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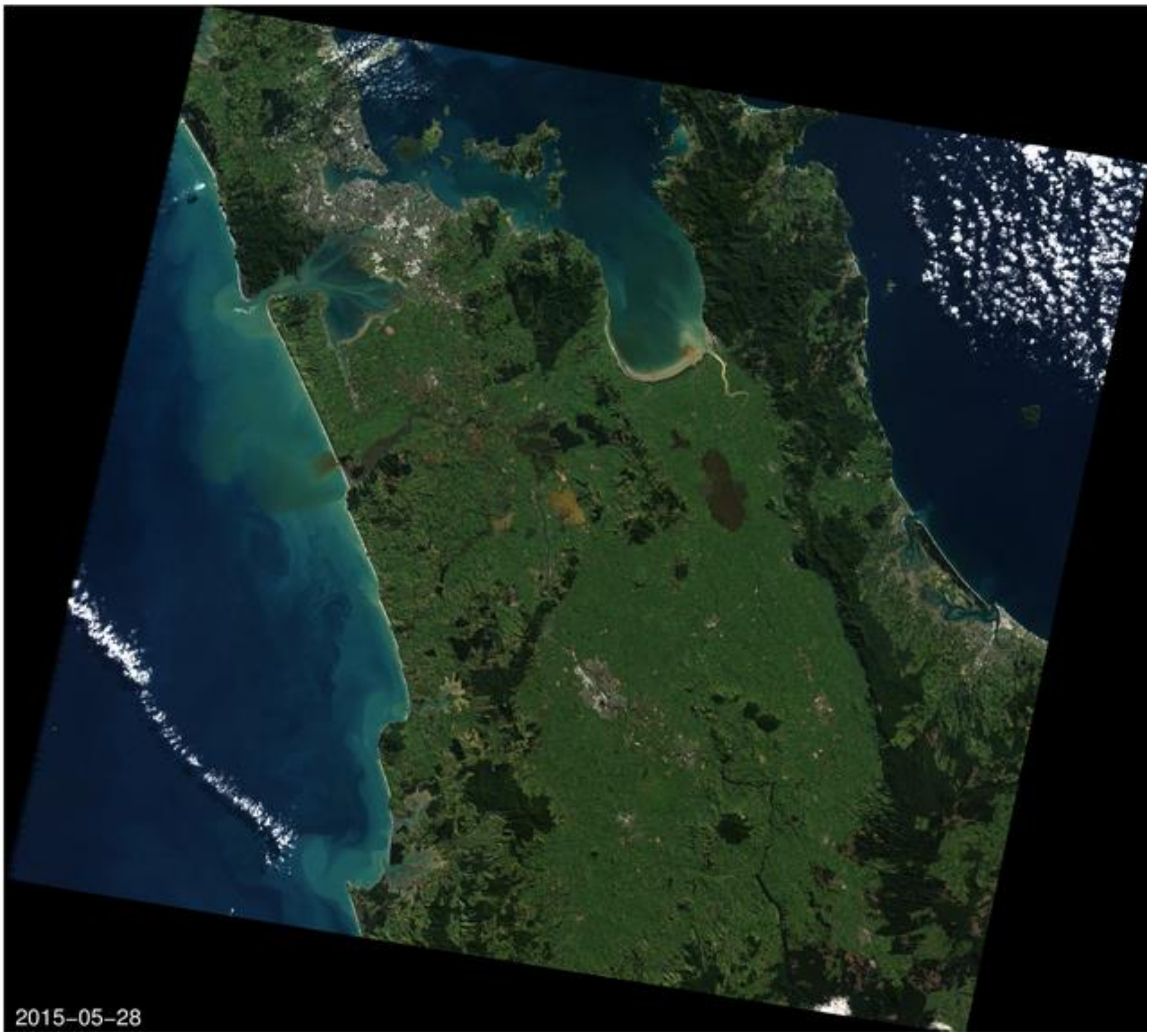
Ecological modelling of water quality management options in Lake Waahi to support Hauanga Kai species: Technical report



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Te Pūtahi Rangahau Taiao
THE UNIVERSITY OF WAIKATO

THE UNIVERSITY OF WAIKATO
Te Whare Wānanga o Waikato

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Cover photo

Landsat seven image captured 28/5/2015

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Availability

Executive Summary

This report describes a numerical modelling approach for Lake Waahi to assess the effects of various future scenarios on water quality attributes known to affect the appearance of the lake's multiple hauanga kai species across differing spatial scales. Hauanga kai refers to customary gathering and use of naturally occurring and cultivated foods. Numerical models were used to support shared learning and development of future scenarios in response to different management options and environmental pressures.

The shallow lakes of the Waikato region have been particularly impacted by anthropogenic eutrophication. This is due to the fact that many of these lakes have already undergone a degree of natural eutrophication due to their geomorphic properties, such as their low elevation and shallow depth. The main contributing factors to water quality decline in shallow Waikato lakes have been identified as: high external nutrient and sediment loads, internal loading exacerbated by high sediment resuspension and bioturbation from pest fish (carp), reduced water depth due to installation of water control structures, reduction in wetland filtering potential due to draining, and stock access to wetlands.

A 1-D hydrodynamics model, the General Ocean Turbulence Model was coupled to a complex ecological model (PClake) via the Framework for Aquatic Biogeochemical Models (FABM). FABM is an open source FORTAN-based code that facilitates the coupling of hydrodynamic and ecological models. FABM-PCLake is a redesigned PCLake.

The consensus from the model simulations carried out in the study suggests that, for Lake Waahi to undergo restoration whereby the lake is restored to clear-water stable state, both external load needs to be reduced significantly by greater than 50%, along with changes in lake ecological structure, resulting from re-establishment of macrophytes aided by the control of carp biomass. These finding supports the general view within the literature that for lake restoration success, external load reduction (catchment load) is critical.

Finally, if the ultimate goal of lake restoration in Lake Waahi is to address the decline in tuna (shortfin eel - *Anguilla australis*) biomass, a return to a clear-water stable state should facilitate improvement in tuna habitat and provide the conditions for development of a healthy eel population. Coupled modelling approaches will provide a more complete picture of potential responses of Hauanga kai species to land and lake management scenarios.

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Acknowledgements

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Introduction

In New Zealand, pastoral land use has intensified over recent decades, reflecting a shift in land use and farming systems to more intensification, notably higher stocking rates and increased fertiliser application resulting in increases in external nutrient loading. These factors have been identified as critical in the eutrophication and degradation of New Zealand's freshwater ecosystems (PCE 2012; PCE 2013), resulting in a reduction in biodiversity, ecological function and aesthetic values of waterbodies (PCE 2012). The 2014 implementation of the National Policy Statement for Freshwater Management (Ministry for the Environment 2015) included the National Objectives Framework (NOF), a numeric and descriptive framework which enables regional councils to set environmental policy regarding freshwater management. Now regional councils have a legal obligation to enforce water quality standards. Mahinga kai, known to Waikato-Tainui as Hauanga kai, is a key value for freshwater management that needs to be articulated in the National Objectives Framework (NOF) limit-setting process (Collier et al. 2014).

Eutrophication is often defined as an increase in nutrients including phosphorus and nitrogen which stimulate algal growth, potentially leading to a decrease in water quality. Eutrophication is a natural process whereby, with age, lakes gradually infill with sediment, nutrients, and organic matter. Human-induced changes within catchments can speed up this natural process, which is then termed anthropogenic or cultural eutrophication. Cultural eutrophication can result in decreased water clarity, harmful algal blooms, increased suspended sediment, hypoxia and anoxia, losses or increased growth of submerged macrophytes and lake biota, and, ultimately, risks to human health (Williamson et al. 2008). Lakes can be thought of as integrators of environmental change which has occurred within their catchments (Williamson et al. 2008).

The shallow lakes of the Waikato region have been particularly impacted by anthropogenic eutrophication. This is due to the fact that the many lakes have already undergone a degree of natural eutrophication due to their geomorphic properties, such as their low elevation and shallow depth. The main contributing factors to water quality decline in shallow Waikato lakes have been identified as: high external nutrient and sediment loads, internal loading exacerbated by high sediment resuspension and bioturbation from pest fish (carp), reduced water depth due to installation of water control structures, reduction in wetland filtering potential due to draining, and stock access to wetlands (Lehmann et al. 2017). Lake Waahi has been subject to these contributing factors, in addition to the effects of coal mining in the catchment. Recent modelling studies indicate that shallow lake restoration is challenging, especially since the lake has undergone a regime shift to a turbid state.

In a recent study which modelled multiple shallow Waikato lakes (Lehmann et al. 2017), one of the conclusions was that a change to an alternate clear-water stable state would require greater than 50% reduction in external loads of sediment and nutrient, along with pest fish control. If these requirements were met, geochemical engineering was recommended to follow, in order to reduce internal loading from lake sediments (namely phosphate release). This would potentially shift Waikato shallow lakes into macropyte limitation as opposed light limitation (Dokulil & Teubner 2003). However, under any restoration scenario, climate change makes recovery difficult (Rolighed et al. 2016), due to a number of factors but mainly increased surface water temperatures and stratification, increased algal growth, and greater likelihood of anoxia. However, a review of the literature shows that for any lake restoration success external load reduction is critical (Søndergaard et al. 2007). For example, significant and sustained changes in the biological community and water transparency of shallow temperate freshwater lakes cannot be expected to appear unless the total phosphorus (TP) concentration has been reduced to a level below 0.05-0.1 mg P L^{-1} (Jeppesen et al. 2000), for which significant reductions in external load would be needed in many Waikato lakes, including Lake Waahi.

Lake Waahi is a highly culturally significant shallow Waikato lake. The lake has a proud history of being the food source of the Kiingitanga (Kingett 1984). When Maori first colonized the area, tuna (eels) were plentiful and fishing camps were often seen, with tuna hung in trees to dry near the banks of Waahi Stream. There is a long held cultural tradition of tuna fishery preservation. In order to conserve fish stocks a rest period (Rahui pokeka) could be instilled when tuna populations were deemed to be low. Hakanoa was where they danced to lift the Rahui. Historically other important hauanga kai within Lake Waahi were whitebait and kaaeo (freshwater mussel). The Lake Waahi restoration project was commenced in 2011, and one of its main aims is to enhance fish communities, especially tuna. The project has been successful in planting riparian zones around the lake margin and tributaries, working with the iwi and communities to educate about lake water quality.

In order to enhance Lake Waahi, effective lake water quality assessment is needed to determine historical and current water quality trends, and the potential effect of future intensification or restoration initiatives. Water quality assessment may be categorised into three main types: traditional in-situ sampling, numerical modelling and autonomous remote sensing using buoys or satellite imagery (Dekker et al. 1996). *In situ* methods using grab samples are generally suited to monitoring at low temporal resolution. By contrast autonomous water quality monitoring sensors allow for monitoring at high frequency and potentially in real time. However, neither of these methods is well suited to effectively capturing horizontal heterogeneity of water quality and temperature. However, remote sensing using satellite imagery, and water quality modelling can increase the spatial and temporal resolution of ecosystem monitoring.

This report describes a numerical modelling approach for Lake Waahi to assess the effects of various future scenarios on water quality attributes known to affect the appearance of the lake's multiple hauanga kai species across differing spatial scales. The models were used to support shared learning and development of future scenarios in response to different management options and environmental pressures.

Methods

Lake Waahi study site

Lake Waahi (Figs 1 and 2) has a volume of 32,271,160 m^3, and an area of 4,609,200 m^2. The lake was formed when alluvial deposits diverted the original path of the Waikato River, damming valleys and tributaries. Today, it is a relatively shallow, super-eutrophic lake dominated by high turbidity and high algal biomass since the macrophyte collapse in the late 1970s, which was attributed to low lake levels, high nutrient concentrations, and continued sediment input from mining (Dell 1988). Potentially associated with the collapse of macrophytes is the reduction of the biomass of shortfin eels from a catch per unit effort (CPUE) of 1886 in 1987 to 789 in 1992 (Chisnall et al. 1992). Recent fish surveys carried out in 2011 and 2015 also showed a reduction in CPUE (Ratana & Baker 2015), however, the report states that environmental factors may have skewed these results. Potentially related was the large reduction in mysid shrimp populations. While there has been no direct evidence

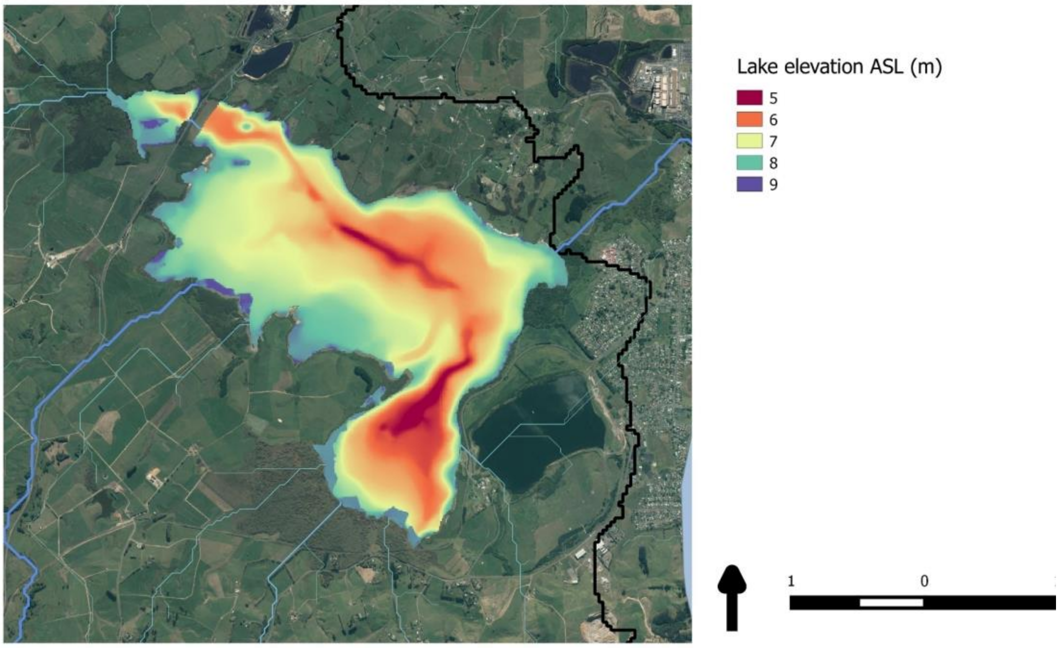


Figure 1. Lake Waahi map. Streams are displayed as blue lines, with lake elevation displayed on a colour ramp ranging from 5 m above sea level to 9 m above sea level.

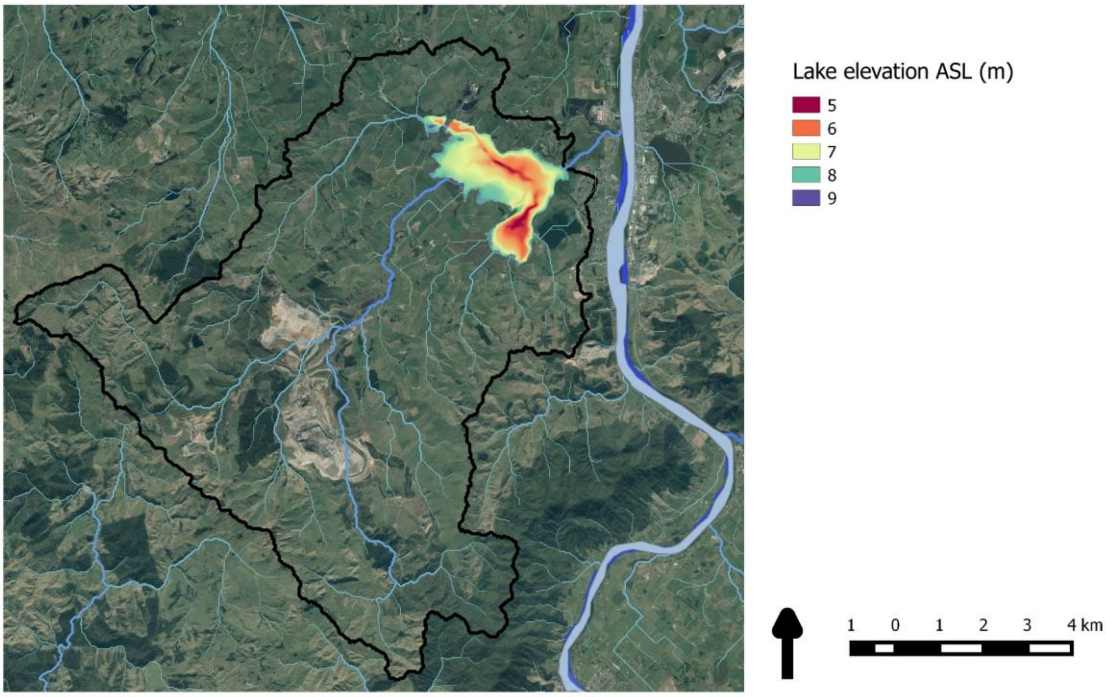


Figure 2. Lake Waahi catchment boundary (black line), showing in-flowing streams derived from the River Environment Classification (National Institute of Water and Atmospheric Research).

found that macrophyte collapse has influenced tuna populations in Lake Waahi, within other turbid aquatic environments the shift to a turbid state resulted in a shift of the foraging strategy of small eels to prey on smaller invertebrates such as *Chironomus* larvae (Kelly & Jellyman 2007). In addition, the loss of macrophytes could potentially result in more eel predation with less sheltering habitat available.

GOTM-FABM-PCLake

A 1-D hydrodynamics model (Fig. 3), the General Ocean Turbulence Model (GOTM, Burchard et. al. 1999) was coupled to a complex ecological model (PClake; Hu et al., 2016, Fig. 4) via the Framework for Aquatic Biogeochemical Models (FABM, Bruggeman and Bolding, 2014). FABM is an open source FORTAN-based code that facilitates the coupling of hydrodynamic and ecological models. FABM-PCLake is a redesigned PCLake (Janse & van Liere 1995).

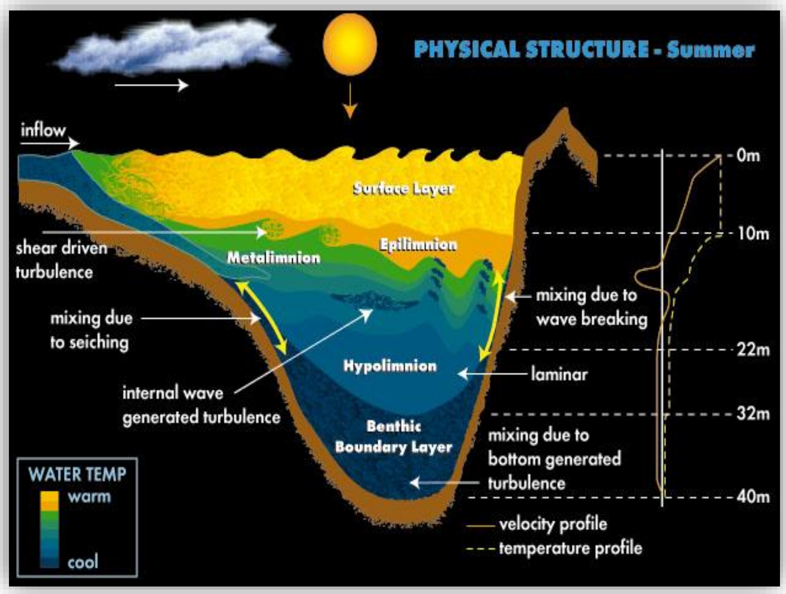


Figure 3. Pictorial representation of physical processes simulated within the 1D hydrodynamic model (GOTM).

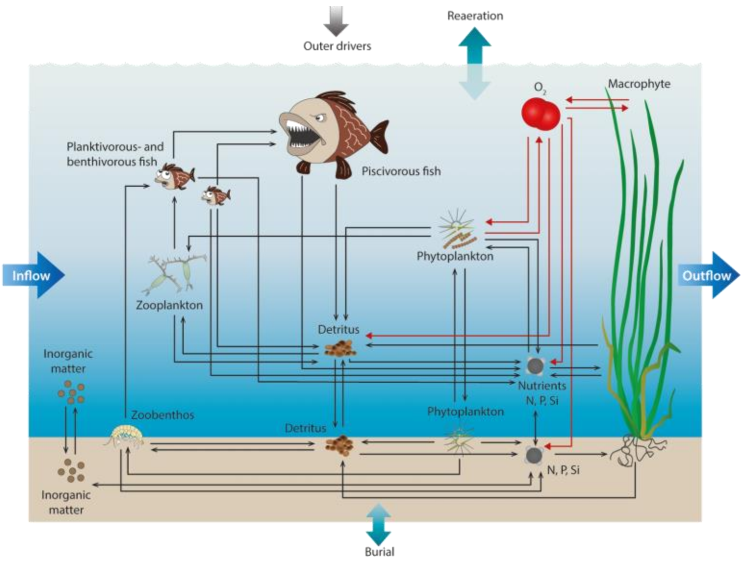


Figure 4. Links between functional processes represented within PCLake.

Lake surface elevation and discharge are monitored by Waikato Regional Council. Lake hypsography was adopted from Jones and Hamilton (2014) (Fig. 5). Inflow data for Lake Waahi was derived from time series flow from a calibrated TopNet model for Lake Waahi (Zammit 2015). In order to incorporate seepage and other factors, TopNet results were adjusted to match CLUES estimations of the average annual inflow (Fig. 6).

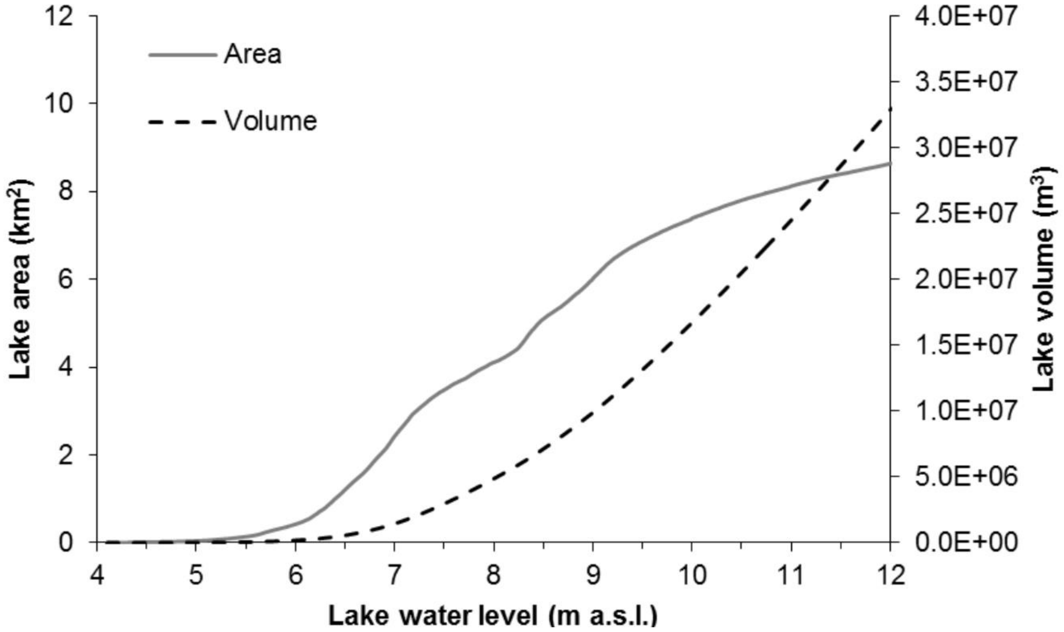


Figure 5. Lake Waahi hypsographic curve from (Jones and Hamilton 2014)

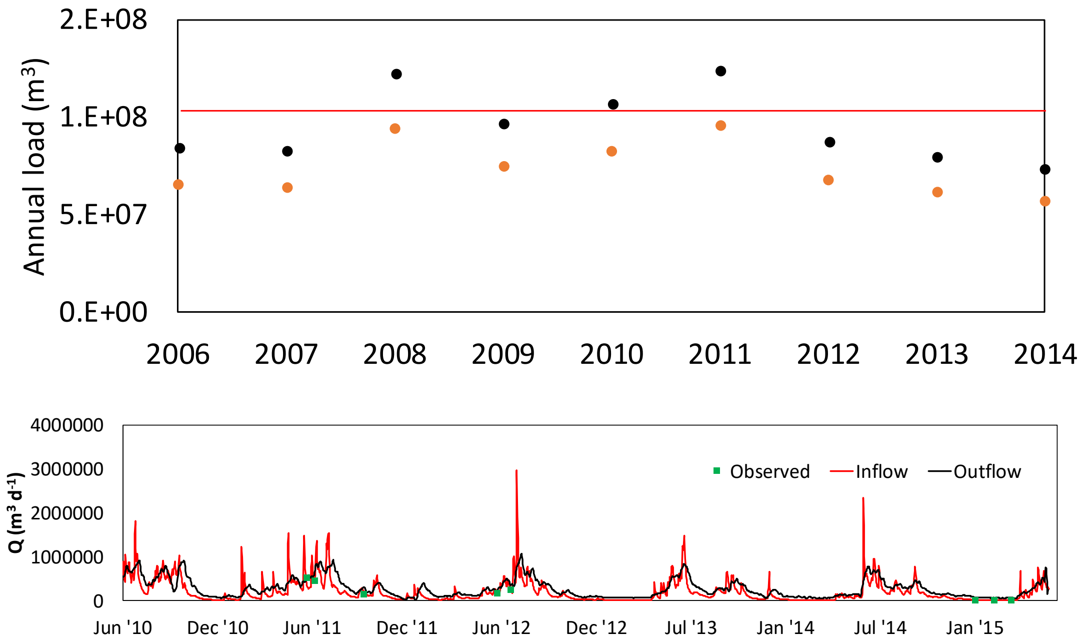


Figure 6. Top: TopNet estimate of the total annual flow (orange), adjusted annual total flow (black) and CLUES annual total flow (red) for Lake Waahi. Bottom: time series inflow volume (red) and outflow (black) volume used to force the model. Green dots represent observed inflow values.

Air temperature measured at the Hamilton Ruakura meteorological station (37.783°S., 175.317°E) was adjusted to Lake Waahi air temperature on the buoy, and used to determine lake surface water temperature, which was then used to define the inflow temperature. The adjustment was based on the relationship between air temperature measured at the Lake Waahi high frequency monitoring buoy and the Ruakura meteorological station data (Fig. 7).

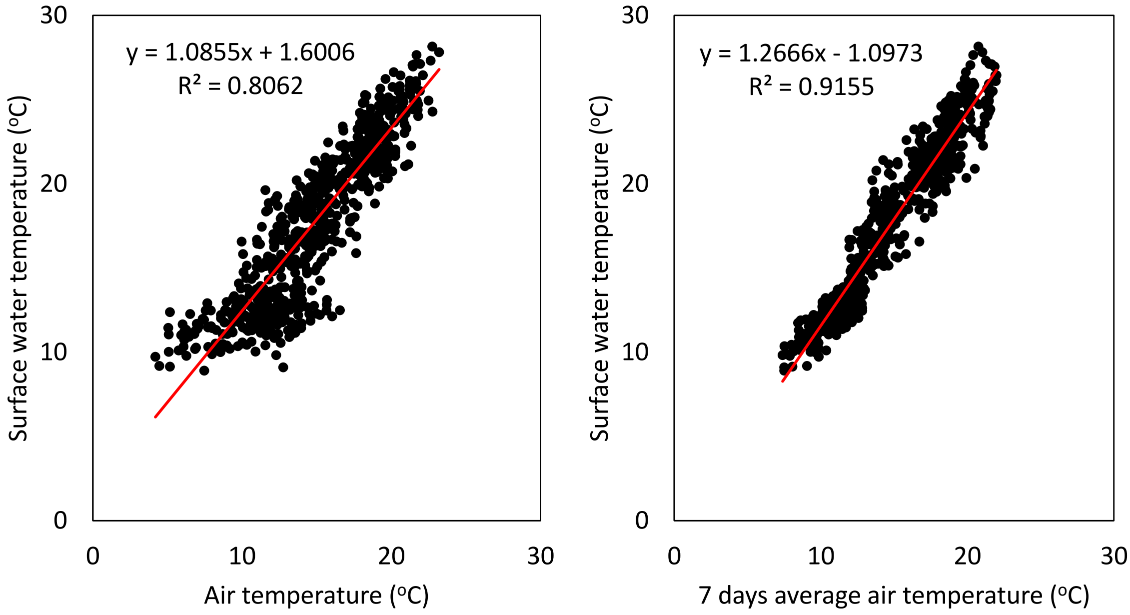


Figure 7. Lake Waahi surface observed temperature and adjusted air temperature at site. This relationship was used to define Lake Waahi inflow water temperature.

Inflow water quality forcing data was primarily taken from the Awaroa Stream monthly monitoring program (**Error! Reference source not found.**), with one exception where inflow suspended solids concentrations were estimated from monthly inflow black disc observations using the following relationship: (West and Scott, 2016)

$$\mathrm{TSS} = \left(\frac{\mathrm{BD}}{4.3}\right)^{\frac{1}{-0.71}} \quad (1)$$

where: TSS is total suspended solids and BD is black desk.

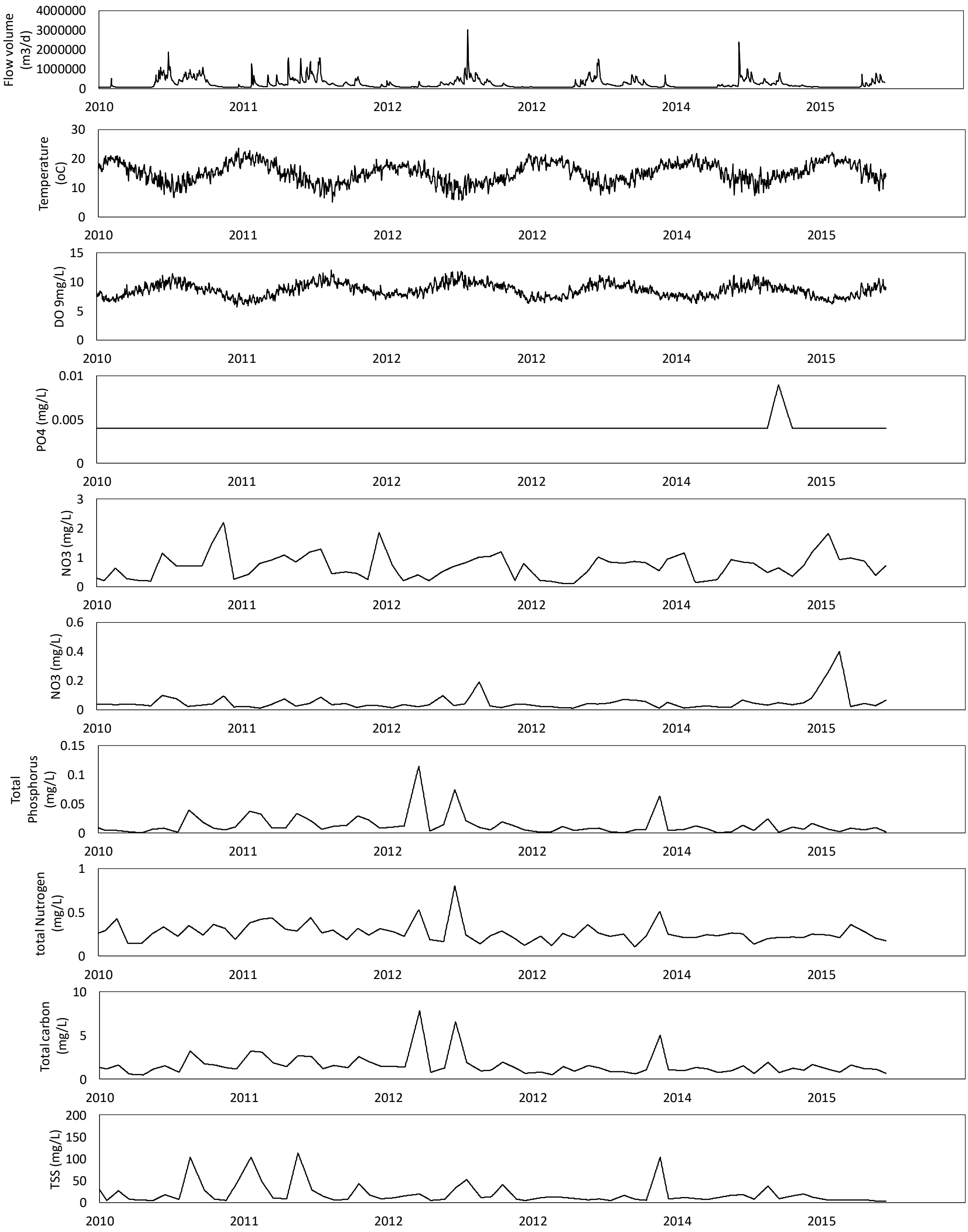


Figure 8. Lake Waahi inflow and concentrations of nutrients in the inflow (expressed as concentrations of PO_4-P, NO_3-N and NH_4-N), and outflow.

PCLake tuna and koi carp

Tuna (shortfin eel or *Anguilla australis*) and koi carp (*Cyprinus carpio*) populations were represented within PCLake using two separate fish groups already included within the model, tuna being assigned to the piscivorous group, and carp assigned to the adult benthivorous group. Both species were simulated at a constant biomass of 50 kg per hectare. The model was then calibrated to match theoretical koi carp and tuna excretion estimated using an allometric scaling model (Morgan & Hicks 2013), with daily lake water temperature in Kelvin (T) simulated by PCLake and assuming a mean fish mass (M) of 610 g fish^{-1}.

$$TP=e^{(b \ln(M) + \ln(Po) - E / (kT))} / 1000 \quad (2)$$
$$TN=e^{(b \ln(M) + \ln(Po) - E / (kT))} / 1000 \quad (3)$$
$$NH_4\text{-}N=e^{(b \ln(M) + \ln(Po) - E / (kT))} / 1000 \quad (4)$$
$$PO_4\text{-}P=e^{(b \ln(M) + \ln(Po) - E / (kT))} / 1000 \quad (5)$$

where: b is the slope and P_o is a normalisation constant for temperature corrected excretion rates in Morgan & Hicks (2013) (see Table 1), k is the Boltzmann constant (8.62 x 10^{-5}), and E is the average activation energy of metabolic reactions (0.65 eV).

The division by 1000 converts the units from μg to mg. These results were scaled up to the entire lake by multiplying by total number of fish within the lake.

With PCLake excretion is model using the following equation (TN):

$$Ex= (rNDFiAd \,/\, cNDFishRef) * kDRespFiAd * uFunTmFish * sNFiAd \quad (6)$$

where: $rNDFiAd$ is the nitrogen to dry weight ratio of fish, $cNDFishRef$ (mgP/mgDW) is the reference nitrogen to carbon ratio, $kDRespFiAd$ (d^{-1}) is the maintenance respiration constant of fish, $uFunTmFish$ is the temperature function of biotic processes, and $sNFiAd$ (gN m^{-2}) is the nitrogen content.

For TP, the equation is the same but with phosphorus substituted for nitrogen. The background respiration constant was modified iteratively until the model excretion matched the estimated excretion using the allometric scaling model. For carp, background respiration was set to 0.001 (d^{-1}) for TP excretion, and 0.0002 (d^{-1}) for TN excretion. For tuna background respiration was set to 0.0000025 for TP excretion, and 0.000025 for TN excretion. For a detailed breakdown of the parameters used in the model see Appendix 1. The fish-specific parameters modified within the model related mainly to fish density and nutrient to dry weight ratios. Nutrient to carbon ratio data were available for carp, however, for tuna the values measured from catfish were adopted, assuming a similar benthic diet would lead to similar nutrient to carbon ratios. The default parameters for carp sediment resuspension due to feeding activity were applied, with the parameter relative resuspension by adult fish browsing of 1.0 (g gfish^{-1} d^{-1}).

Table 1. Least-squares regression estimates of whole-body, temperature-corrected N and P excretion for common carp against the natural log of fish wet mass, directly reproduced from Morgan & Hicks (2013).

Nutrient and rate	N	$\ln(P_0)$	b	r^2	P
$\ln$ (μg NH_4 fish^{-1} h^{-1} e$^{E/kT}$)	28	29.98	0.729	0.91	<0.001
$\ln$ (μg PO_4 fish^{-1} h^{-1} e$^{E/kT}$)	28	28.72	0.439	0.37	0.001
$\ln$ (μg TN fish^{-1} h^{-1} e$^{E/kT}$)	26	31.46	0.595	0.79	<0.001
$\ln$ (μg TP fish^{-1} h^{-1} e$^{E/kT}$)	26	31.28	0.330	0.26	0.008

Table 2. PCLake fish-related parameters which were set to non-default values.

PARAMETER NAME	PARAMETER LONG NAME	DEFAULT	SOURCE
KMIGRFISH: 0.1	fish migration rate (d-1)	0.0010	Calibrated
CDFIJVIN: 0.1	external fish density (gDW m-2)	0.0050	Calibrated
CDFIADIN: 1.7	external fish density (gDW m-2)	0.0050	Calibrated
CDPISCIN: 1.7	external Pisc. density (gDW m-2)	0.0010	Calibrated
KMIGRPISC: 0.1	piscivorous migration rate (d-1)	0.0010	Calibrated
CDAYREPRFISH: 245.0	reproduction day of year for fish ([-])	120.0000	Calibrated
CDAYAGEFISH: 365.0	aging day of year for fish ([-])	365.0000	Calibrated
CDCARRPISCMAX: 2	maximum carrying capacity of piscivorous fish (gDW m-2)	1.2000	Calibrated
CDCARRPISCMIN: 1.6	minimum carrying capacity of piscivorous fish (gDW m-2)	0.1000	Calibrated
CDCARRPISCBARE : 2	carrying capacity of piscivorous fish (gDW m-2)	0.1000	Calibrated
CDPHRAMINPISC: 0	minimum reed biomass for piscivorous fish (gDW m-2)	50.0000	Calibrated
CCOVVEGMIN: 0.0	minimum submerged macrophytes coverage for piscivorous fish (%)	40.0000	Calibrated
CPDFISHREF: 0.0259	reference P/C ratio of fish (mgP/mgDW)	0.0220	Hicks et al. unpubl.
CNDFISHREF: 0.148	reference N/C ratio of fish (mgN/mgDW)	0.1000	Hicks et al. unpubl.
CPDPISC: 0.0878	reference P/C ratio of piscivorous fish (mgP/mgDW)	0.0220	Hicks et al. unpubl.
CNDPISC: 0.268	reference N/C ratio of piscivorous fish (mgN/mgDW)	0.1000	Hicks et al. unpubl.
CDFIADMIN: 0.1	minimum benthivorous fish biomass in system (gDW m-2)	0.0001	Calibrated
CDPISCMIN: 1.7	minimum piscivorous fish biomass in system (gDW m-2)	0.0001	Calibrated

Remote sensing

Satellite imagery and software

A detailed description of the remote sensing method used in the present study can be found in Allan (2016). USGS on-demand, atmospherically-corrected Landsat imagery was ordered from http://espa.cr.usgs.gov/index/. Atmospheric correction applies the radiative transfer model 6sv (Second Simulation of a Satellite Signal in the Solar Spectrum), which corrects for atmospheric scattering, and absorption effects of gases and aerosols (Kotchenova et al. 2008). Landsat 7 and 8 satellite images were captured between 1999 and 2015. All image processing routines were automated using scripts written in Interactive Data Language (IDL). Maps of derived TSS outputs were generated using Quantum GIS (QGIS). The study downloaded all available Landsat 7 and 8 images (235 images) from Landsat path/row 072/087 (84 L7, 17 L8) and 073/086 (118 Landsat 7, 16 L8).

TSS was estimated from Landsat subsurface remote sensing reflectance using both empirical and semi-analytical algorithms similar to those found in Allan et al. (2015) and Dekker et al. (2002). The empirical equation adopted to estimate TSS used symbolic regression was:

$$TSS = 22983.65 \; B3 \; B4 \qquad (7)$$

where: B3 and B4 represents atmospherically-corrected subsurface reflectance in bands three and four for Landsat 7.

For the application to Landsat 8 these bands were substituted for bands four and five. The application of this relationship to estimate TSS resulted r^2 0.87, and a RMSE 25.5 mg L^{-1} (n = 24).

Linear spectral unmixing

For the estimation of chlorophyll *a* (chl *a*) within optically-complex, turbid Waikato shallow lakes a more complex algorithm is required that accounts for the fact that TSS and chlorophyll (and sometimes coloured dissolved organic matter (DOM)) interact to produce a mixed pixel-mixed reflectance satellite image. Linear spectral unmixing (LSU) is an inversion technique that aims to estimate the mixture components responsible for the mixed spectral signature of a pixel, defined as a linear-matrix equation:

$$R_i = \sum_{k=n}^{n} f_k R_{ik} + \varepsilon_i \qquad (8)$$

where: R_i is the reflectance of the mixed pixel for each band *i*, including one or more end-members; f_k is the fraction of each end-member *k* within the pixel, R_{ik} is the reflectance of the end-member *k* within the pixel on band *i*, and ε_i is the error for band *i* (or the difference between the measured and modelled DN in each band).

To solve these equations, the end-members must be independent from each other, the number of end-members should be less than or equal to the number of spectral bands used, and the spectral bands used should not be highly correlated.

The water surface reflection received by satellites is a mixed spectrum as it is affected by chl *a*, TSS, DOM concentrations, as well as other factors. LSU attempts to unmix this signature, deriving proportion estimates of the mixture components. These mixture components are referred to as end-members, which are idealized pure signatures. In this study, an image based end-member selection process was used where bands are plotted in feature space, and end-members identified manually using a similar approach to Tyler et al. (2006). Spectrally-pure end-members are found at the vertices of the polygon bounding the data cloud in feature space. A polynomial-based symbolic regression derived relationship was used to estimate chl *a* concentration from the chlorophyll end-member. Observed vs. estimated chl *a* resulted in an r^2 of 0.34, and RMSE of 27.4 µg L^{-1} (Fig. 12):

$$\text{Chl } a = 12.17 + 508.43 \text{ chl\_end-member}^3 - 163.46 \text{ chl\_end-member}^4 \quad (9)$$

It should be noted that within the error analysis relatively clear lakes were included, for which the symbolic regression in members were not derived, and potentially not applicable. Therefore accuracy of this method needs to be further investigated for target lakes including Lake Waahi.

Water colour from modelling

Modelled water colours were represented using standardised red green blue (sRGB) colour space allowing colour to be directly displayed on monitors/projectors and print. PCLake estimated chl *a* and TSS were applied to bio-optical models which estimated remote sensing reflectance $r_{rs}(\lambda)$, which was then resampled to chromaticity coordinates, and finally to sRGB. The bio-optical model (eq. 9-15) was run on a daily time step for each PCLake output (including nine depth layers) to estimate an above surface reflection spectrum. The relationship between $r_{rs}(\lambda)$ and the total backscattering coefficient b_b (m^{-1}) and total absorption a (m^{-1}) is:

$$r_{rs}(\lambda) = g_0 u(\lambda) + g_1 [u(\lambda)]^2 \quad (9)$$

where: u is defined as (from Dekker et al. 1996):

$$u(\lambda) = \frac{b_b(\lambda)}{a(\lambda) + b_b(\lambda)} \quad (10)$$

where: g_o and g_1 are empirical constants that depend on the anisotropy of the downwelling light field and scattering processes within the water. The constant g_o is equivalent to f/Q where f represents geometrical light factors and Q represents the light distribution factor, which is defined as upwelling subsurface irradiance/upwelling subsurface radiance.

The absorption and backscattering coefficients are comprised of individual, optically-active constituents:

$$b_b(\lambda) = b_{bw}(\lambda) + B_{bSP}\, b^*_{SP}(\lambda)\, C_{SP} \quad (11)$$

$$a(\lambda) = a_w(\lambda) + C_\phi\, a^*_\phi(\lambda) + a_{CDOMD}(\lambda) \quad (12)$$

$$a_{CDOMD}(\lambda) = a_{CDOMD}(\lambda_{440}) \exp[-S\,(\lambda - \lambda_{440})] \quad (13)$$

where:
$b_{bw}(\lambda)$ = backscattering coefficient of water,

B_{pSP} = backscattering ratio from SP,
$b^*_{SP}(\lambda)$ = specific scattering coefficient of SP,
C_{SP} = concentration of SP or TSS,
$b^*_{\phi}(\lambda)$ = specific scattering coefficient of phytoplankton,
$a_w(\lambda)$ = absorption coefficient of pure water,
C_{ϕ} = concentration of chl *a*,
$a^*_{\phi}(\lambda)$ = specific absorption coefficient of phytoplankton,
$a_{CDOMD}(\lambda)$ = absorption coefficient for coloured dissolved organic matter (CDOM),
S = spectral slope coefficient

Values of $a_w(\lambda)$ and $b_{bw}(\lambda)$ were prescribed from the literature (Pope & Fry 1997). The backscattering ratio of SP, B_{pSP}, was set to 0.011. The specific scattering coefficient of SP was estimated using a power function (Morel & Prieur 1977):

$$b^*_{SP}(\lambda) = b^*_{SP}(555)\left(\frac{555}{\lambda}\right)^n \tag{14}$$

where: the value b^*_{SP} (555) was set to 0.6 m^2 g^{-1}. The hyperbolic exponent n was set to 0.63, equating to a value measured in Lake Taupo, New Zealand (Belzile et al. 2004). The specific absorption coefficient of phytoplankton $a^*_{\phi}(\lambda)$ was adopted from the average value measured in eight Dutch lakes (Dekker et al. 2002). For cyanobacteria concentrations $a^*_{\phi}(\lambda)$ was adopted for cyanobacteria sp. The $a_{CDOMD(440)}$ was fixed at 0.6011 m^{-1} (measured in Waahi, Uyen Nguyen pers. comm.).

The $r_{rs}(\lambda)$ was then converted into chromaticity coordinates (International Commission on Illumination (CIE)) XYZ using equations described in detail within Dierssen et al. (2006).

$$x=X/(X+Y+Z),\ y=(Y/(X+Y+Z) \tag{15}$$

Trophic Level Index calculation

The lake TLI value was calculated for each year of the simulation period to indicate overall changes in water quality. The relevant equations for determination of the TLI are:

$$\mathrm{TL}_{\mathrm{Chl}a} = 2.22 + 2.54\ \log(\mathrm{Chl}a) \tag{16}$$

$$\mathrm{TL}_{\mathrm{SD}} = 5.1 + 2.27\ \log\left(\frac{1}{\mathrm{SD}} - \frac{1}{40}\right) \tag{17}$$

$$\mathrm{TL}_{\mathrm{TP}} = 0.218 + 2.92\log(\mathrm{TP}) \tag{18}$$

$$\mathrm{TL}_{\mathrm{TN}} = -3.61 + 3.01\ \log(\mathrm{TN}) \tag{19}$$

$$\mathrm{TLI\ \text{-}\ 4} = \frac{1}{4}\sum(\mathrm{TL}_{\mathrm{Chla}}, \mathrm{TL}_{\mathrm{SD}}, \mathrm{TL}_{\mathrm{TP}}, \mathrm{TL}_{\mathrm{TN}}) \tag{20}$$

$$\mathrm{TLI\ \text{-}\ 3} = \frac{1}{4}\sum(\mathrm{TL}_{\mathrm{Chla}}, \mathrm{TL}_{\mathrm{TP}}, \mathrm{TL}_{\mathrm{TN}}) \tag{21}$$

where:
TL_{Chla}, TL_{SD}, TL_{TP} and TL_{TN} represent the individual trophic level indices for the variables of chl *a* (mg/m^3), Secchi depth (m), TP (mg/L) and TN (mg/L), respectively.

As Secchi depth is not explicitly included in the model, this variable was derived from the model-predicted attenuation coefficient for photosynthetically active radiation as (Holmes 1970):

$$z_{SD} = 1.44\, K_d \qquad (22)$$

where:

z_{SD} is the Secchi depth (m), K_d is the diffuse attenuation coefficient for PAR (m^{-1}), K_d is calculated as (Gallegos 2001):

$$K_d = K_w + K_c * chl_a + K_y * DOC + K_s * TSS \qquad (23)$$

where:

K_w is the background extinction coefficient with a value of 0.3315 (m^{-1})

K_c is the specific attenuation coefficient for chl *a* with a value of 0.0122 (m^2 (mg chl *a*)$^{-1}$)

K_y is the specific attenuation coefficient for dissolved organic carbon with a value of 0.0507 (m^2 (g DOC)$^{-1}$)

K_s is the specific attenuation coefficient for total suspended sediment with a value of 0.0778 (m^2 (g TSS)$^{-1}$)

Potential scenarios

In this project a collaborative approach to lake restoration scenario development was sought which included consultation with the programme Technical Advisory Group, members of the Waahi Whanui Trust, and community members. The scenarios were inspired by the following:

> *"Waikato-Tainui aspires to have waters that are drinkable, swimmable, and fishable with the water quality at least at the level it was when Kiingi Taawhiao composed his maimai aroha…"* (Waikato-Tainui 2013).

The presentation of modelling results was designed to be easily interpreted.

Scenarios were developed progressively (Table 3), with the ultimate goal being to create a scenario representing an aspirational future that would be valued by traditional collectors of Hauanga kai on Lake Waahi.

The scenarios included manipulation of external load through to integrated catchment management involving a process whereby plans comprise of formal processes covering management over large areas (Feeney et al. 2010). For example, ICM can refer to methods that reduce the risk of soil erosion and flooding, reduce sediment entering waterways,

Table 3. Simulated scenarios within the study, and primary scenario assumptions. CC is climate change, WL is water level, NF is no fish (realistically 10 kg ha-1), ICM is integrated catchment management, ICMG is integrated catchment management-Gold, MAC is macrophyte establishment, and DREG is dredging.

SCENARIO NAME	SCENARIO ASSUMPTIONS
S0	Current conditions
S0 CC	Increase in average temperature by 3°C
S1 WL	1 m increase in water level
S1 WL NF	1 m increase in water level, with carp control to a biomass 10 kg ha^{-1}
S2 NF	Carp control to a biomass 10 kg ha^{-1}
S3 ICM MAC NF	Integrated catchment management resulting in reduction of external load of TN and TP by 50% (McDowell et al. 2013), with carp control to a biomass 10 kg ha^{-1}
S3 ICM CC MAC NF	integrated catchment management resulting in reduction of external load of TN and TP by 50% (McDowell et al. 2013), including an increase in average temperature by 3°C, with carp control to a biomass 10 kg ha^{-1}
S4 ICMG MAC NF	integrated catchment management resulting in reduction of external load of TN and TP by 60% (McDowell et al. 2013), with a 87 % reduction in suspended sediment, with carp control to a biomass 10 kg ha^{-1}
S4 ICMG CC NF	integrated catchment management resulting in reduction of external load of TN and TP by 60% (McDowell et al. 2013), with a 87 % reduction in suspended sediment(Smith 1989), Increase in average temperature by 3°C, with carp control to a biomass 10 kg ha^{-1}
S5 MAC	Macrophyte re-establishment
S5 MAC NF	Macrophyte re-establishment with carp control
S6 MAC WL	Macrophyte re-establishment and a 1 m increase in water level
S6 MAC WL NF	Macrophyte re-establishment, a 1 m increase in water level, and can't control
S7 DREG MAC	Lake sediment dredging, and macrophyte re-establishment, average depth increases to 5.3 m, resulting from multiplying depth by 1.1.
S8 DREG MAC NF ICM	Lake sediment dredging, macrophyte re-establishment, average depth increases to 5.3 m (resulting from multiplying depth by 1.1 an increasing water level by 0.3 m) combined with a 60% reduction in external load of TP and TN
S9 CARP75	Carp biomass at 75 kg ha^{-1}
S10 CARP300	Carp biomass at 300 kg ha^{-1}

improve river stability and habitat, and finally improve water quality. Within the context of the current study for the Lake Waahi catchment, ICM is envisaged to include management of wetlands and riparian zones, combined with on-farm nutrient loss mitigation practices. Multiple mitigation strategies have been assessed in terms of effectiveness and cost within McDowell et al. (2013; Figs 3-5).

Within-lake management strategies simulated by the model include the re-establishment of macrophytes, raised water levels, dredging and carp removal. The influence of climate change was estimated simplistically at a worse-case scenario by increasing the mean temperature by an average of 3°C within the model meteorological forcing conditions. Both carp removal scenarios (to a biomass of 10 kg ha^{-1}), and increases in carp biomass above literature cited critical biomasses of 75 and 300 kg ha^{-1} (S9 and S10 respectively) were run. A recent literature review on the effects of common carp on water quality revealed that a significant increase in turbidity could occur at biomasses between 50 and 75 kg ha^{-1} (Vilizzi et al. 2015), and sudden nonlinear changes from clear water to turbid water associated with carp biomasses of between 174 and 300 kg ha^{-1}. Most scenarios were presented with no invasive fish (NF) and without the effects of invasive fish removal.

Results

Model calibration

The PCLake 1-D ecologically-coupled, hydrodynamic model was calibrated over the time period spanning 1 January 2010 until 15 June 2015 (Table 4). The statistical comparison between simulated and observed water quality variables was deemed reasonable in comparison to other lake modelling studies. The NRMSE between simulated and observed variables ranged from 9% (temperature) to 44% (nitrate). The average error of the variables listed in Table 4 was 25%. This shows that the PCLake provided a reasonable representation of the biological and physical conditions observed in Lake Waahi.

Table 4. Errors in state variable estimation within Lake Waahi, whereby errors are calculated between *in situ* samples captured at the lake surface. TLI4 is trophic level index including chlorophyll *a*, TP, TN and Secchi depth. NRMSE is normalised root mean squared error.

VARIABLE	RMSE	NRMSE (%)
TEMPERATURE (^{O}C)	1.58	0.09
DISSOLVED OXYGEN BOTTOM (g m^{-3})	1.51	0.16
DISSOLVED OXYGEN (g m^{-3})	0.94	0.12
SECCHI DEPTH (m)	0.32	0.43
TSS (g m^{-3})	88.48	0.30
TP (g m^{-3})	0.05	0.25
TN (g m^{-3})	0.58	0.23
NO_3 (g m^{-3})	0.30	0.44
NH_4 (g m^{-3})	0.08	0.22
CHLA (g m^{-3})	24.55	0.25
TLI4	0.49	0.25

Stratification was well represented within the model (Fig. 9) and agreed with observed data. Modelled temperature displayed polymictic stratification whereby no seasonal stratification is present. During summer, short periods of stratification persisted in the order of days to a few weeks. Dissolved oxygen (DO) concentrations in surface and bottom waters were also reasonably well simulated by the model (Fig. 9), however, during the late summer of 2015 buoy measured DO reduced to c. 2 mg L^{-1} while simulated concentrations during summer only reached a minimum of c. 4.1 mg L^{-1}.

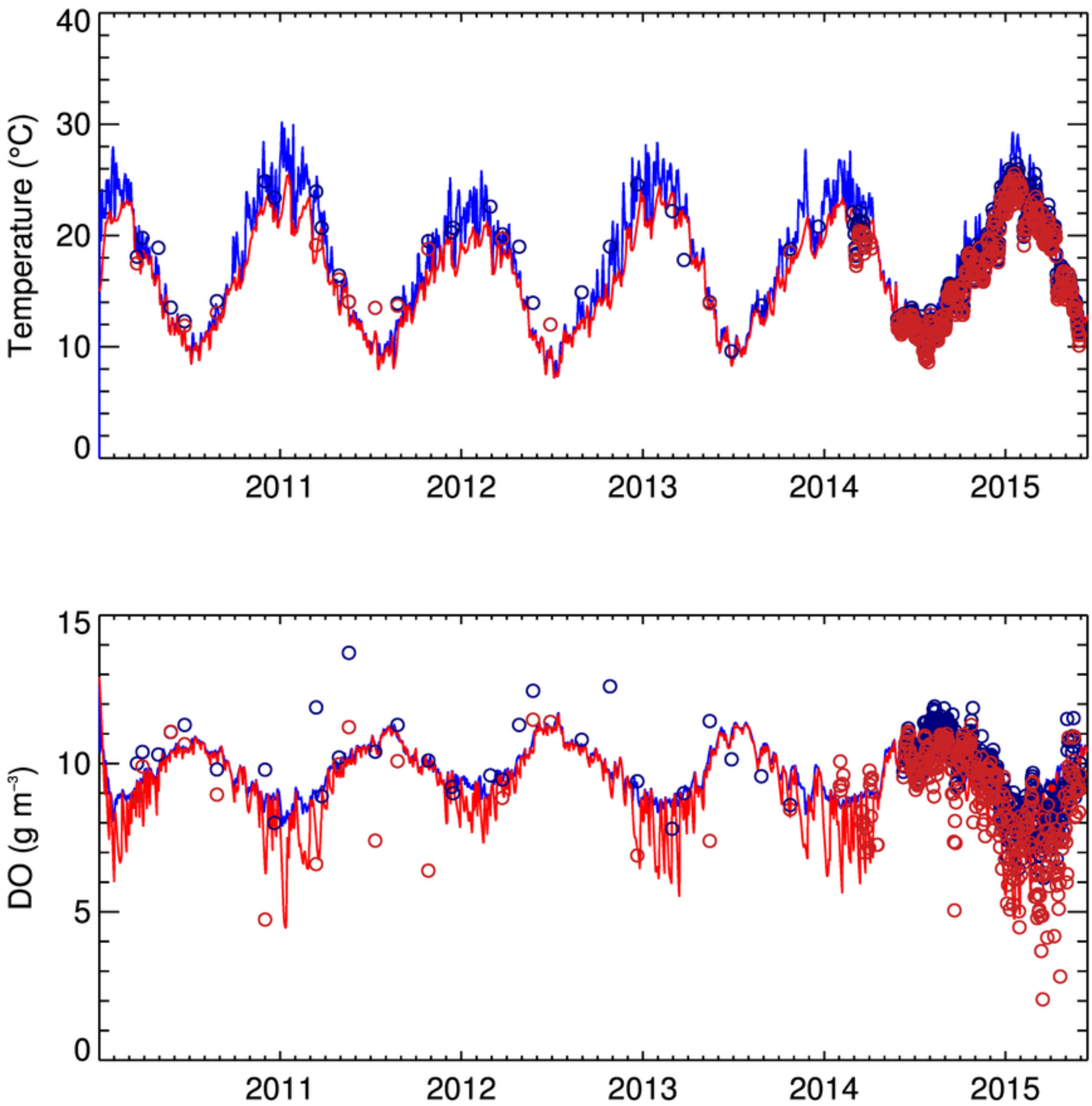


Figure 9. Surface (blue line) and bottom temperature (redline) and dissolved oxygen (DO) with field observed surface equivalent (blue open circle), and bottom (read open circle).

Most nutrient-related parameters were well represented within the simulations (Fig. 10), however, large observed peaks in TP during the summer of 2013 and 2014 were not well simulated within the model. These events coincided with high observed TSS (Fig. 11), which were also not well represented within the simulations. During these times, chl *a* was among the highest simulated and observed over the simulation period. It may be that these high TP events were associated with high algal biomass, or associated with desorption of phosphorus bound to suspended minerals. Under the calibration scenario (S0), macrophyte maximum standing crop was limited to 10 g DW m^{-2} in order to represent the current de-vegetated lake state.

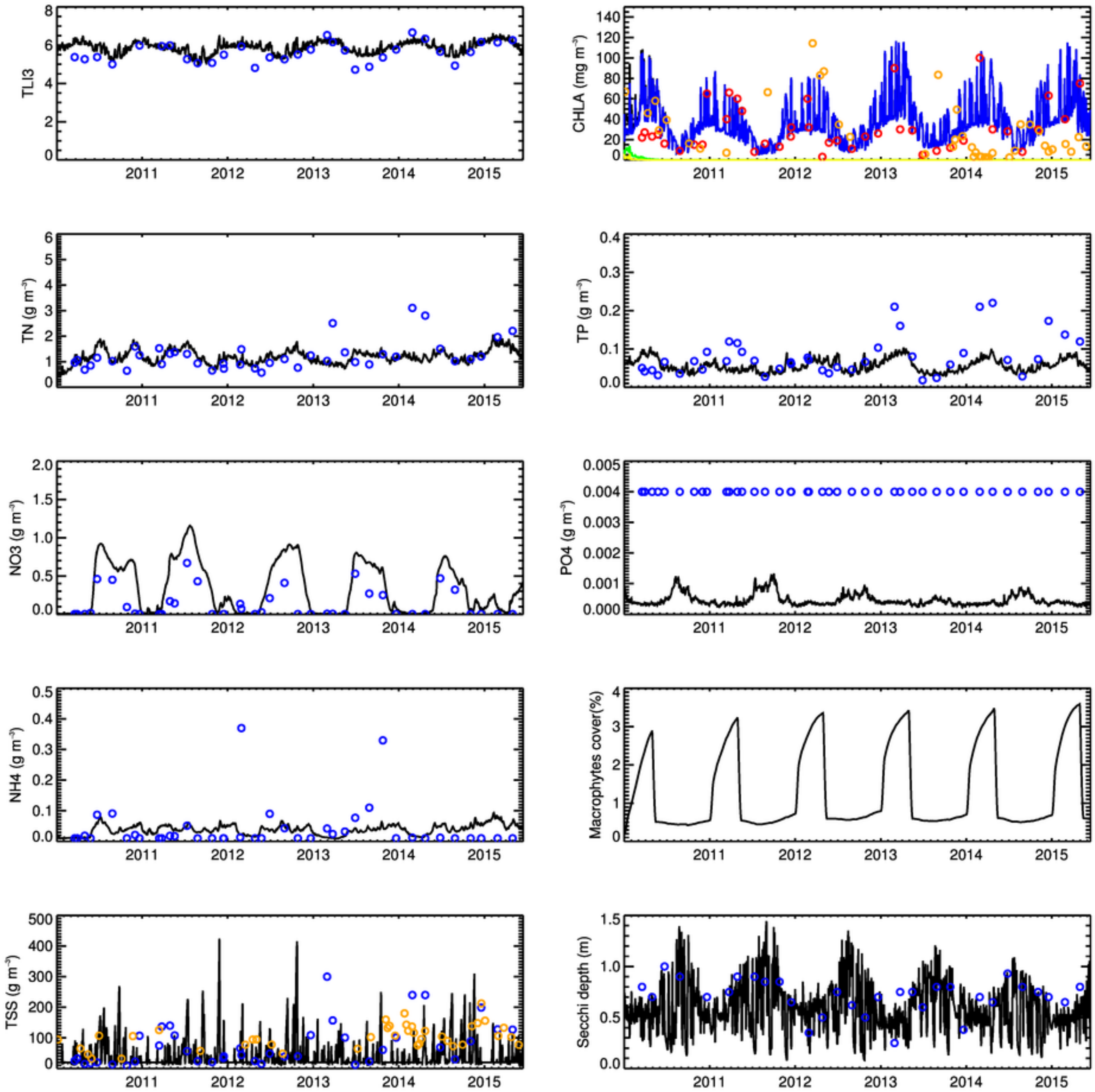


Figure 10. Calibration scenario (S0) displaying physical and biological parameters simulated by PCLake in Lake Waahi (black line) with observed (blue open circles) and satellite observed (orange open circles). Simulated cyanobacteria (top right, blue line) comprised of the majority of simulated chlorophyll a (black line), along with green algae (green line) and diatoms (yellow line), which are only visible at the start of the simulation. Observed chlorophyll *a* is represented using an open blue circle.

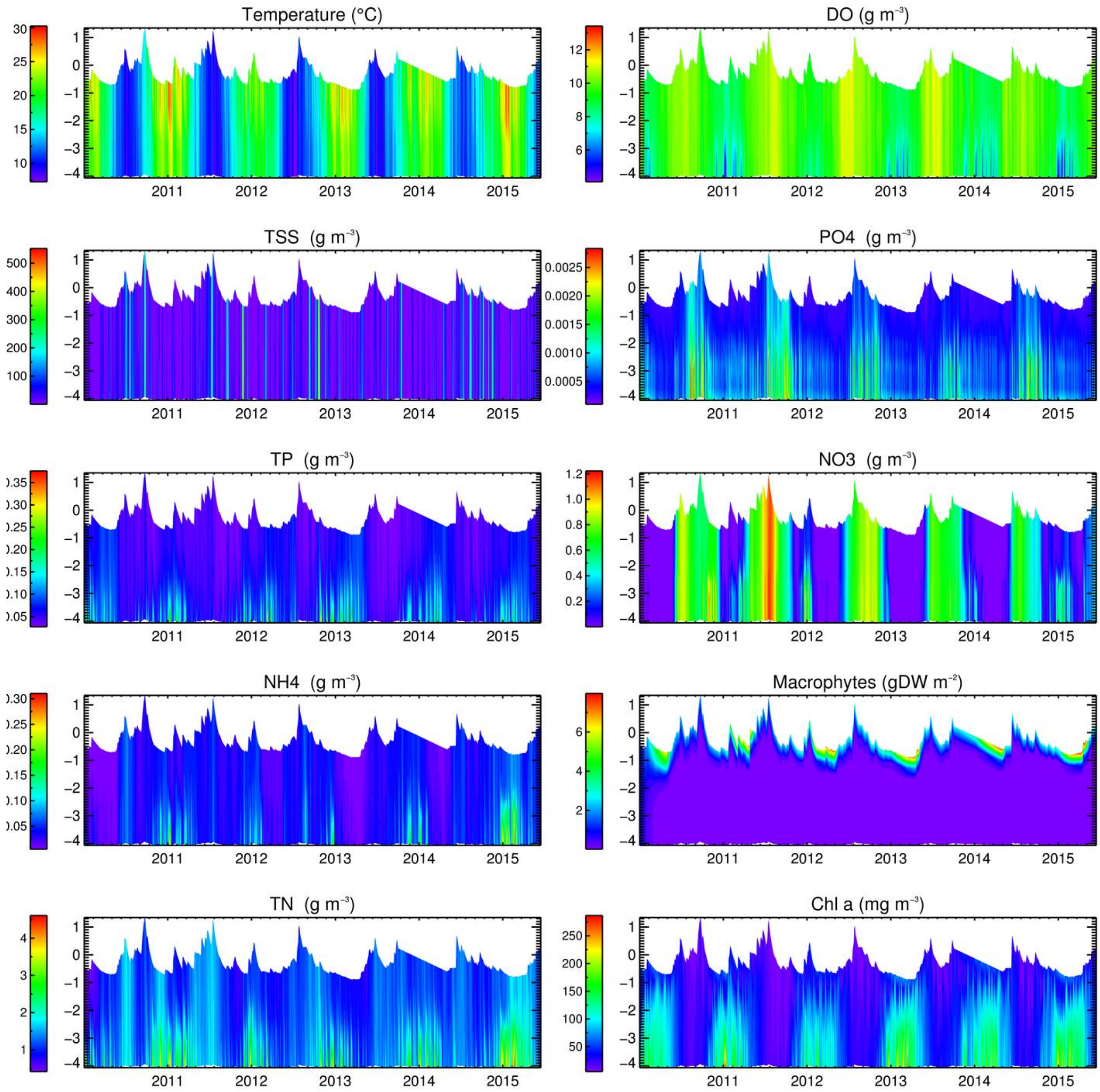


Figure 11. Colour contour plots of the calibration scenario (S0) displaying physical and biological parameters simulated by PCLake in Lake Waahi.

Tuna (shortfin eel) and koi carp excretion

Carp and tuna excretion were well simulated by the model, with seasonal dynamics of higher excretion and summer month's related to higher temperature and metabolic rates.

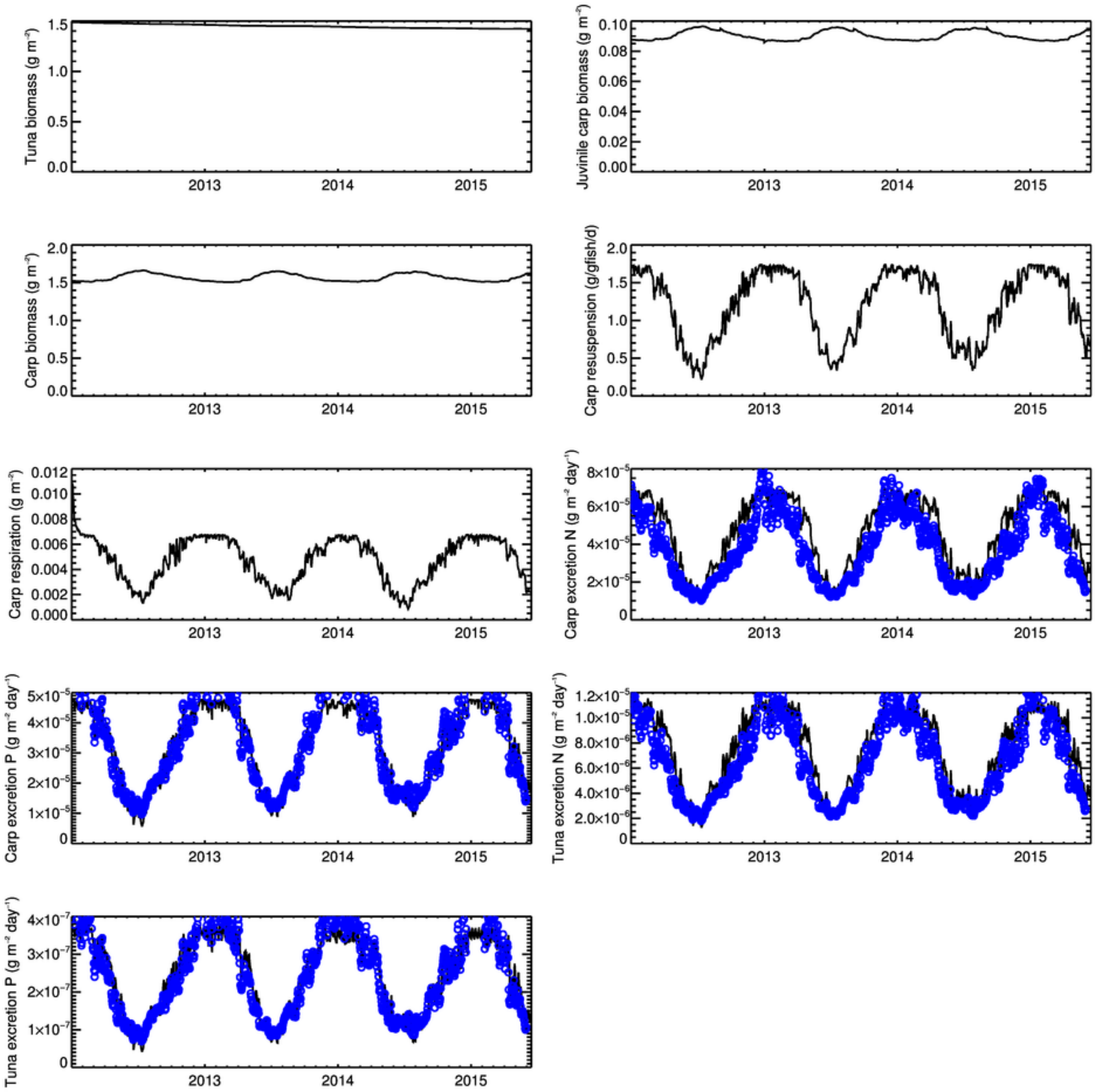


Figure 12. Calibration scenario (S0) state variables related to simulation of carp and to tuna biomass and excretion. The black line represents simulated data, with blue open circles representing excretion derived from empirical relationships.

Scenario results

The scenario simulations covered a broad range of TN, TP, and chl *a* (Fig. 13). Scenario 4 (S4 ICMG MAC NF**)** simulated the largest decrease in nutrient concentrations and chl *a*, and increases in Secchi depth. This is not surprising as this scenario had the largest reduction in external load, and also included the effect of macrophyte re-establishment and carp control. Out of all scenarios, the greatest reduction in within lake nutrient concentrations and increases in Secchi depth were simulated where scenarios combined external load reduction with internal lake manipulation (S3 and S4). Carp control alone (S2), or macrophyte re-establishment alone (S5 MAC) had a relatively small influence on water quality or clarity compared to combining these with external load reductions.

Scenario 1 simulated the influence of increasing water levels. While the increased water level reduced suspended sediment by c.50% (Fig. 14), stratification increased, which favoured cyanobacteria growth and enhanced nutrient release mediated by greater de-oxygenation of hypolimnion. However, adding carp control to the scenario (S1 WL NF) resulted in a lower trophic status than the calibration scenario, which was driven by higher water clarity, lower TSS, TP and TN.

Scenario 2 simulated the effect of carp control alone could still significantly decrease trophic status with a 17% reduction in TP, and a 25% increase in Secchi depth. I note here that there is significant uncertainty in the parameterisation of carp within modelling. In particular carp resuspension rates can vary greatly depending on food availability.

Scenario 3 simulated that a 50% reduction in external load alongside carp control and macrophyte re-establishment could double water clarity and result in a greater than 50% reduction of TN, TP and TSS.

Scenario 4 simulated that 60% reduction in external load alongside carp control and macrophyte re-establishment resulted in 120% increase in water clarity, a 65% reduction in TP, a 70% reduction in TN, and a 56% reduction in TSS. However, with the addition of climate change, these water quality gains were effectively reduced by a factor of approximately 50% in the case of chl *a* and Secchi depth.

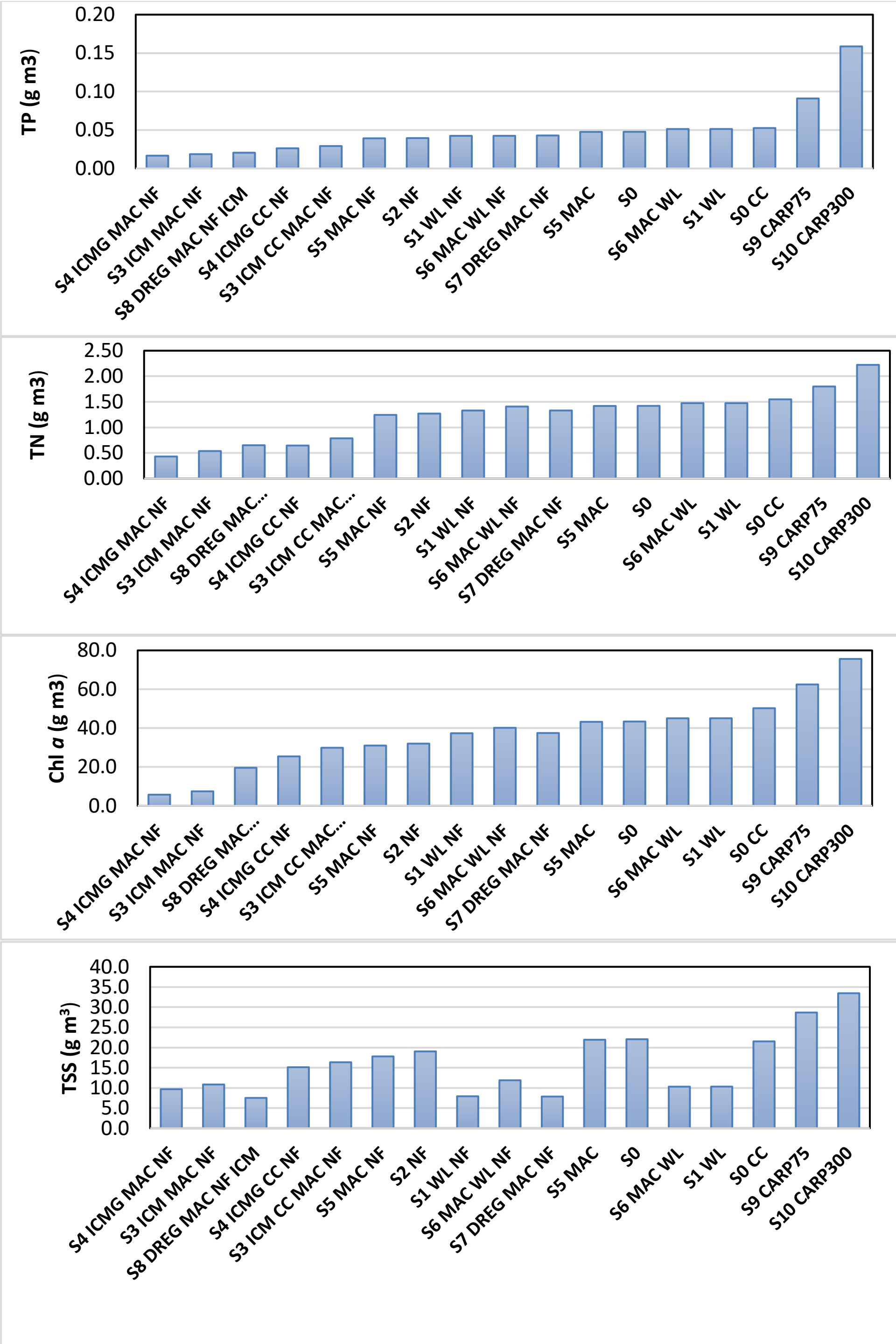
Scenario 5 simulated macrophyte re-establishment, however, without the reduction in external loading macrophyte coverage only reached 16.6%, in contrast to 51% for S3 ICM MAC NF. Therefore there were only minor reductions in TSS. When the scenario was run with carp control, macrophyte coverage increased to 28%, with a 29% increase in water clarity, and an 18% reduction in TP.

Scenario 6 (S6 MAC WL NF) simulated that an increase in water level leads to lower concentrations of TSS, however, macrophytes only reached a coverage of 6%, potentially due to greater light limitation associated with increased depth.

Scenario 8 simulated the effect of dredging, macrophyte re-establishment, and carp removal alongside integrated catchment management. Macrophytes only reached a coverage of 9%, again potentially due to light limitation, however, the combination of reduced TSS and external nutrient loads resulted in a 71% increase in water clarity, a 57% and 54% reductions in TP and TN, respectively, and a 16% reduction in the TLI4.

Scenario 9 simulated a carp biomass of 75 kg per hectare, and increased TP concentrations by 91%, chl *a* by 44%, with a 30% reduction in water clarity.

Scenario 10 simulated a carp biomass of 300 kg per hectare, and increased TP concentrations by 233%, chl *a* by 74%, with a 43% reduction in water clarity.


TP (g m3)
0.20
0.15
0.10
0.05
0.00
S4 ICMG MAC NF
S3 ICM MAC NF
S8 DREG MAC NF ICM
S4 ICMG CC NF
S3 ICM CC MAC NF
S5 MAC NF
S2 NF
S1 WL NF
S6 MAC WL NF
S7 DREG MAC NF
S5 MAC
S0
S6 MAC WL
S1 WL
S0 CC
S9 CARP75
S10 CARP300
TN (g m3)
2.50
2.00
1.50
1.00
0.50
0.00
S4 ICMG MAC NF
S3 ICM MAC NF
S8 DREG MAC...
S4 ICMG CC NF
S3 ICM CC MAC...
S5 MAC NF
S2 NF
S1 WL NF
S6 MAC WL NF
S7 DREG MAC NF
S5 MAC
S0
S6 MAC WL
S1 WL
S0 CC
S9 CARP75
S10 CARP300
Chl a (g m3)
80.0
60.0
40.0
20.0
0.0
S4 ICMG MAC NF
S3 ICM MAC NF
S8 DREG MAC...
S4 ICMG CC NF
S3 ICM CC MAC...
S5 MAC NF
S2 NF
S1 WL NF
S6 MAC WL NF
S7 DREG MAC NF
S5 MAC
S0
S6 MAC WL
S1 WL
S0 CC
S9 CARP75
S10 CARP300
TSS (g m3)
40.0
35.0
30.0
25.0
20.0
15.0
10.0
5.0
0.0
S4 ICMG MAC NF
S3 ICM MAC NF
S8 DREG MAC NF ICM
S4 ICMG CC NF
S3 ICM CC MAC NF
S5 MAC NF
S2 NF
S1 WL NF
S6 MAC WL NF
S7 DREG MAC NF
S5 MAC
S0
S6 MAC WL
S1 WL
S0 CC
S9 CARP75
S10 CARP300

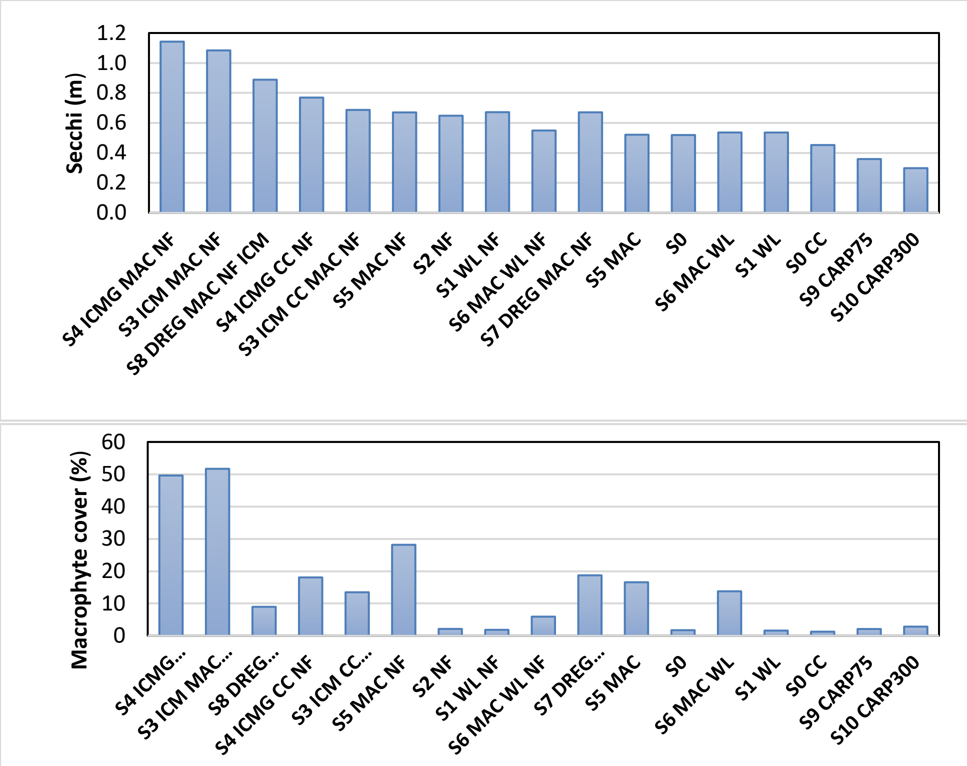


Figure 13. **Scenario simulation results in Lake Waahi including simulation averages of total phosphorus (TP, g m-3), total nitrogen (TN, g m-3), chlorophyll a (Chl a, mg m-3), Secchi depth (m), and macrophyte percentage cover. CC is climate change, WL is water level, NF is no fish (realistically 10 kg ha-1), ICM is integrated catchment management, ICMG is integrated catchment management gold.**

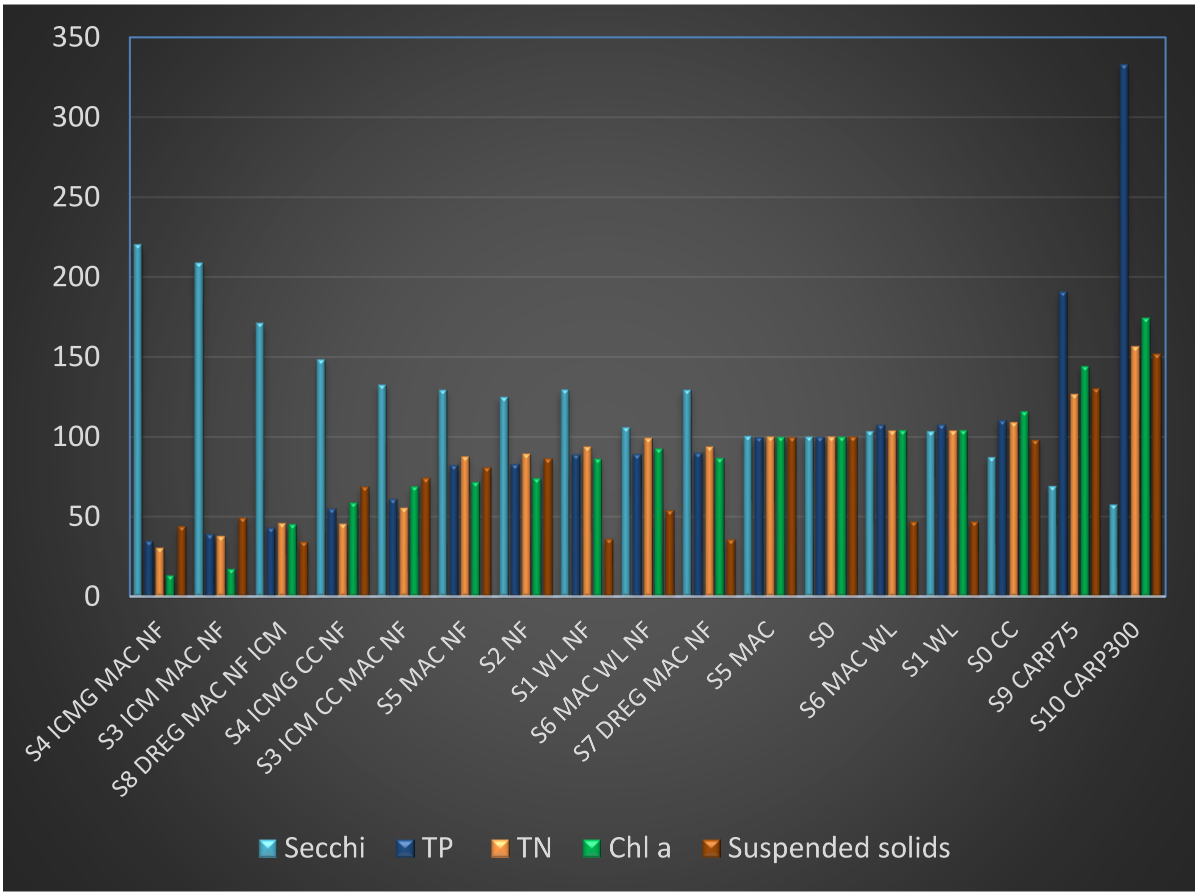


Figure 14. Percentage change from calibration (S0) of scenario simulation results in Lake Waahi. CC is climate change, WL is water level, NF is no fish (realistically 10 kg ha^{-1}), ICM is integrated catchment management, ICMG is integrated catchment management-Gold, MAC is macrophyte establishment, and DREG is dredging (see Table 3).

In terms of national objectives framework (NOF) attribute state, the only water quality restoration scenario that changed NOF attributes were those that included external load nutrient reductions (Table 5). Scenario four, including integrated catchment management, macrophyte re-establishment and carp control, changed TP attribute state from C to B, and TN attribute state from D to B. Scenario three simulated integrated catchment management, macrophyte re-establishment and carp control changing NOF attribute state from D to C, however, this change was reversed when climate change was included in simulation.

When comparing median simulated water colour between scenarios, simulation median water colour changes are hard to discern using visual interpretation (Table 5), apart from those scenarios that combined ICM with carp control. Water colour shifts from a light green colour to a dark green colour represents chl *a* concentration decrease, and species dominance shifts from cyanobacteria to diatom and green algae.

Table 5. National objective framework categories for Lake Waahi simulations, colour coded, alongside median simulated water colour in the last two years of the simulation. D attribute is orange, C yellow, B green, and A blue. Simulations with higher concentrations of cyanobacteria are generally brighter green. Low concentrations of phytoplankton are darker, as the influence of absorption by coloured dissolved organic matter and scattering by suspended sediment are greater relative to phytoplankton.

NOF	S0 _Current conditions	s0_CC	s1_WL	s1_WL_NF	s2_NF	s3_ICM	s3_ICM_MAC_NF	s3_ICM_MAC_NF_CC	s3_ICM_NF	s4_ICMG_MAC_NF	s4_ICMG_CC_MAC_NF	s5_MAC	s5_MAC_NF	s6_WL_MAC	s6_WL_MAC_NF	s7_DREG_MAC_NF	s8_DREG_MAC_NF_ICM	s9_CARP175	s10_CARP300
TP	C	C	C	C	C	C	C	C	C	B	C	C	C	C	C	C	C	D	D
TN	D	D	D	D	D	D	C	D	C	B	C	D	D	D	D	D	C	D	D
CHLA	D	D	D	D	D	D	C	D	C	C	D	D	D	D	D	D	D	D	D
MCHLA	D	D	D	D	D	D	D	C	C	D	C	D	D	D	D	D	C	D	D
NH4	B	B	B	B	B	A	A	A	A	A	A	B	B	B	B	B	A	B	B
Max NH4	B	B	B	B	B	A	B	A	B	A	A	B	B	B	B	B	A	B	B
95 NO3	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A
NO3	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A	A
MEDIAN WATER COLOUR																			

Model communication

In order to avoid the common pitfalls of scientific presentations to non-technical audiences, model results were communicated using simplistic diagrams conveying catchment and lake processes, ecological responses and water colour (Fig. 15a). Within the current calibration the diagram displayed the de-vegetated, cyanobacteria-dominated state of the lake, with sRGB colour representation being the average simulated lake water colour calculated using the bio-optical model. Water colour was also plotted using a colour contour plot of simulated sRGB colours (Fig. 15b). The simulated water colour appears bright green during periods of high cyanobacteria concentration, and yellow-brown when high TSS concentrations are also present. Diatom and green algae at high concentrations generate a darker green colour.

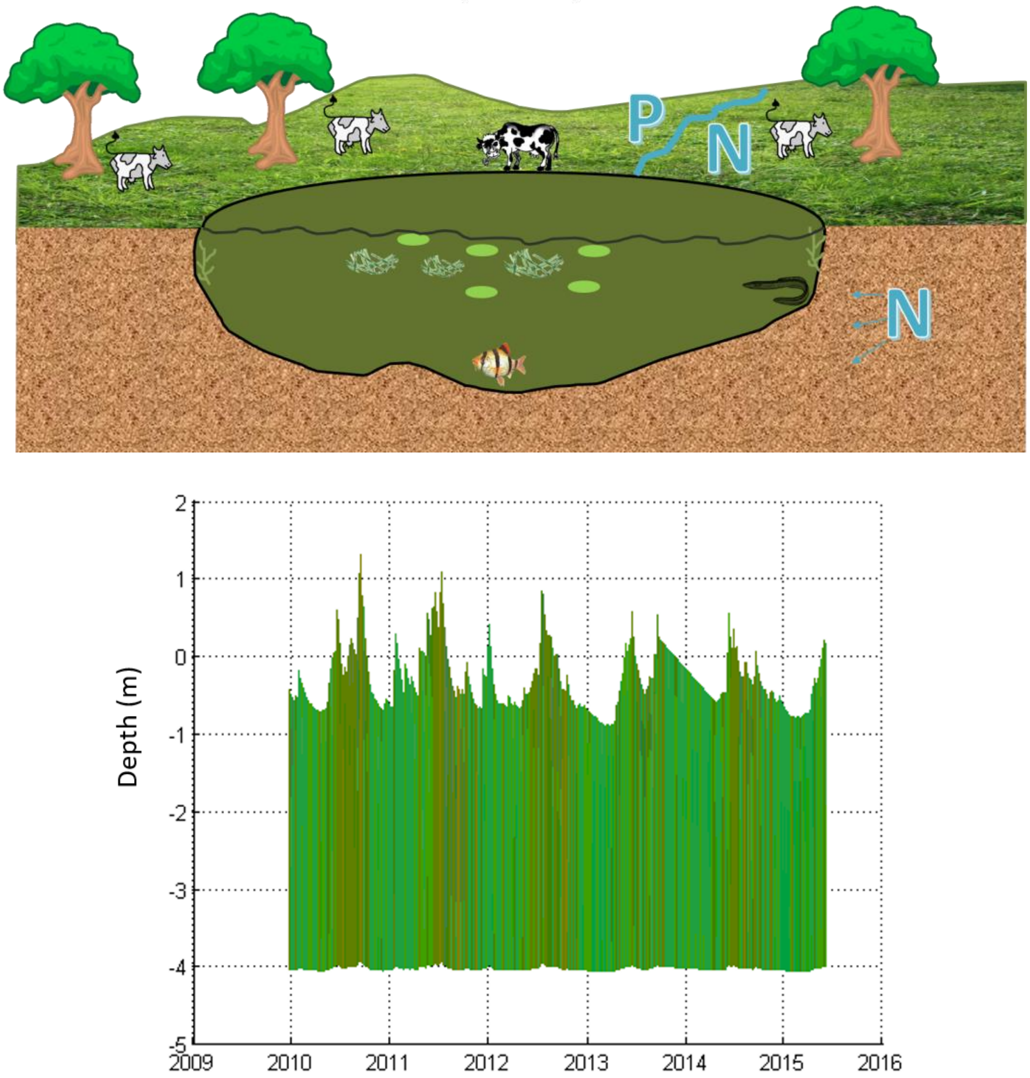


Figure 15. An illustration of average water colour in Lake Waahi within the current calibration. Pictorial representation of the current degraded state is shown above, being dominated by cyanobacteria, de-vegetation, and the associated impact on tuna (eel) populations. Below, water colour is simulated throughout the water column and over the model simulation period.

As can be seen under scenario S4 ICMG MACRO CARP (Fig. 16) phytoplankton dominance shifts from cyanobacteria to diatom dominance causing a shift in water colour from bright green to darker green, which is caused by the change in scattering and absorption properties of phytoplankton.

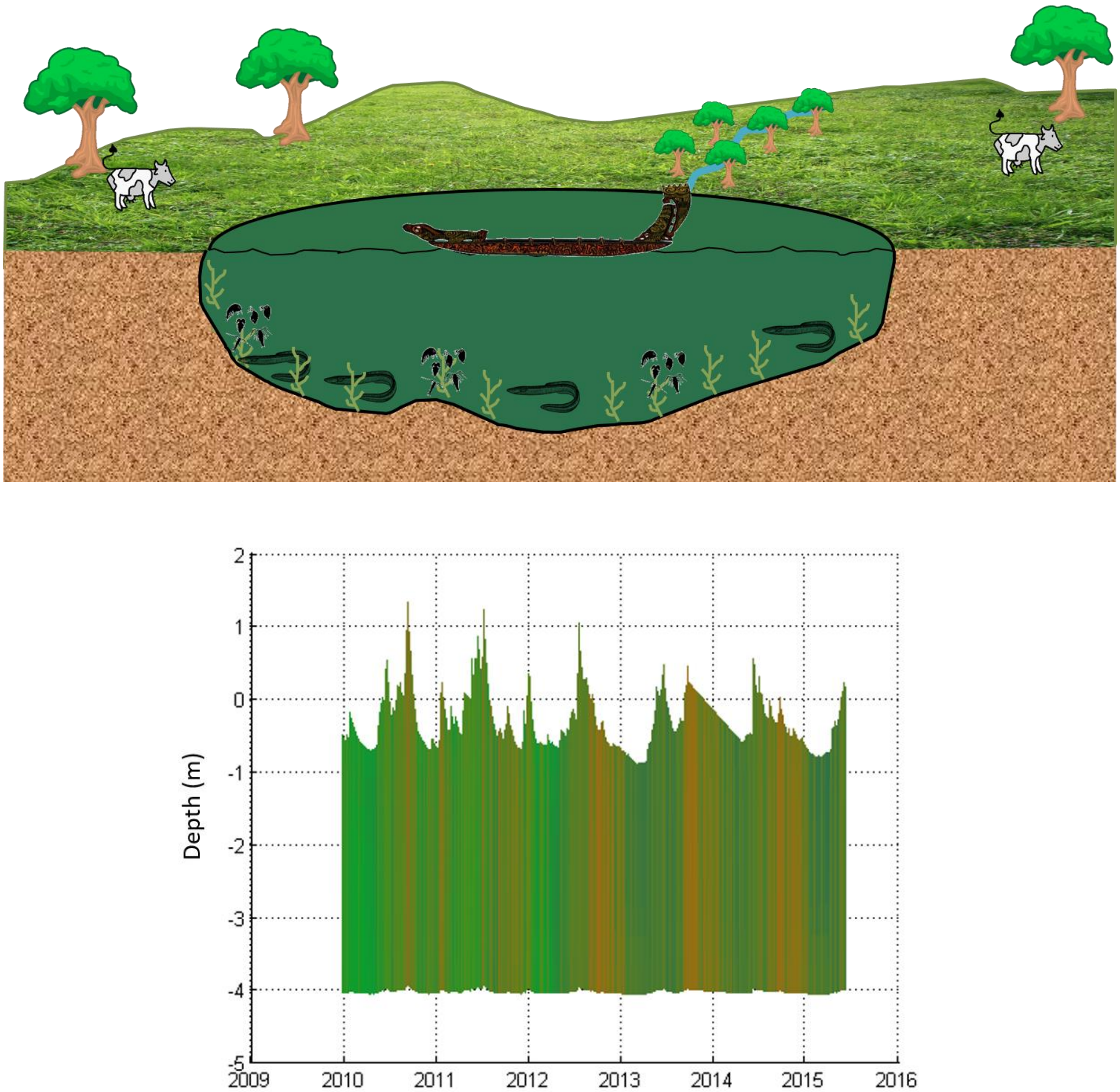


Figure 16. An illustration of average water colour in Lake Waahi with Scenario 4 including pictorial representation of the potential re-establishment of macrophytes, and the associated impact on tuna (eel) (above). The scenario included carp population control and a 50% reduction of external nutrient loading. Under this scenario the lake flips back to a macrophyte-limited, alternate stable state. Below, water colour is simulated throughout the water column and over the model simulation period.

Cyanobacteria contain gas vacuoles, and scatter more light than diatoms, therefore producing a brighter colour. During periods of low chlorophyll *a* concentration within the last two years of the simulation, water colour has either a brown colour (high suspended sediment) or a blue-green colour where low concentrations of all constituents occur.

Conclusions

The consensus from the model simulations carried out in the study suggests that, for Lake Waahi to undergo restoration whereby the lake is restored to a clear-water stable state, both external load (sediment and nutrients) needs to be reduced significantly by greater than 50%, along with changes in lake ecological structure, resulting from re-establishment of macrophytes aided by the control of carp biomass. These findings support the general view within the literature that for lake restoration success external load reduction (catchment load) is critical (Søndergaard et al. 2007). The remote sensing of total suspended sediments and chlorophyll *a* provided additional calibration and validation data within the study. However, to make use of the spatial nature of estimates, future modelling may include 3-D ecologically-coupled, hydrodynamic models.

Climate change effects must be considered in any restoration scenario (Ministry for the Environment 2015). The simplistic way in which climate change is simulated within the scenarios in this study can lead to a potential overestimation of algal biomass. The increase in average air temperature by 3° leads to more stratification and more algal blooms. However, potential climate impacts also include larger storm systems, disrupting stratification, potentially mitigating some of the predicted temperature effects. Changes in lake stratification and thermal regimes can affect complex ecological interactions, with unexpected potential consequences (Williamson et al. 2008).However, the consensus from the literature points to the fact that climate change will enhance eutrophication and harmful algal bloom occurrence (Williamson et al. 2008; Trolle et al. 2011)

Currently no sensitivity analysis has been completed, and this is a limitation of the study. PCLake is undergoing ongoing development part of which includes an automated auto calibration and sensitivity analysis routine. Model uncertainty will need to be defined quantitatively for lake models to be used in decision-making processes (Wallace 2017).

The simulation of water colour, associated with simulated concentrations of chlorophyll *a*, suspended sediment and coloured dissolved organic matter using the 1D model, could provide communities with an interpretable framework for understanding lake science related to water quality modelling and scenario generation. The ability to visualize lake restoration scenarios in terms of colour also has the potential to aid in lake management by enhancing the ability to engage a range of audiences and contribute to robust reporting (McInerny et al. 2014; Gough et al. 2014).

The simulated water quality under the current calibration results in a NOF D grade for the variables TN, TP, and chlorophyll *a*. Under the NOF, environmental policy regarding freshwater management needs to address this failure, especially in the context of tangata whenua values. The involvement of iwi and hapu within freshwater management are key to meeting obligations under the Treaty of Waitangi (NPSFM 2014). The collaborative approach to restoration scenario generation used in this study, through consultation with the TAG and Waahi Whanui, will need to be expanded and continued in order to make effective lake management plans to enable lake restoration in line with Waikato-Tainui aspirational futures for lakes and rivers. These restoration scenarios will need to be carefully considered for cost versus benefit (McDowell et al. 2013).

Finally, if the ultimate goal of lake restoration in Lake Waahi is to address the decline in tuna biomass, it’s likely that a return to a clear-water stable state will facilitate improvement in tuna habitat and provide the conditions for development of a healthy eel population. There is evidence that changes in trophic pathways resulting from declining water quality can influence tuna production (Kelly & Jellyman 2007; Noth et al. 2008).

While PCLake is not applicable to Waahi tributaries, the boosted regression tree modelling of Death et al. (2017; see Appendix 2) could be useful to provide guidance on catchment restoration scenarios and their effects on stream ecology in relation to Huanga kai. Coupled modelling approaches will provide a more complete picture of potential responses of Hauanga kai species to land and lake management scenarios.

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Appendices

Appendix 1. PCLake parameters

cDFiJvIn: 0.1	# external fish density (gDW m-2)	default = 0.005
cDFiAdIn: 1.7	# external fish density (gDW m-2)	default = 0.005
cDPiscIn: 1.7	# external Pisc. density (gDW m-2)	default = 0.001
kMigrPisc: 0.1	# piscivorous migration rate (d-1)	default = 0.001
fDBone: 0.35	# fraction of fish C fixed in bones and scales ([-])	default = 0.35
fPBone: 0.5	# fraction of fish P fixed in bones and scales ([-])	default = 0.5
cDCarrFish: 15.0	# carrying capacity of fish (gDW m-2)	default = 15.0
fDissEgesFish: 0.25	# soluble nutrient fraction of fish egested food ([-])	default = 0.25
fDissMortFish: 0.1	# soluble nutrient fraction of dead fish (excl bones and scales) ([-])	default = 0.1
cTmOptFish: 25.0	# optimal temperature of zoo- and benthivorous fish (degree C)	default = 25.0
cSigTmFish: 10.0	# temperature constant of fish (sigma in Gaussian curve) (degree C)	default = 10.0
cDayReprFish: 245.0	# reproduction day of year for fish ([-])	default = 120.0
cDayAgeFish: 365.0	# aging day of year for fish ([-])	default = 365.0
fReprFish: 0.02	# yearly reproduction fraction of benthivorous fish	daily rate ([-]) default = 0.02
fAgeFish: 0.5	# yearly ageing fraction of zooplanktivorous fish	daily rate ([-]) default = 0.5
kDAssFiJv: 0.12	# maximum assimilation rate of zooplanktivorous fish (d-1)	default = 0.12
hDZooFiJv: 1.25	# half-saturation zooplankton for zooplanktivorous fish predation (g m-2)	default = 1.25
fDAssFiJv: 0.4	# C assimilation efficiency of zooplanktivorous fish ([-])	default = 0.4
kDRespFiJv: 0.01	# maintenance respiration constant of zooplanktivorous fish (d-1)	default = 0.01
kMortFiJv: 0.00137	# specific mortality of zooplanktivorous fish (d-1)	default = 0.00137
kDRespFiAd: 0.004	# maintenance respiration constant of benthivorous fish (d-1)	default = 0.004
kMortFiAd: 0.00027	# specific mortality of benthivorous fish (d-1)	default = 0.00027
cDCarrPiscMax: 2	# maximum carrying capacity of piscivorous fish (gDW m-2)	default = 1.2
cDCarrPiscMin: 1.6	# minimum carrying capacity of piscivorous fish (gDW m-2)	default = 0.1
cDCarrPiscBare: 2	# carrying capacity of piscivorous fish (gDW m-2)	default = 0.1
cDPhraMinPisc: 0	# minimum reed biomass for piscivorous fish (gDW m-2)	default = 50.0
cCovVegMin: 0.0	# minimum submerged macrophytes coverage for piscivorous fish (%)	default = 40.0
cRelPhraPisc: 0.075	# relative piscivorous fish density per reed (gDW m-2)	default = 0.075
cRelVegPisc: 0.03	# relative piscivorous fish density per reed if aCovVeg>cCovVegMin (gDW m-2)	default = 0.03
kDAssPisc: 0.025	# maximum assimilation rate for piscivorous fish (d-1)	default = 0.025
hDVegPisc: 5.0	# half-saturation constant for macrophytes on piscivorous fish (g m-2)	default = 5.0
hDFishPisc: 1.0	# half-saturating DFish for piscivorous fish predation (g m-2)	default = 1.0
fDAssPisc: 0.4	# C assimilation efficiency of piscivorous fish ([-])	default = 0.4
fDissEgesPisc: 0.25	# soluble P fraction of fish egested food ([-])	default = 0.25
kDRespPisc: 0.005	# respiration constant of piscivorous fish (d-1)	default = 0.005
kMortPisc: 0.00027	# specific mortality of piscivorous fish (d-1)	default = 0.00027

fDissMortPisc: 0.1	# soluble nutrient fraction of dead piscivorous fish ([-])	default = 0.1
cTmOptPisc: 25.0	# optimal temperature for piscivorous fish (degree C)	default = 25.0
cSigTmPisc: 10.0	# temperature constant for piscivorous fish (sigma) (degree C)	default = 10.0
cPDFishRef: 0.0259	# reference P/C ratio of fish (mgP/mgDW)	default = 0.022
cNDFishRef: 0.148	# reference N/C ratio of fish (mgN/mgDW)	default = 0.1
cPDPisc: 0.0878	# reference P/C ratio of piscivorous fish (mgP/mgDW)	default = 0.022
cNDPisc: 0.268	# reference N/C ratio of piscivorous fish (mgN/mgDW)	default = 0.1
Manipulate_FiAd: false	# turn on/off benthivorous fish manipulation	default = false
Manipulate_FiJv: false	# turn on/off zooplanktivorous manipulation	default = false
Manipulate_Pisc: false	# turn on/off piscivorous fish manipulation	default = false
cDFiJvMin: 0.0001	# minimum zooplanktivorous fish biomass in system (gDW m-2)	default = 0.0001
cDFiAdMin: 0.1	# minimum benthivorous fish biomass in system (gDW m-2)	default = 0.0001
cDPiscMin: 1.7	# minimum piscivorous fish biomass in system (gDW m-2)	default = 0.0001
fFisDOMW: 0.8	# dissolved organic matter fraction from fish ([-])	default = 0.5
fDAssFiAd: 0.4	# C assimilation efficiency of adult fish ([-])	default = 0.4
cRelVegFish: 0.009	# decrease of fish feeding per macrophytes cover (max. 0.01) ([-])	default = 0.009
kDAssFiAd: 0.06	# maximum assimilation rate of adult fish (d-1)	default = 0.06
hDBentFiAd: 2.5	# half-saturation constant for zoobenthos on adult fish (g m-2)	default = 2.5

Appendix 2. Boosted regression tree modelling of koura and tuna density (from Professor Russell Death, Massey University)

Biological data
Biological data was supplied from Waikato Regional Council (WRC) environmental monitoring. Mussels of two species *Echyridella aucklandica* and *Echyridella menziesii* were surveyed by visual inspection at 104 sites between 2010 and 2014 in the western Waikato region. *Paranephrops planifrons* (koura) was sampled 256 times at 127 locations between 2010 and 2016 using electro-fishing (Joy et al. 2013). Two species of eel, the longfin and shortfin, were sampled by electro-fishing on 299 occasions at 162 sites using the protocols of (Joy et al. 2013).

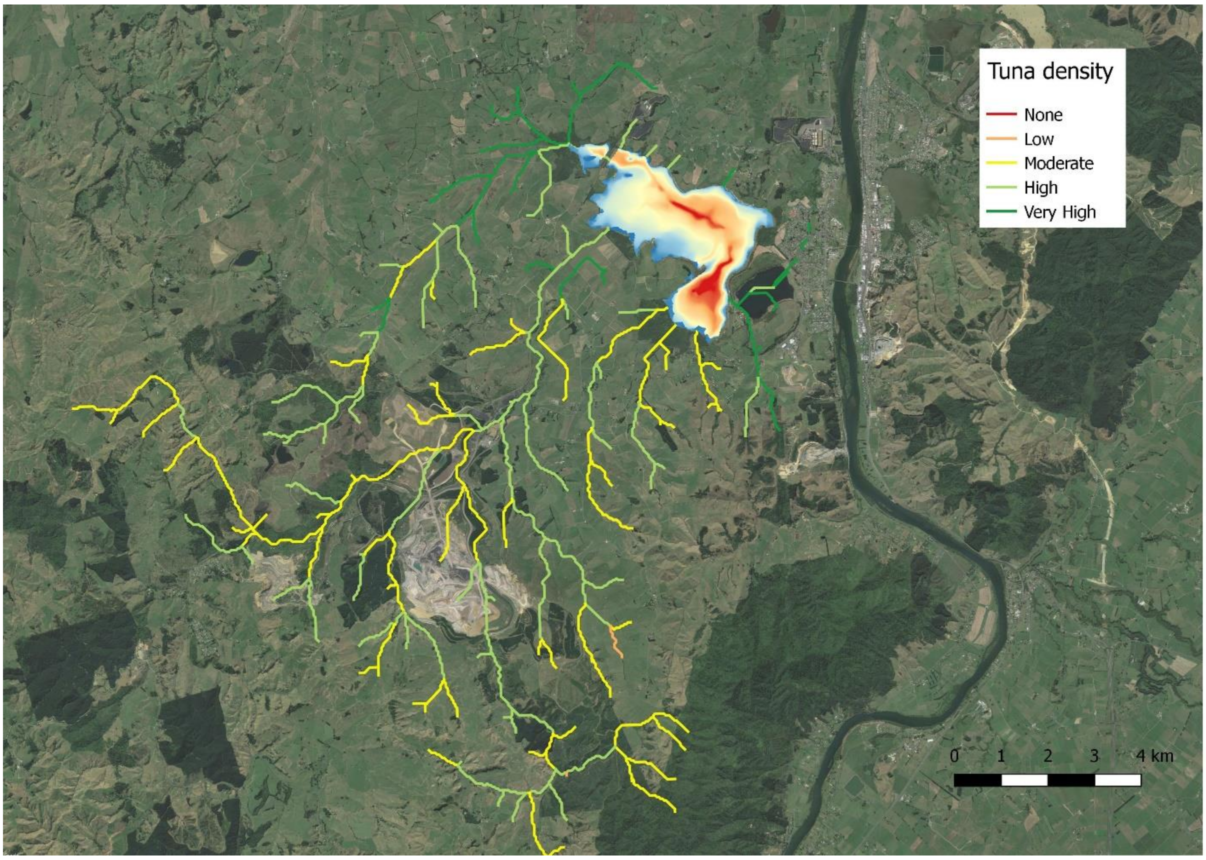
Field data
Concomitant with biological sampling, standard REMS field assessment for site canopy cover, fencing, dissolved oxygen, conductivity etc was recorded (Collier and Kelly 2005). Water samples were collected for ammonia/metal/hardness analysis. At the end of the sampled reach WRC staff measured the wetted width, thalweg depth (maximum depth at that transect), percent macrophyte cover, estimated the percent substrate (clay, silt, sand, small gravel etc. for the 10m reach), percent shade and percent wood. Finally, the soft bottomed QHA (habitat assessment) form was filled out.

GIS data
GIS environmental data on nutrients, flow regime, catchment geology and topography, temperature and shading. MCI and deposited sediment for each sampled reach were included in the model construction as potential predictor variables. Modelled nutrient, MCI and *E. coli* were sourced from (Unwin and Larned 2013), and flow data from (Booker and Woods 2014). Catchment geology, topography and temperature were retrieved from the Freshwater Ecosystems of New Zealand (FENZ) database (Leathwick et al. 2010), and modelled sediment data from (Clapcott et al. 2103).

Data analysis
There are many machine learning techniques that can be used for modelling of ecological data, however, each offers its own unique advantages and disadvantages (Death 2015; Kuhn and Johnson 2016). Boosted regression trees (BRTs) are a powerful modification of classification and regression tree analysis that is now widely used in ecology (De'ath and Fabricius 2000; Elith et al. 2008). It combines the algorithms of regression trees, that relate predictors of a single response variable by recursive binary splits, and boosting, an ensemble method that combines multiple simple models to increase predictive performance (Elith et al. 2008). One disadvantage of the approach is that models are complex and cannot be represented by a single decision tree that might allow a researcher to explain the model relationships to a potential end-user. However, the advantage of the technique is it can provide robust models of predictor and response variable relationships from a limited dataset and allow extrapolation from that model to new data scenarios.

The *gbm* package (Ridgeway 2013) was used in R (version 3.3.1; R Project for Statistical Computing, Vienna, Austria) to fit the BRTs using the Gaussian family of relationships as data were quantitative (number / m^2). Tree complexity was 8, learning rate was 0.001, bag fraction was 0.5, and cross-validation was run on 10% of the data.

Predictions are presented in Figures A2.1 and A2.2.



FigureA2.1. Boosted regression tree estimation of relative tuna density in Lake Waahi tributaries.

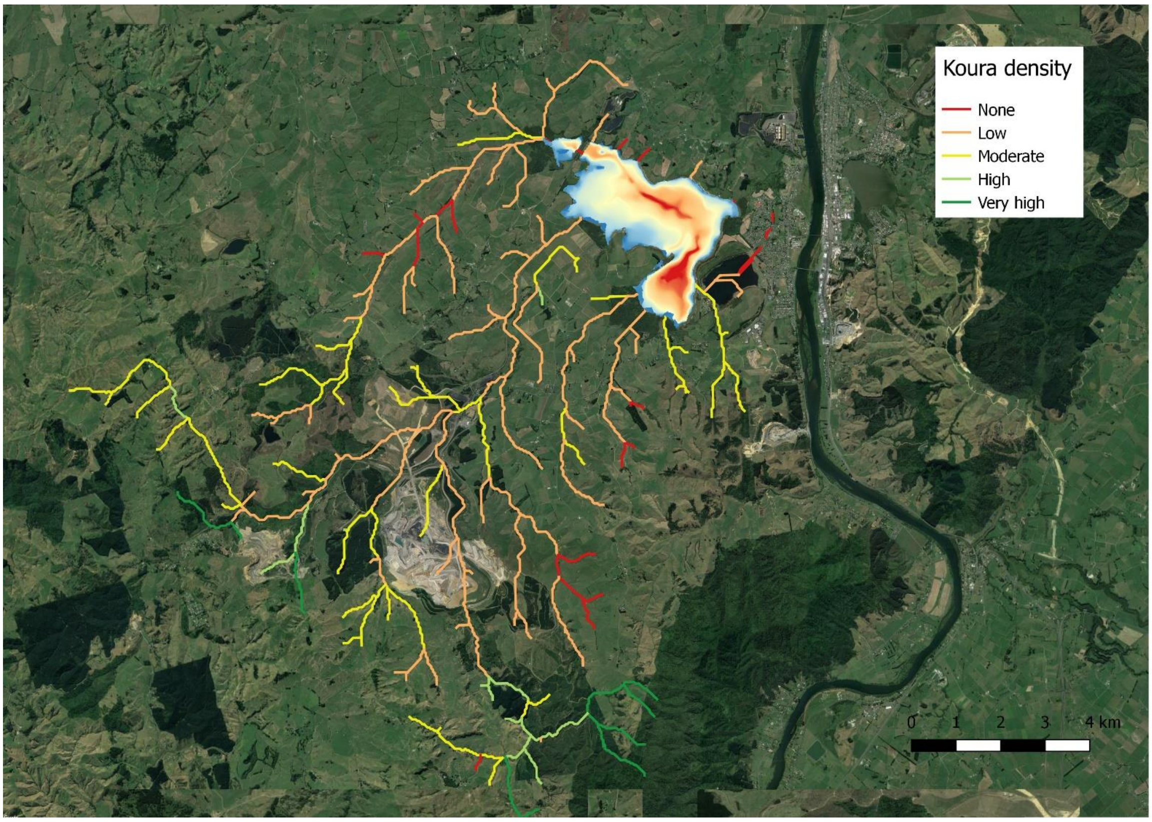


Figure A2.2. Boosted regression tree estimation of relative Koura density in Lake Waahi tributaries.