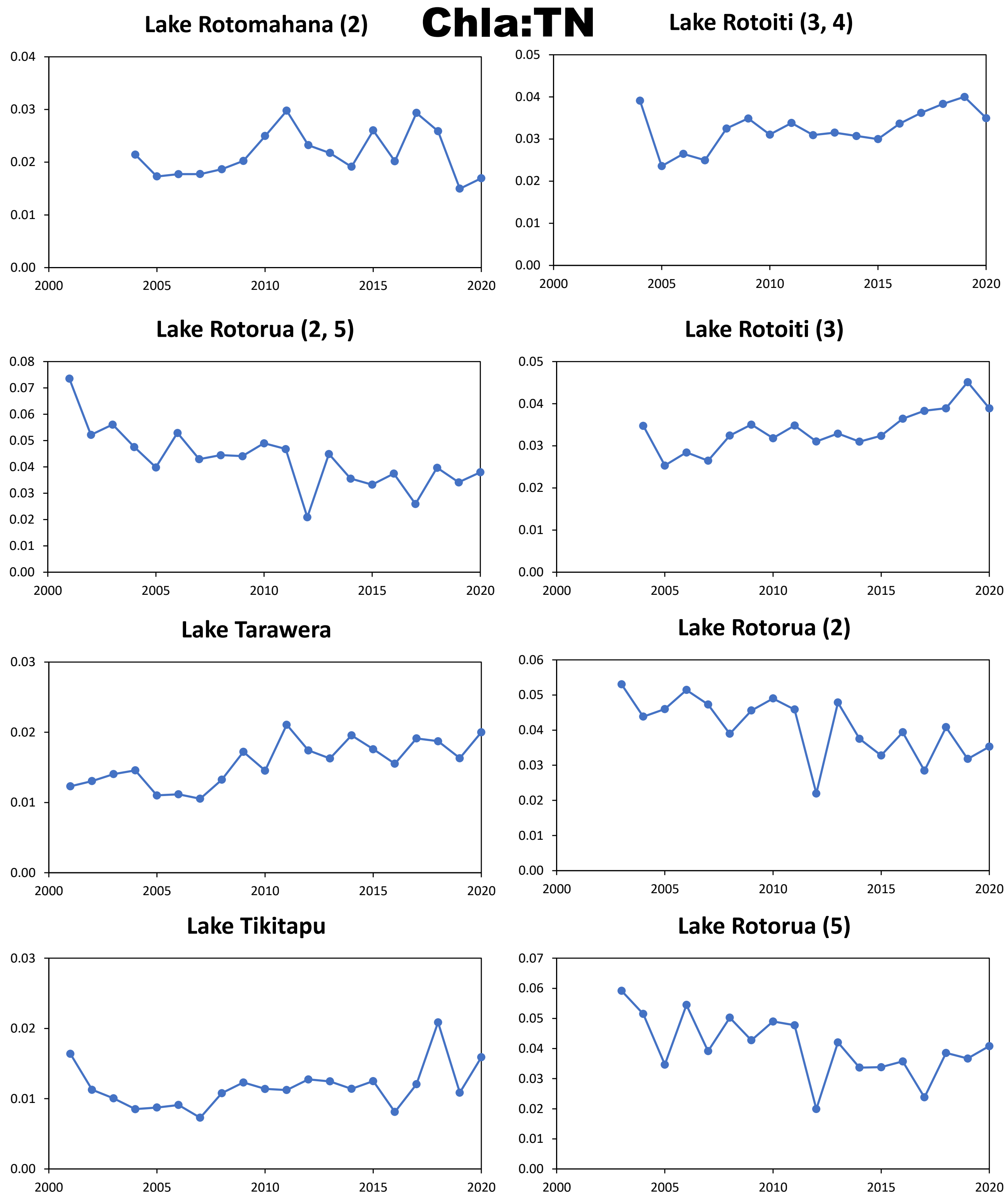


Lake Rerewhakaaitu

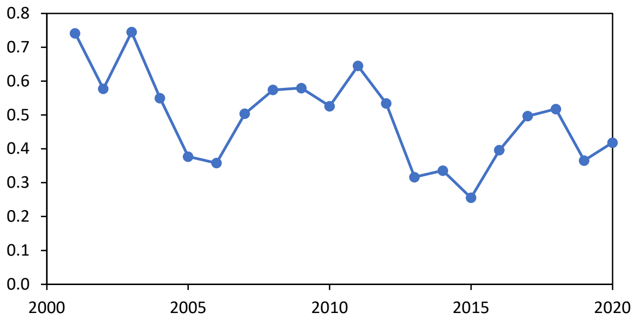


Lake Rotoma




Chla:TN
Lake Rotomahana (2)
Lake Rotoiti (3, 4)
Lake Rotorua (2, 5)
Lake Rotoiti (3)
Lake Tarawera
Lake Rotorua (2)
Lake Tikitapu
Lake Rotorua (5)
2000
2005
2010
2015
2020

Chla:TP


Lake Okareka
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0.2
0.3
0.4
0.5
0.6
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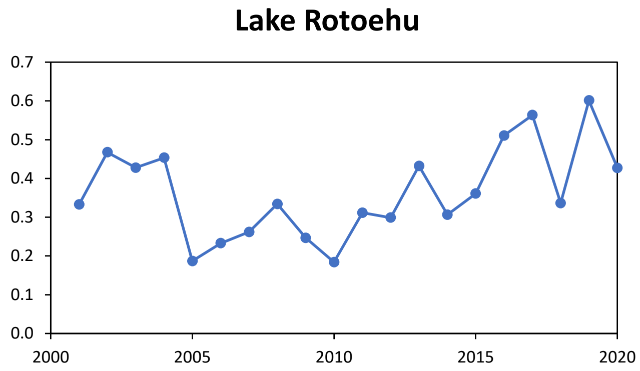
2005
2010
2015
2020


Lake Rotoehu
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2005
2010
2015
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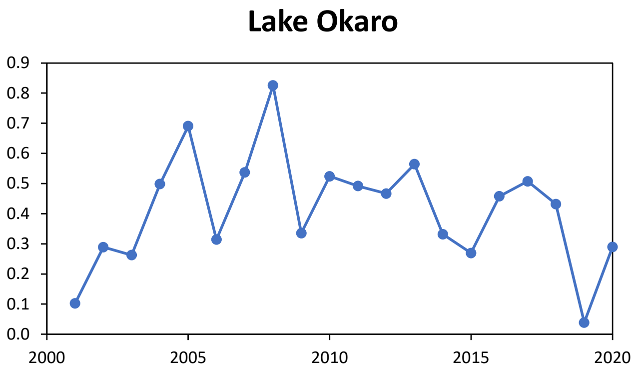
Lake Okaro
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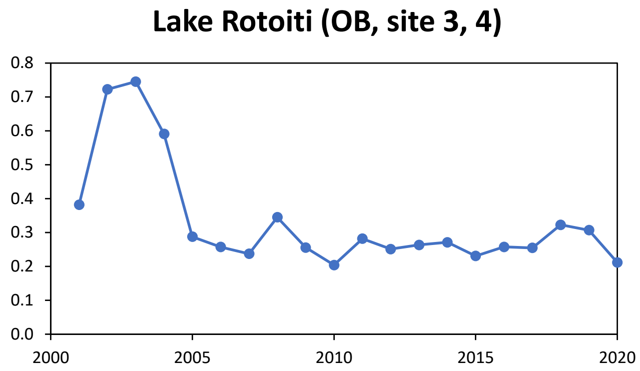
2000
2005
2010
2015
2020


Lake Rotoiti (OB, site 3, 4)
0.0
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0.2
0.3
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0.7
0.8
2000
2005
2010
2015
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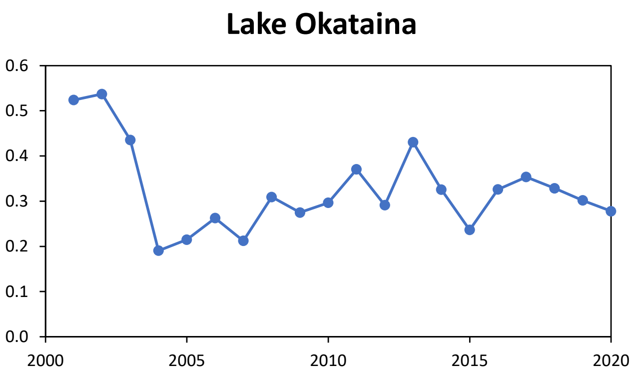
Lake Okataina
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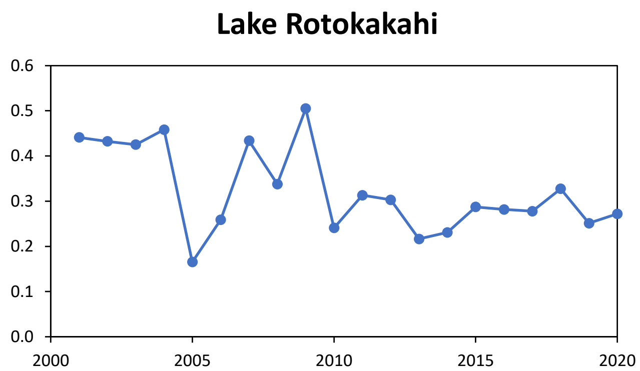
Lake Rotokakahi
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2015
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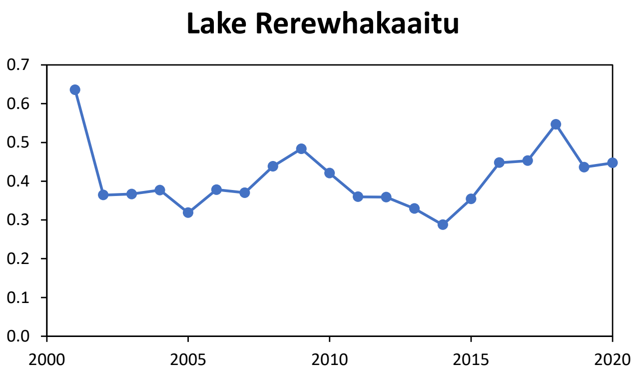
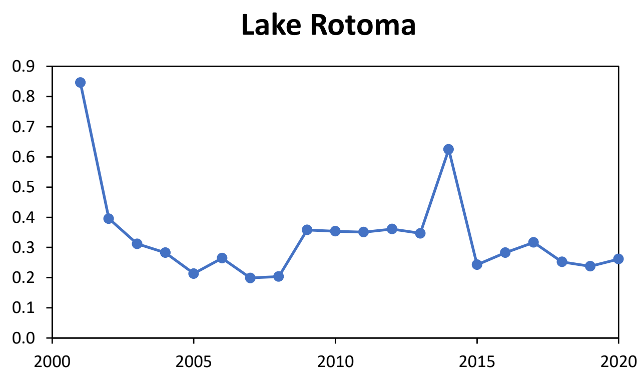
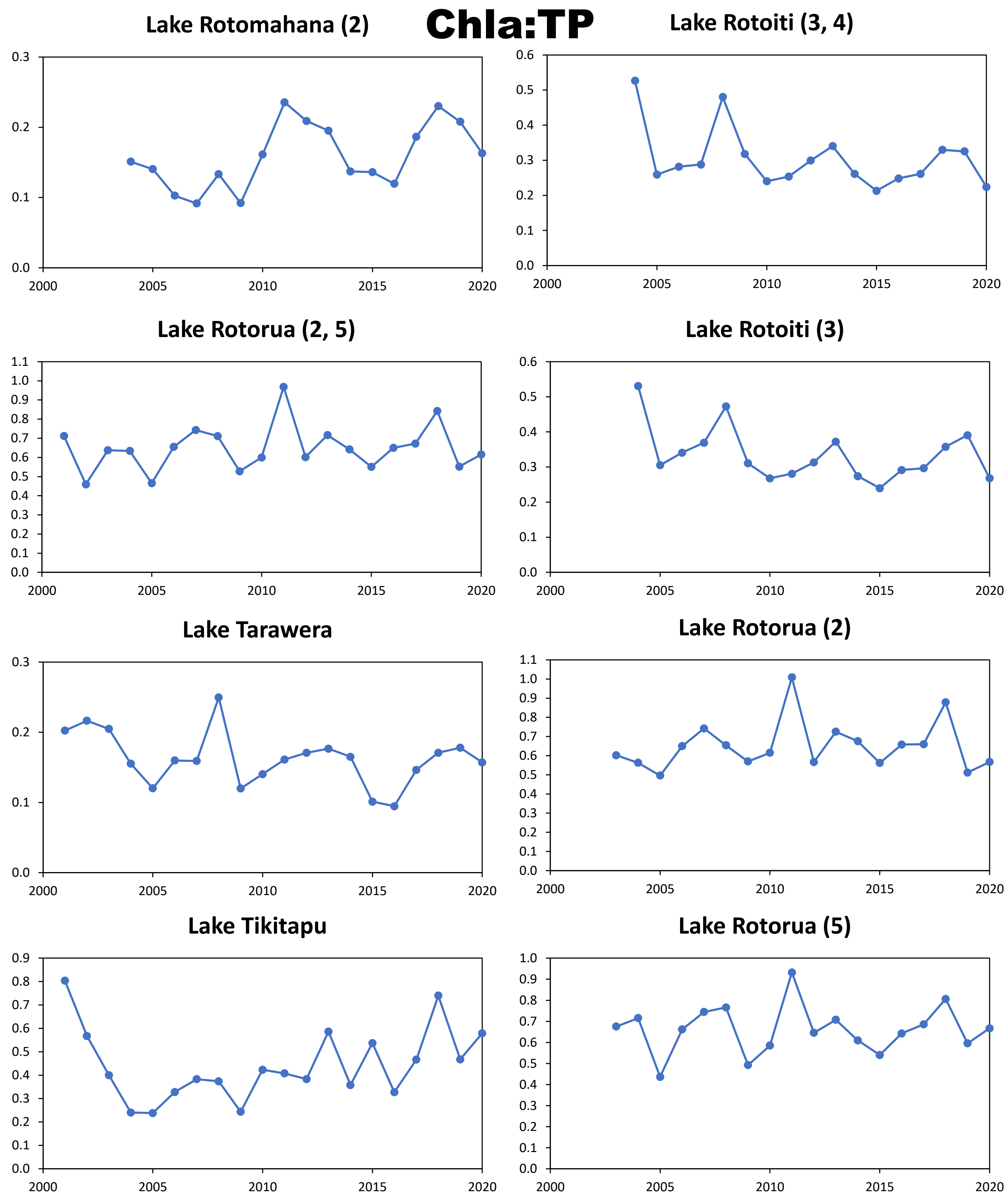

Lake Rerewhakaaitu
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
2000
2005
2010
2015
2020


Lake Rotoma
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
2000
2005
2010
2015
2020


Chla:TP
Lake Rotomahana (2)
Lake Rotoiti (3, 4)
Lake Rotorua (2, 5)
Lake Rotoiti (3)
Lake Tarawera
Lake Rotorua (2)
Lake Tikitapu
Lake Rotorua (5)
2000
2005
2010
2015
2020