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# Effect of intensive catchment and in-lake restoration procedures on phosphorus concentrations in a eutrophic lake

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ABSTRACT

Lake Okaro is a small, warm monomictic lake in central North Island, New Zealand, which progressed from oligotrophic to eutrophic through the 1960s. Trends in phosphorus (P) concentrations in the lake are linked to multiple restoration efforts over a 5-year period (2003–2008). The restoration procedures include a 2.3 ha constructed wetland established in February 2006 and riparian margin protection to reduce external loading, as well as an Alum application in December 2003 and sediment capping using modified zeolite in September 2008 to reduce internal loading. The annual average total phosphorus (TP) concentration in the lake decreased by 41% from 2004–2005 to 2007–2008. Two predictive models based on external P loading data generally underestimated the measured TP concentrations in the water column due to internal P loading. The relatively rapid response of TP concentrations after reduction of the internal loading using modified zeolite suggests that this technique can effect a rapid decrease in lake water TP concentrations though the trophic state of Lake Okaro showed high resilience to the reduced P loading. It is concluded that the combined effect of all restoration procedures resulted in a relatively rapid decrease in TP concentrations in Lake Okaro, which may be prolonged by continued external load reduction.


## 1. Introduction

Attempts to manage lake eutrophication have most frequently involved controls of nitrogen (N) and/or phosphorus (P) loads from both diffuse sources (Jeppesen et al., 1999) and point sources (Ahlgren, 1978), as well as internal loads from lakebed sediments (Cooke et al., 2005). Many of these restoration attempts have fallen short of expectations for improvement in water quality as the persistence or re-establishment of internal loading from lakebed sediments has significantly reduced the effectiveness of restoration efforts and delayed improvements in water quality (Jeppesen et al., 2005). This ecological resilience in lakes is defined by the amount of disturbance that the system can absorb, which tends to reinforce a eutrophic state, without a change in its structure or composition (Carpenter, 2003). It may therefore be possible to further hasten recovery with the control of internal nutrient loading, but the longevity of treatment effects is a major issue, particularly if external loads are not reduced concurrently.

Flocculation or P precipitation with iron-, calcium- or aluminium-salts has in several cases been effective in reducing internal P loading and water column total phosphorus (TP) concentrations (Cooke et al., 2005). Aluminium sulfate (Alum) is often a preferred choice of flocculent because the resulting floc is chemically relatively stable, even under low redox states commonly encountered under anoxic conditions. Sediment capping with calcite (Berg et al., 2004), modified clay minerals (Robb et al., 2003) and iron slag (Yamada et al., 1987) have all been used to attempt to permanently bind P in the sediments or isolate it from the water column, with varying levels of success. The longevity of sediment capping with calcite, for example, was found to be only of the order of 2–10 months for different calcite materials (Berg et al., 2004) and of the order of 6 months for Phoslock™ (Robb et al., 2003).

The management of external nutrient loads has become increasingly important in New Zealand, where only 10% of the pre-European colonisation wetlands remain (Cromarty and Scott, 1996). In the Bay of Plenty Region, where our study lake is located, only 3% of the original wetland area is intact (Park, 2002). Constructed wetlands and riparian buffer zones have been used successfully in New Zealand, some for many years (Collier, 1994; Howard-Williams and Pickmere, 1999) to partially compensate for the loss of natural wetlands. There is growing interest in their application in reducing catchment nutrient loads to surface waters (Tanner et al., 2004) which are also affected by increasing agricultural land use intensity (Hamilton, 2005). These engineered systems are designed to take advantage of natural nutrient removal processes such as sedimentation, denitrification and plant uptake

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**Table 1**
Equations for calculating trophic level index (TLI) (Burns et al., 1999) and trophic state index (TSI) (Carlson, 1977) using annual mean surface water values of the parameters chlorophyll *a* (Chl *a*) concentration (mg m$^{-3}$), Secchi depth (SD) (m), total phosphorus (TP) concentration (mg m$^{-3}$), and total nitrogen (TN) concentration (mg m$^{-3}$).

| Parameter | TLI equation | TSI equation |
|---|---|---|
| Chl *a* | $TLI_{(Chl\,a)} = 2.22 + 2.54 \log(Chl\,a)$ | $TSI_{(Chl\,a)} = 9.81 \ln(Chl\,a) + 30.6$ |
| SD | $TLI_{(SD)} = 5.10 + 2.27 \log(1/SD - 1/40)$ | $TSI_{(SD)} = 60 - 14.41 \ln(SD)$ |
| TP | $TLI_{(TP)} = 0.218 + 2.92 \log(TP)$ | $TSI_{(TP)} = 10(6 - (\ln(48/TP)/\ln 2))$ |
| TN | $TLI_{(TN)} = -3.61 + 3.01 \log(TN)$ | |
| Index | $\sum(TLI_{(TN)} + TLI_{(TP)} + TLI_{(SD)} + TLI_{(Chl\,a)})/4$ | $\sum(TSI_{(TP)} + TSI_{(SD)} + TSI_{(Chl\,a)})/3$ |

that take place in natural wetlands (Mitsch et al., 2000; Vymazal, 2007).

Water quality in several of the Rotorua Lakes, which are located in the Rotorua Region, has declined on time scales that fit with relatively recent development of land for agriculture (mostly since the 1950s) (McColl and Hughes, 1981; Hamilton, 2005) and with increasing fertilizer applications (Parliamentary Commissioner for the Environment, 2006). The Regional Water and Land Plan for the Environment Bay of Plenty region sets water quality targets for the Rotorua Lakes on the basis of the trophic level index (TLI) (Burns et al., 1999). Values of TLI are determined annually from annual mean surface water concentrations of chlorophyll *a* (chl *a*), total nitrogen (TN) and TP, and Secchi depth (SD) (Table 1). This index is used as part of the Regional Land Water Plan to evaluate long-term trends in lakes' water quality, as management actions are triggered with a pre-defined statistically significant trend in annual TLI value with time.

Here we document the changes in water quality as a result of a unique combination of restoration measures in Lake Okaro, New Zealand. We hypothesized that a progressive catchment and in-lake restoration programme should produce a rapid improvement in water quality. We assessed trends in P dynamics in Lake Okaro over 6 years, including periods immediately before and during the active restoration period. We compared observations in Lake Okaro during the restoration period with two predictive models, which derive the annual average P concentration in the water column from the external P loading. These models were necessary to evaluate the individual effects of restoration measures, because detailed pre-restoration data were limited. Furthermore, we investigated if short-term changes in water quality could be tracked within the study period using two closely related trophic level indicators.

## 2. Methods

### 2.1. Study site

Lake Okaro (Fig. 1) is a small (area 0.32 km$^2$), shallow lake (max. depth 18 m) that is seasonally stratified for c. 8 months each year and mixes fully for c. 4 months. It is the smallest of 12 lakes in the Rotorua region that are collectively referred to as the Rotorua Lakes. Lake Okaro is located in the Waiotapu geothermal area, and was filled after a geothermal explosion crater about 800 years ago (Lloyd, 1959). It has surface water inputs from two small unnamed streams that enter in the north-west of the lake and the only surface outflow is via Haumi Stream in the south-east of the lake (Forsyth et al., 1988). The lake catchment area (3.89 km$^2$) had been almost entirely cleared of native vegetation by the early 1950s and more than 95% is now in pasture, primarily for dairy production.

Limnological records for Lake Okaro extend back to 1955 (Jolly, 1977) when no cyanobacteria were observed in the lake. By contrast, persistent cyanobacterial blooms were observed in 1963 (Forsyth et al., 1988) and the lake was the study site for an international scientific investigation on cyanobacterial blooms in 1987 (Vincent, 1987). At the outset of the period under consideration in this study (2002–2008) Lake Okaro was the most eutrophic of the Rotorua Lakes and had persistent cyanobacterial blooms (Environment Bay of Plenty, 2006).

### 2.2. Overview of restoration procedure

Environment Bay of Plenty, the regional environmental manager, developed an Action Plan for Lake Okaro as a part of its Regional Land and Water Plan. Lake Okaro is the second lake of the Rotorua Lakes to have an Action Plan established. In the Lake Okaro Action Plan, the TLI goal for the lake is a reduction from 5.5 (3 year average to June 2004) to 5.0 (Environment Bay of Plenty, 2006), which still categorizes the lake as eutrophic. An even reduction was applied across the four TLI parameters (TP, TN, chl *a*, SD) in order to calculate the required TN and TP reduction. The following actions were formulated to assist with meeting both N and P reduction targets: (i) a phosphorus-adsorbent lakebed cap, (ii) a constructed wetland to remove nutrients from stream flows, (iii) protection of all riparian margins in the permanently flowing streams of the catchment, and (iv) the introduction of agricultural nutrient management practices to reduce nitrogen leaching from this source.

### 2.3. Alum application

The first attempt at in-lake removal of phosphorus was the application of 13 m$^3$ of Alum solution on 16 and 17 December 2003. Alum was applied to the surface of the lake by spraying from a moving boat as aluminium sulfate solution (47% $Al_2(SO_4)_3 \cdot 14H_2O$) to achieve a concentration of 0.6 g Al m$^{-3}$ in the epilimnion (0–3 m). A relatively low Al concentration was chosen by Environment Bay of Plenty in order to avoid the need for addition of buffering chemicals to the lake, as Lake Okaro has relatively low alkalinity (McColl, 1972). Intensive monitoring was carried out from 2 December 2003 to 13 January 2004 in order to document the short-term effects of the Alum application (Paul et al., 2008).

### 2.4. Constructed wetland

In February 2006, the two of the permanent stream inflows were diverted into a 2.3 ha surface–flow constructed wetland. The annual percentage removal of TN and TP was estimated *a priori* to be 45% (165–210 kg N yr$^{-1}$) and 10–15% (5–6 kg P yr$^{-1}$), respectively, of the lake load from this source (Tanner et al., 2007). More than 60,000 plants, including tall spike-rush (*Eleocharis sphacelata*), lake clubrush (*Bolboschoenus fluviatilis*), and jointed twigrush (*Baumea articulata*), were planted throughout the wetland.

### 2.5. Riparian protection

Riparian protection works were undertaken progressively through the study period, including livestock exclusion, fencing and planting of native plant species along the stream banks and lake margins. The importance of the riparian protection of

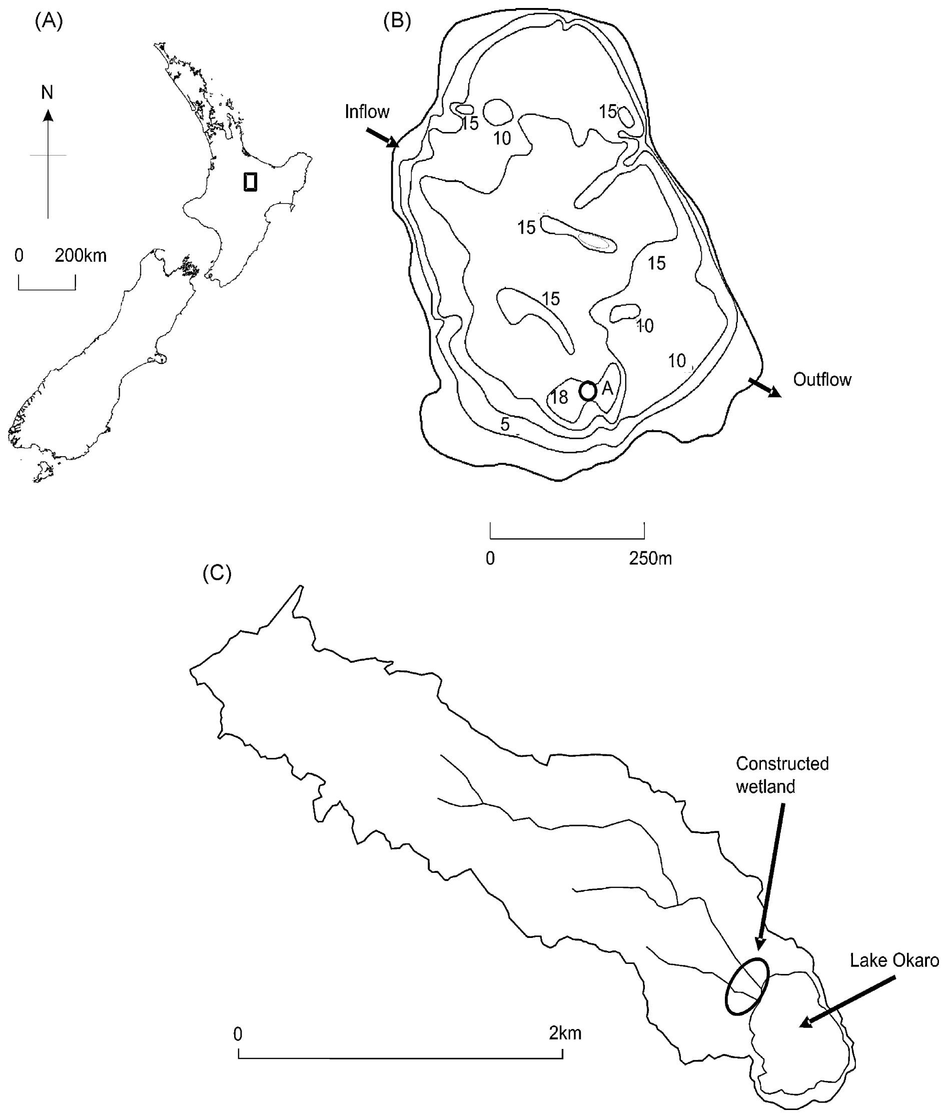


**Fig. 1.** Map of New Zealand showing location of Lake Okaro (A), Lake Okaro and its depth contours of 5, 10, 15, and 18 m, inflows and outflow and the location of the lake sampling station (B), and the lake catchment and location of the constructed wetland (C).

waterways is reflected in the Regional Water and Land Plan which defines a goal of complete riparian protection of all Rotorua lakes' streams and margins by 2012 (Environment Bay of Plenty, 2006).

### *2.6. Modified zeolite application*

Between 25 and 28 September 2007, 110 metric tonnes of a proprietary P-inactivation agent, aluminium-modified zeolite (distributed by Blue Pacific Minerals, Matamata, New Zealand), equivalent to a dose rate of 350 g m$^{-2}$ (grain size 1–3 mm), was applied as a sediment capping agent to Lake Okaro. Zeolite is a porous alumino-silicate material that has a large specific surface area for nutrient adsorption due to a fine pore structure. Zeolite can be formed synthetically at great expense, but also occurs naturally in large volcanic deposits in the Rotorua region. Scion (Rotorua, New Zealand) has developed a aluminium-modified zeolite that significantly improves the nutrient uptake capacity of the natural mineral. Modified zeolite was applied over the surface area corresponding to where lake depths were greater than 5 m; equivalent to 20 ha of lakebed above which the waters become anoxic during the seasonal stratification period of around 8 months annually. The primary objective of the modified zeolite application was an immediate (within one stratification cycle) reduction in concentrations of phosphate in the hypolimnion to less than 50% of historical values (2002–2007).

### *2.7. Farm nutrient budget*

Under the Resource Management Act, implemented by the New Zealand Government in 1991, regional councils regulate non-point discharges through land use controls or discharge permits. However, to date there are no national standards and most councils advocate a voluntary approach (Drummond, 2006). Environment Bay of Plenty takes an approach more reliant on regulations and is applying environmental programmes as a contractual agreement between rural landowners and the Regional Council or the Rotorua District Council in order to assist landowners to implement various best practice options on their farms. In this regard, nutrient budgets have been constructed for farms within the Lake Okaro catchment using a farm-scale model, Overseer®. This model allows determination of nutrient inputs and outputs from a farm for a variety of input data (e.g., stock numbers, soil type, fertilizer regime, and climate). The model also provides a means to investigate mitigation options to reduce the environmental impact of nutrients for a specific land use (Currie and Hanly, 2003).

### 2.8. Sampling and monitoring

Samples for water column nutrients and chl *a* were taken monthly at 0–4 m (integrated sample) and 14 m depths from sampling station A (Fig. 1) as part of a Natural Environment Regional Monitoring Network conducted by Environment Bay of Plenty. Secchi depth was determined with a disk of 20 cm diameter. Sampling of inflows for nutrient concentrations and measurements of discharge were added to the programme in July 2004 and are included in this study to June 2008. Inflows were measured some 10 m upstream of the lake margin. Discharge was measured for only 1 year, from July 2007 to June 2008. Unfiltered water samples were frozen prior to analysis for concentrations of TP and total Kjeldahl nitrogen (TKN). Additional samples were filtered (Whatman GF/C) and frozen for subsequent analysis as nitrate of the total oxidised nitrogen (TON = $NO_2$ + $NO_3$) using APHA method 4500 $NO_3$–N (APHA, 2005) and soluble reactive phosphorus (SRP) using a standardised molybdate blue method. Analysis of TKN was carried out using APHA method 4500B (APHA, 2005). Total nitrogen was then calculated as TN = TKN + TON. Total phosphorus was analysed using an acid persulfate digestion procedure (APHA, 2005). The filters were frozen for subsequent analysis of chl *a* concentration using an acetone extraction procedure (Arar and Collins, 1997).

Conductivity-temperature-depth (CTD) profiles (SBE 19 *plus* SEACAT Profiler, Seabird Electronics Inc.), with additional CTD mounted sensors for dissolved oxygen (DO) concentration (Seabird Electronics), were taken at each sampling occasion. To calculate the water balance incorporating evaporation climate data were obtained from Rotorua Airport AWS—Station B86133 (20 km north of Lake Okaro). Daily averages were calculated from hourly observations of wind speed (m $s^{-1}$), atmospheric pressure (hP), ambient temperature (°C) and rainfall (mm) over the whole sampling period.

### 2.9. Data analysis

For the variables chl *a*, SD, TP, and TN, arithmetic means for the surface waters (from 0 to 4 m) during the period between July and June for each year were calculated. A TLI was then calculated annually for Lake Okaro according to Burns et al. (1999) for the period of July 2002 until June 2007 (Table 1). In addition, a trophic state index (TSI) was calculated annually according to Carlson (1977) for the same period, for comparison (Table 1). The TLI is a numerical value of lake trophic level and can range from 0 (ultra-microtrophic) to 7 (hypertrophic) with major divisions of 1 (e.g., 0–1, 1–2, 2–3, etc.). The TSI is a numerical value in a scale of 0–100 with major divisions of 10 (10, 20, 30, etc.). It is notable, that both values can theoretically exceed their upper values.

A critical external P loading for Lake Okaro, using a target trophic state value for TP (68 mg $m^{-3}$) (Environment Bay of Plenty, 2006), was determined from the equation of Vollenweider (1976):

$$\text{Critical loading}(\text{mg m}^{-2}\,\text{yr}^{-1}) = \frac{\text{TP}_{\text{target}}(1+\sqrt{\tau})z_{\text{m}}}{\tau} \quad (1)$$

where $z_m$ is the mean depth of the lake (m). The theoretical water retention time $\tau$ (yr) was calculated as discharge ($Q_{out}$) divided by lake volume (V). For the periods of July 2004 until June 2007, $Q_{out}$ was calculated as the residual term in a complete lake water balance from July 2007 until June 2008, using the methodology of Wetzel and Likens (2000). The critical loading was calculated for the period from 2004 to 2008.

An approach to predict annual average TP within a lake is given by OECD (1982) and is hereafter referred to as the Vollenweider model:

$$\text{TP} = 1.55\left[\frac{\text{TP}_{\text{inflow}}}{1+\sqrt{\tau}}\right]^{0.82} \quad (2)$$

where TP is the annual average in-lake TP concentration, and $TP_{inflow}$ is the discharge weighted annual mean inflow TP concentration.

An alternate model to predict TP in cases where it is still responding to a change in loading, i.e., non-equilibrium, is given by Sas (1989) and is hereafter referred to as the Sas model:

$$\text{TP} = \text{TP}_{\text{lake pre}}\left(\frac{\text{TP}_{\text{inflow post}}}{\text{TP}_{\text{inflow pre}}}\right)^{0.65} \quad (3)$$

where the subscripts pre and post refer to pre-reduction and post-reduction periods. Both models were applied to 1-year periods (July–June) from 2004 to 2008. Predictive models that account for internal P loading as well as external P loading were not considered in this study.

The Pearson Chi-Square ($X^2$) test was used to compare annual average water column TP with values derived from the Vollenweider and Sas models. A non-parametric Wilcoxon matched pairs test was used to compare arithmetic mean bottom and surface water concentrations of TP and SRP for 1 year before the restoration programme was started (2002–2003) and for 5 years during the active restoration period (2003–2008). Annual means of surface concentrations for chl *a*, TP, and TN and SD were tested for significant correlations using Spearman rank correlation coefficient ($r_S$) values.

## 3. Results

### 3.1. Temperature, stratification and dissolved oxygen distribution

Lake Okaro was thermally stratified for an average of 8 months each year based on monthly sampling (Fig. 2) with the criteria for stratification defined as the presence of a thermocline where d$T$/d$z$ = minimum, where $T$ is the water temperature (°C) and $z$ is water depth (m). The lake was considered to be fully mixed when dT/dz was >−0.25 °C $m^{-1}$ throughout the water column. Stratification usually commenced in September, was strongest at the end of January and broke down in June. The period of stratification was 1 month longer in our study than that recorded in the 1970s (McColl, 1972). Temperature in surface waters ranged from 23.5 to 9.1 °C over the entire study period. During stratification, temperature near the bottom of the lake remained close to the winter value of c. 8.5 °C but increased slightly (c. 2–3 °C) towards the end of summer (Fig. 2).

Surface waters were regularly supersaturated in DO during stratification, with a maximum concentration of 17.3 mg $L^{-1}$ during the study period (Fig. 2). However, the water near the bottom of the lake became anoxic shortly after onset of stratification, though monthly sampling prevents more detailed resolution of the timing. The duration of bottom water anoxia was 28 weeks on average. For comparison, in 1962 bottom waters did not become anoxic (Jolly, 1977), while between 1965 and 1986 the hypolimnion was anoxic for an average of 21 weeks during each seasonal stratification period (Forsyth et al., 1988).

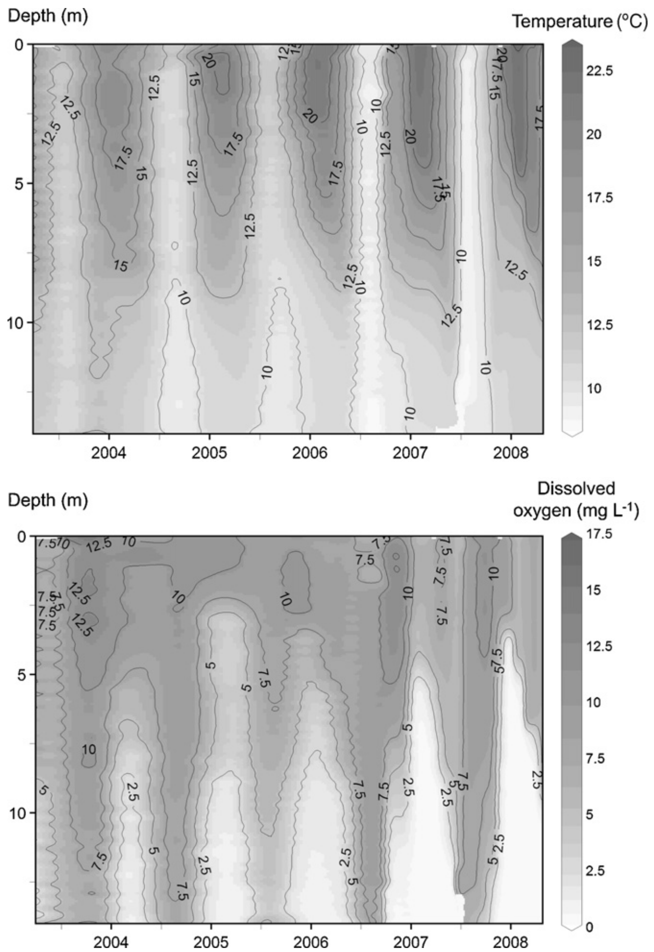


**Fig. 2.** Temperature (°C) (A), and dissolved oxygen concentrations (mg $L^{-1}$) (B) in Lake Okaro for March 2003 until June 2008.

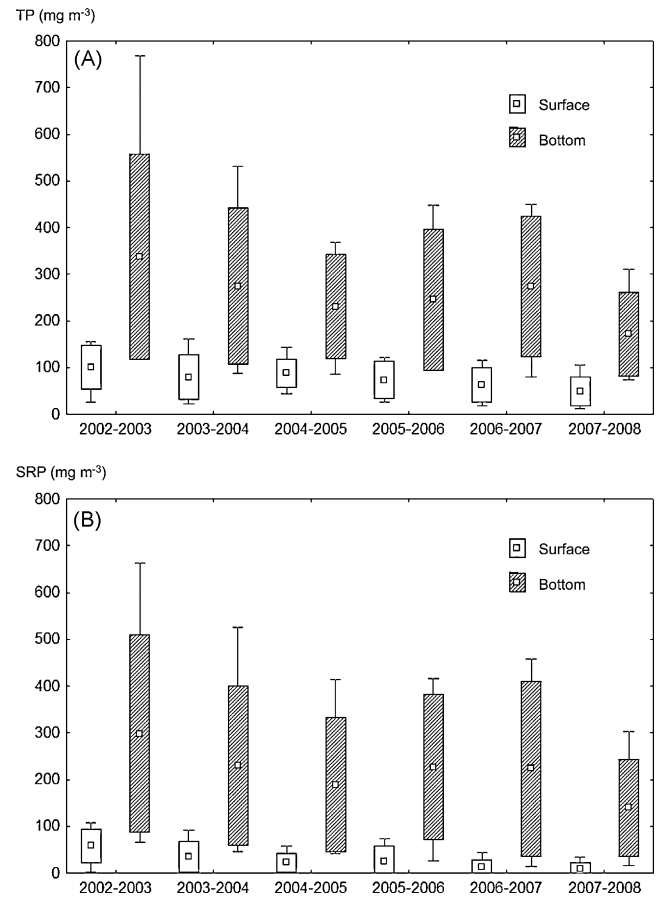


**Fig. 4.** Annual mean (July–June) surface and 14 m concentrations of TP (A) and SRP (B). Box represents ±one standard deviation, and bars represent the range.

## 3.2. Total phosphorus trend in the bottom and surface waters

Fig. 3 shows monthly TP and DO concentrations in Lake Okaro at 14 m depth from July 2002 until June 2008. Total phosphorus concentrations usually increased shortly after onset of anoxic conditions in the bottom waters, indicating P release from the sediments. Concentrations of TP at 14 m were up to 769 mg $m^{-3}$ during the period of July 2002–June 2003 while in the following year, following the December 2003 Alum dosing, the maximum declined to 531 mg $m^{-3}$. Concentrations were further reduced for the periods July 2004–June 2007 with a maximum TP of 448 mg $m^{-3}$ (March 2007), although there was no obvious decline in TP immediately after the construction of the wetland (February 2006). Following the modified zeolite application in September 2007, TP was lower than recorded in previous years, with a maximum of 310 mg $m^{-3}$ in April 2008.

The mean concentration of TP in surface waters (0–4 m) in 2002–2003 was 106.1 mg $m^{-3}$ (Fig. 4) and decreased follow-

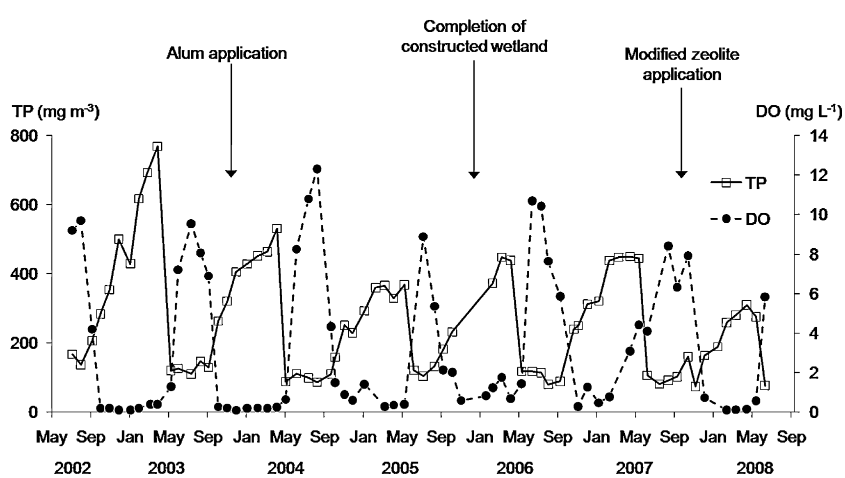


**Fig. 3.** TP concentration (mg $L^{-1}$) and dissolved oxygen concentration (mg $L^{-1}$) at 14 m depth in Lake Okaro for the period of July 2002 until June 2008.

**Table 2**
Results of Wilcoxon matched pair tests for annual mean concentrations of total phosphorus (TP) and soluble reactive phosphorus (SRP) at the surface and 14 m depth for periods of July–June between 2002 and 2008.

| | TP (surface) | TP (bottom) | SRP (surface) | SRP (bottom) |
|---|---|---|---|---|
| *p*-Levels for Wilcoxon matched pairs test | | | | |
| 2002–2003 vs 2003–2004 | 0.29 | 0.01* | 0.01* | 0.01* |
| 2003–2004 vs 2004–2005 | 0.86 | 0.06 | 0.03* | 0.05 |
| 2004–2005 vs 2005–2006 | 0.67 | 0.21 | 0.46 | 0.75 |
| 2005–2006 vs 2006–2007 | 0.29 | 0.40 | 0.06 | 0.09 |
| 2006–2007 vs 2007–2008 | 0.01* | 0.01* | 0.20 | 0.02* |

* Denotes significant difference at $p < 0.05$.

**Table 3**
Annual mean values (July–June) for Secchi depth (SD) (m), surface concentrations of chlorophyll *a* (Chl *a*) (mg m$^{-3}$), total phosphorus (TP) (mg m$^{-3}$), total nitrogen (TN) (mg m$^{-3}$), Trophic Level Index (TLI), Trophic State Index (TSI) and the TN:TP. $TLI_{Chla}$, $TLI_{SD}$, $TLI_{TP}$ and $TLI_{TN}$, $TSI_{Chla}$, $TSI_{SD}$ and $TSI_{TP}$ represent the individual values for the trophic level indices for the each variable.

| | Jul 2002–Jun 2003 | Jul 2003–Jun 2004 | Jul 2004–Jun 2005 | Jul 2005–Jun 2006 | Jul 2006–Jun 2007 | Jul 2007–Jun 2008 |
|---|---|---|---|---|---|---|
| Chl *a* | | | | | | |
| mg m$^{-3}$ | 26.54 | 19.73 | 77.18 | 17.05 | 19.97 | 27.28 |
| $TLI_{(Chl\ a)}$ | 5.84 | 5.51 | 7.01 | 5.35 | 5.52 | 5.87 |
| $TSI_{(Chl\ a)}$ | 62.76 | 59.85 | 73.24 | 58.42 | 59.97 | 63.03 |
| SD | | | | | | |
| m | 1.86 | 2.38 | 1.48 | 2.74 | 2.42 | 2.37 |
| $TLI_{(SD)}$ | 4.44 | 4.18 | 4.68 | 4.04 | 4.17 | 4.19 |
| $TSI_{(SD)}$ | 51.08 | 47.50 | 54.36 | 45.50 | 47.25 | 47.59 |
| TP | | | | | | |
| mg m$^{-3}$ | 106.09 | 73.10 | 87.82 | 74.78 | 62.25 | 46.50 |
| $TLI_{(TP)}$ | 6.13 | 5.66 | 5.89 | 5.69 | 5.46 | 5.09 |
| $TSI_{(TP)}$ | 64.27 | 60.54 | 62.38 | 60.77 | 58.93 | 56.02 |
| TN | | | | | | |
| mg m$^{-3}$ | 867.22 | 849.82 | 1236.45 | 830.93 | 894.42 | 803.00 |
| $TLI_{(TN)}$ | 5.23 | 5.21 | 5.70 | 5.18 | 5.27 | 5.13 |
| TLI | 5.41 | 5.14 | 5.82 | 5.06 | 5.11 | 5.07 |
| TSI | 59.37 | 55.96 | 63.32 | 54.89 | 55.39 | 55.55 |
| TN:TP ratio | 8.17 | 11.63 | 14.08 | 11.11 | 14.37 | 17.27 |

ing the Alum application of December 2003 to 73.1 mg m$^{-3}$ (2003–2004); however, this reduction was not statistically significant (Wilcoxon matched pairs test, $p > 0.05$; Table 2). In contrast, a reduction in the annual mean TP in bottom waters, from 337 mg m$^{-3}$ (2002–2003) to 274.7 mg m$^{-3}$ (2003–2004), was statistically significant ($p < 0.05$). Annual mean surface water TP concentration decreased significantly ($p < 0.05$) from 274.4 mg m$^{-3}$ (2006–2007) to 172 mg m$^{-3}$ (2007–2008) at 14 m depth and from 62.3 mg m$^{-3}$ (2006–2007) to 46.5 mg m$^{-3}$ (2007–2008) in surface waters. Annual mean concentrations of SRP in the bottom waters declined significantly ($p < 0.05$) from 298.4 mg m$^{-3}$ (2002–2003) to 230.8 mg m$^{-3}$ (2003–2004), while concentrations of SRP at 14 m were also significantly lower ($p < 0.05$) for the period 2007–2008 (139.6 mg m$^{-3}$) compared to 2006–2007 (223.4 mg m$^{-3}$).

### 3.3. External and internal phosphorus loads

Fig. 5 shows the external P load, $TP_{ext}$, which was 571 mg m$^{-2}$ yr$^{-1}$ for 2004–2005 and 575.2 mg m$^{-2}$ yr$^{-1}$ for 2005–2006. Following implementation of the wetland in February 2006, $TP_{ext}$ decreased to 367.2 mg m$^{-2}$ yr$^{-1}$ (2006–2007) and declined further to 306 mg m$^{-2}$ yr$^{-1}$ in 2007–2008. The annual percentage removal of TP in the surface inflow was 42% for the 2 years following the construction of the wetland compared with the 2 years preceding its construction.

The calculated average critical TP loading (Vollenweider, 1976) for 2004–2005 and 2005–2006 was 35% higher on average than $TP_{ext}$ and 100% higher for the years after the wetland had been implemented (2006–2008). The critical TP loading ranged from 626.7 to 822.4 mg m$^{-2}$ yr$^{-1}$ across all years.

Annual mean water column (average of 0–4 and 14 m depth) concentrations of TP (Fig. 5) declined from 241.8 to 192.7 mg m$^{-3}$ between the years 2002–2003 and 2005–2006, respectively, increased to 215.5 mg m$^{-3}$ in 2006–2007 and then declined again to 141.1 mg m$^{-3}$ for 2007–2008. The Vollenweider model produced TP concentrations ranging from 111.1 mg m$^{-3}$ (2005–2006) to 59.6 mg m$^{-3}$ (2007–2008), with values consistently lower than both the Sas model and the measured data for this period. The Vollenweider model underestimated TP concentrations in the lake (Pearson Chi-Square test, $X^2 = 533.9$) by 45% for 2004–2006 and by 65% lower for 2006–2007. For the period of 2007–2008, which

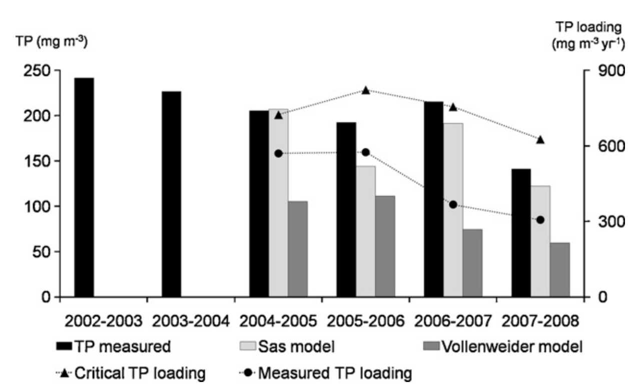


**Fig. 5.** Annual mean water column (July–June) measured and modelled total phosphorus (TP) concentrations (mg m$^{-3}$) for Lake Okaro and measured and critical areal TP loading (mg m$^{-2}$ yr$^{-1}$). Critical loading is the value required to achieve a target TP concentration of 68 mg m$^{-3}$ (Environment Bay of Plenty, 2006).

included the modified zeolite application, Vollenweider-predicted TP was 57% lower than the measured value. The Sas model predictions matched the measured TP concentrations in the water column better throughout the entire study period (Pearson Chi-Square test, $X^2 = 22.5$) and produced a range from 206.9 mg m$^{-3}$ (2004–2005) to 122.1 mg m$^{-3}$ (2007–2008). This model matched the measured TP concentration most closely for 2004–2005 and was 16% on average lower than measured TP for the remainder of the study period (2005–2008).

### 3.4. *Trophic level index and trophic state index*

Annual values of the TLI ranged from 5.06 (July 2005–June 2006) to 5.82 (July 2004–June 2005; Table 3), indicating that Lake Okaro is hypertrophic. For the same period TSI ranged from 54.9 (July 2005–June 2006) to 63.3 (July 2004–June 2005; Table 3), representing a highly eutrophic state. TSI was highly significantly correlated ($R = 0.99$, $p < 0.01$) with TLI for the entire study period.

Annual mean chl *a* concentration ranged from 17.1 mg m$^{-3}$ (2006–2006) to 77.2 mg m$^{-3}$ (2004–2005) and showed no apparent decline towards the end of the study period, i.e., after implementation of all restoration procedures. Annual mean SD increased overall from 1.86 m (2002–2003) to 2.37 m (2007–2008) with the lowest annual mean SD of 1.48 m for the period 2004–2005. Annual mean surface water TP concentrations declined throughout the study period from a maximum of 106.1 mg m$^{-3}$ (2002–2003) to a minimum of 46.5 mg m$^{-3}$ (2007–2008), while TN was relatively consistent at 803 mg m$^{-3}$, with a peak of 1236.5 mg m$^{-3}$ in 2004–2005. Ratios of TN:TP increased progressively over the duration of the study, from a minimum of 8.2 (2002–2003) to a maximum of 17.3 (2007–2008). Annual mean concentrations of chl *a* were not correlated with annual mean concentrations of TP (Spearman rank correlation coefficient $r_S = 0.14$, $p > 0.05$), but chl *a* was significantly correlated with 1/SD ($r_S = 0.89$, $p < 0.05$) (Table 4).

## 4. Discussion

### 4.1. *External P loading vs internal P loading*

Both external and internal P loading were reduced during the study period; however, Lake Okaro remains eutrophic. The external TP loading reduction can be attributed to the riparian protection measures, the farm nutrient budgeting, and the constructed wetland, the combination of which was estimated to reduced TP loading by c. 53 kg yr$^{-1}$ (Environment Bay of Plenty, 2006). The TP loading from the surface inflow alone produced a reduction of c. 40 kg yr$^{-1}$.

Although the Sas loading model yielded better agreement with measured TP than the Vollenweider model, both models consistently underestimated lake water TP concentrations as a result of either ignoring or underestimating, respectively, internal P loading. If it is assumed that the difference between measured TP concentrations and those calculated from the Vollenweider model emanates from internal loading, then this source contributes c. 50% of the water column TP in Lake Okaro. The critical P loading model, which was used to quantify the external P loading to meet the target trophic state value for TP for Lake Okaro, also indicates clearly that internal P loading is important, since the measured P load was consistently lower than the model prediction, but TP concentrations in the lake were still higher than either of the models and the target water column TP concentration of 68 mg m$^{-3}$ (Environment Bay of Plenty, 2006) which was used as the basis to define the TLI target of 5.0.

**Table 4**
Spearman rank correlation coefficients ($r_S$) for annual mean concentrations of surface chlorophyll *a* (chl *a*), total nitrogen (TN), total phosphorus TP, and Secchi depth (SD).

| | 1/SD | TP | TN |
|---|---|---|---|
| Spearman rank correlation coefficients ($r_S$) | | | |
| Chl *a* | $r_S = 0.89^*$ | $r_S = 0.14$ | $r_S = 0.37$ |
| 1/SD | | $r_S = 0.54$ | $r_S = 0.43$ |
| TP | | | $r_S = 0.54$ |

* Denotes significant difference at $p < 0.05$.

Internal loading decreased following the Alum application (December 2003) when P concentrations were lower in both surface and bottom waters. Although annual mean phosphorus concentrations in bottom waters remained lower for the remainder of the study period when compared with 2002–2003, the effectiveness of the Alum application appeared to be only temporary and TP concentrations increased slightly in 2006–2007.

The mean hypolimnetic SRP concentration in 2007–2008 was 38% lower than in previous years (2002–2007). The reduction in SRP concentration did not reach the restoration target of 50%. Internal loading was clearly reduced for the last period (2007–2008) when the Vollenweider model began to more closely approximate the measured TP, most likely as a response to the modified zeolite application.

### 4.2. *Longevity of restoration measures*

The decrease in external P loading indicates efficient P retention in the constructed wetland for the 2 years of measurements after its construction. The P cycle in constructed wetlands is sedimentary rather than gaseous (as with the N cycle) and predominantly involves formation of complexes with organic matter or inorganic sediments. As a result, P retention in constructed wetlands can be high in the period immediately following construction, but decreases as the soil substrate of the wetland becomes saturated with P and potentially acts as a P source (Mitsch et al., 2000; Fink and Mitsch, 2004). This is not always the case, as White et al. (2000) found that wetland sediments were not saturated with P despite several years of high P inputs in a large (1250 ha) restored prairie marsh. Successful reduction of P loading to Lake Okaro through the farm nutrient management system and riparian protection could significantly extend P retention in the constructed wetland.

Lake Okaro appeared to respond relatively quickly to reduced external nutrient loading compared with other stratified lakes with anoxic hypolimnia, where recovery after P input reduction was observed on time scales of around 10–15 years (Jeppesen et al., 2005). Rapid recovery by external load control has generally been observed in lakes with permanently oxic hypolimnetic waters and a short history of nutrient enrichment (Cooke et al., 2005). By contrast, with Lake Okaro having a persistent anoxic hypolimnion, the reduction of both, internal and external P loading appeared to have caused a response in Lake Okaro similar to lakes with oxic hypolimnia.

Phosphorus inactivation by aluminium has been shown to be very effective in the long-term in polymictic lakes (Reitzel et al., 2005) and efficacy has remained high for periods up to 18 years in stratified lakes (Welch and Cooke, 1999). The Alum application in Lake Okaro had limited persistent effects on internal loading, however, most likely due to the low Alum concentrations (0.6 g m$^{-3}$; Paul et al., 2008), which was at the lower range (0.05–30 g m$^{-3}$) used in most case studies (Welch and Cooke, 1999). A further factor influencing the effectiveness of Alum in Lake Okaro may be associated with pH in surface waters exceeding 9, as observed during the time of the Alum application (Paul et al., 2008). While an Alum floc is stable and re-dissolution of phosphate under anoxic conditions is unlikely, an unfavourable water column pH (i.e., outside the recommended range of 6–8) can significantly reduce the P sorption capacity of Alum (Cooke et al., 2005).

The sediment capping layer created with modified zeolite was intended to form a diffusion barrier by binding free orthophosphate released from the bottom sediments; however, visual inspection revealed an incomplete coverage of the bottom sediments. Insufficient dose combined a coarse grain size (1–3 mm) may have contributed to the incomplete coverage and reduced the efficacy of the sediment capping. Berg et al. (2004) showed that an increase in layer thickness of calcite barriers, from 1 to 2–4.5 cm, reduced P release for more than 6 months with different grain sizes tested ranging from 2 to 1200 μm. Other studies confirm that smaller grain size of the sediment capping material, resulting in increased coverage, yields greater suppression of sediment P release (Yamada et al., 1987). However, a diversity of pathways of P transport, such biologically mediated transport, and gas ebullition can also be equally important in affecting the effectiveness of sediment capping (Förstner and Apitz, 2007). As aluminium is the active component leading to P adsorption in Z2G1, water column pH, particularly in the bottom waters close the sediment–water interface during the application may also affect its P uptake capacity. For both the Alum and modified zeolite applications, consideration of dose rate, coverage of the capping material and application timing with regard to water column pH appear to be critical factors in the long-term reduction of internal P loading.

### *4.3. Criteria for improved water quality using trophic level indicators*

Both water quality indices (TLI and TSI) indicate that Lake Okaro is highly eutrophic (Burns et al., 1999; Carlson, 1977), but the TSI shows that Lake Okaro has moved across different trophic divisions (56.0 in 2003–2004, 63.3 in 2004–2005) over the study period, indicating that TSI may be a more sensitive indicator. For example, for the period of July 2004–June 2005, Lake Okaro remained supertrophic according to the TLI whereas the TSI changed from 50 to 60 in divisions. Interestingly, however, the TSI failed to describe an improvement in water quality as a result of TP reductions for 2007–2008 because chl *a* for this period showed a slight increase compared with the previous year.

Both indexes indicate, however, that the trophic state has not responded in a way that might be commensurate with changes in P concentration. Rapid and efficient recycling of P within the lake can maintain an initially high trophic status despite reduction of inputs (Sas, 1989). Although internal loading was reduced in Lake Okaro, the lake trophic state appeared to be resilient to the reduced P loading. Apart from external and internal nutrient loading a number of feedback mechanisms, including the persistence of an anoxic hypolimnion, have been proposed. Eutrophic lakes may therefore be highly resilient to restoration efforts, and multiple factors beyond TP reduction may be required to shift a lake back to a ‘pristine’ state (Suding et al., 2004). It is inconclusive if there is a threshold for TP loading reduction required to effect a rapid improvement in lake trophic status analogous to the well-known hysteresis of alternative stable states of macrophyte dominance and phytoplankton dominance in shallow lakes (Scheffer and Carpenter, 2003). Furthermore, in Lake Okaro reduction of P in the water column was not accompanied with a reduction in N.

Although the regional management organisation, Environment Bay of Plenty, acknowledges the importance of controlling both potentially limiting nutrients, N and P, in its restoration efforts, the focus of nutrient control in Lake Okaro has been on P, partly because of the potential for N fixation to lead to dominance by cyanobacteria in the phytoplankton assemblage (Schindler et al., 2008). The TN:TP ratio increased largely due to decreasing TP concentration in this study. Lean et al. (1987) raised doubts as to the importance of TN:TP ratios in controlling cyanobacterial dominance in Lake Okaro and showed that high internal P concentrations in the phytoplankton would allow a substantial increase in biomass to relieve the shortage of N.

Neither the TLI nor the TSI appears to be suitable for tracking pre-defined water quality restoration targets for short periods. Targeting only one of the index input variables (e.g., TP) during lake restoration when all are weighted equally may not adequately account for stabilisation of the eutrophic status by longer lasting feedback mechanisms. When loadings of both N and P were reduced simultaneously, algal biomass may also be reduced without substantial delays (Köhler et al., 2005). In other lakes, however, where only P loading was reduced, water quality has improved without delay as measured by algal biomass, i.e., chl *a* (Coveney et al., 2005).

### *4.4. Relative success of restoration methods*

The observed significant decline in P concentration in the bottom waters of Lake Okaro in response to the modified zeolite application suggests that sediment capping was the most effective amongst all of the restoration procedures carried out in this lake. The constructed wetland, farm nutrient management, and the riparian restoration were collectively very effective in reducing external loading of TP; however, we could not actually isolate their individual contributions. Case studies have shown that a combination of several restoration measures is often more successful in achieving a sustained improvement in the water quality of lakes than applying a single procedure alone (Bergman et al., 1999; Ruley and Rusch, 2004; Kasprzak et al., 2007). While restoration methods and procedures must have a foundation in the basic science of limnology (Cooke et al., 2005), the reductionism approach of classical scientific studies makes lake restoration difficult to achieve, particularly as degradation may have occurred via multiple stressors; restoration at an ecosystem level is similarly likely to require implementation of multiple procedures. Further research using a watershed P balance model (Schussler et al., 2007) may enable res-

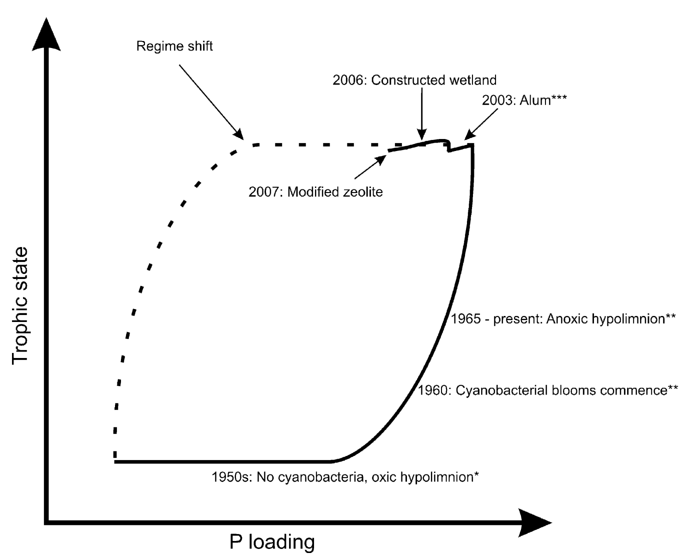


**Fig. 6.** Conceptual model of trophic state in relation to P loading (internal and external) in Lake Okaro demonstrating possible regime shifts in response to changes in P loading in Lake Okaro (dotted line) (*Jolly, 1977; **Forsyth et al., 1988; ***Paul et al., 2008).

olution of the individual contributions of restoration methods in the catchment.

The results presented here reflect the short- to medium-term restoration outcome in a single stratified lake (Fig. 6). Efficient restoration of stratified lakes with anoxic hypolimnia requires that the external load is reduced below a critical threshold, and to avoid delays in recovery, prevention of P recycling from the sediment by phosphorus inactivation or sediment capping can substantially increase the probability that the threshold value might be obtained. Even with a multi-pronged approach, it is still a significant challenge to reach the threshold for P loading that might bring about a regime shift to a distinctly different trophic state with a permanently oxic hypolimnion. Intensive long-term monitoring of restoration measures can be time consuming and costly. However, as shown with the restoration of Lake Okaro, such monitoring is critical to accurate performance analysis and is only possible by pre-defining restoration targets and monitoring appropriate variables, to draw conclusions as to the success of the restoration procedures.

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