Lake Benmore Water Quality: a modelling method to assist with assessments of nutrient loadings.

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Prepared for Environment Canterbury by

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Environment Canterbury
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Executive Summary

Background

The lakes of the Upper Waitaki Basin are a local, regional and national asset for their ecological, recreational, commercial and tourism values. Historically most land-use in the catchment has been low-intensity, dryland sheep farming, with little irrigated land. However in recent years there has been a shift to irrigation, dairying and dairy support. Further irrigation development is proposed and Environment Canterbury (ECan) is currently considering 35 applications for resource consent for irrigation in the Upper Waitaki Basin.

A key concern when land use is intensified is potential impacts on water quality. ECan is responsible for the management of water quality in the Waitaki lakes as part of its functions under the Resource Management Act (RMA). ECan commissioned this study to assist with those functions. The broad goal of this work was to increase understanding of the links between catchment land use changes, associated nutrient loads to the lakes, in-lake processes, lake water quality and associated values.

Purpose

The purpose of this report is to provide technical information and options for ECan's regional plan provisions. This includes options for:

(i) Measurable objectives for water quality in Lake Benmore; and

(ii) Nutrient load caps associated with achieving each of the objective options.

Lake Benmore is the critical downstream receiving basin for nutrients from both the Ahuriri (West) Arm and Haldon (North) Arm catchments and is therefore likely to be the first of the Waitaki Lakes to show water quality effects. Lake Benmore is also likely to be more vulnerable to adverse effects from increased nutrient loads than downstream Lakes Aviemore and Waitaki, due primarily to the longer water residence time in Lake Benmore. Therefore, setting objectives and nutrient load caps for Lake Benmore could establish the "line in the sand" basis for effectively managing the cumulative effects of multiple activities in the Upper Waitaki catchment on water quality for the chain of lakes. By protecting water quality in Lake Benmore, water quality of Lakes Aviemore and Waitaki is also likely to be protected.

Modelling approach

A coupled hydrodynamic-ecosystem modelling approach was used to predict the relationship between nutrient load (total nitrogen [TN] and total phosphorus [TP]) and measures of lake condition such as the Trophic Level Index (TLI). The TLI provides a convenient and pragmatic numeric scale for measuring trophic status of New Zealand lakes. Choices for the desired trophic state of a lake can be equated with a numeric value on the TLI scale and thus TLI can be used for setting measurable regional plan objectives for lakes.

Several hypothetical future scenarios that involved increasing nutrient loads to Lake Benmore were modelled. The relationships between these modelled nutrient loads (TN and TP) and TLI values were presented graphically. In addition to this, relationships between individual components of the TLI, dissolved oxygen and modelled nutrient loads were also presented graphically.

Conclusions

- The model strongly indicates that :

    I. The Ahuriri and Haldon Arms of Lake Benmore behave differently in response to increased nutrient loads.

    II. The Ahuriri Arm is more sensitive to nutrient increases than the Haldon Arm, due primarily to the relatively smaller inflow and relatively longer residence time in the Ahuriri Arm

- We recommend that the two Arms be considered separately in terms of setting management objectives and nutrient load caps.

- The model predicted that water quality in the Lower Benmore basin was strongly influenced by the Ahuriri Arm. By managing the Ahuriri Arm, the Lower Benmore basin is likely to be managed by default. However, if a decision is made that involves setting a higher load cap for the Haldon Arm than for the Ahuriri Arm, the model could be rerun with the specified loads to check that flow interactions between the two Arms do not alter intended trophic level outcomes.

- Options for TLI objectives, descriptions of the related environmental states and associated TN and TP load caps for both the Ahuriri and Haldon Arm are described in detail in Section 6 of this report. A summary is provided in Table ES-1 in terms of ranges in TLI. We note that the TLI is a continuous scale and so any decimal number could be chosen (e.g., 2.9, 3, 3.1, 3.25, etc.) and the associated TN and TP caps calculated for that TLI value. A selection of discrete TLI options is provided in Table 3, Section 6. Generally, the higher the TLI number the greater the risk of associated degradation of environmental values.

- It is not the purpose of this report to recommend any particular TLI objective because that decision is ECan's responsibility and would be developed in the context of the RMA planning provisions. However, we have discussed several issues that should be considered in Section 6 of this report.

Table ES-1. Summary of the modelled summer TLI ranges with associated nutrient load caps for total nitrogen (TN, tonnes N/year) and total phosphorus (TP, tonnes P/year) according to the model output for the Haldon and Ahuriri Arms. For purposes of specifying load caps in the table, the maximum load for a given trophic state corresponds to 0.1 TLI level below the boundary specified by Burns et al. (2000) for classification of trophic states (see Table 1, Section 4.4 in this report), or the maximum value modelled in this study. The minimum TLI shown for each range is the smaller of the trophic level boundary of Burns et al. (2000), or the lowest TLI modelled in this study; the baseline (existing) TLI values and loads are shown in blue.

		Haldon Arm			Ahuriri Arm	
		nutrient loads** (tonnes/yr)			nutrient loads** (tonnes/yr)	
Trophic State	**TLI***	**TN**	**TP**	**TLI ***	**TN**	**TP**
Oligotrophic	2.1 – 2.9	646 - 1140	68 - 125	2.4 – 2.9	173 - 256	24 - 35
Mesotrophic	3 – 3.6	1220 - 1860	134 - 209	3 – 3.9	272 -514	38 - 71
Eutrophic	***	***	***	4 – 4.9	551 - 1070	76 - 148
Supertrophic	***	***	***	5 – 5.6	1180 - 2080	162 - 287

Notes:

* - The TLI is a continuous scale; any decimal number could be chosen (e.g., 2.9, 3, 3.1, 3.25, etc.) and the associated TN and TP caps calculated for that TLI value. A selection of discrete TLI options is provided in Table 3, Section 6.

** - Total loads excluding aerial deposition; includes Ahuriri River + ungauged flows for the Ahuriri Arm (Table E.12, Appendix E), and Ohau C Canal + Tekapo-Pukaki Rivers (including spill from Lakes Tekapo and Pukaki) + ungauged flows for the Haldon Arm (Table E.8, Appendix E).

*** - Trophic state beyond model range.

1 Introduction

1.1 Background

The lakes of the Upper Waitaki Basin are a local, regional and national asset for their ecological, recreational, commercial and tourism values. The waters of the lakes are currently classified as “microtrophic” to “oligotrophic” (Meredith & Wilks, 2006), scientific terms meaning that the lakes have low biological productivity, and have relatively high water quality, i.e., low nutrients, high clarity and limited phytoplankton productivity (Burns et al. 2000, Davies-Colley et al. 1993). It is these characteristics that give the lakes their high aesthetic value and colour, and attractiveness.

Historically most land-use in the catchment has been low-intensity, dryland sheep farming, with little irrigated land. However in recent years there has been a shift to irrigation and associated more intensive land uses such as dairying and dairy support. There has been some land use change already as can be seen in satellite images (see Figure 1). Environment Canterbury (ECan) is also currently considering 60 water permit applications for irrigation in the Upper Waitaki Basin.

Water quality is a key concern when land use is intensified because there is a need to manage increased contaminant loads entering waterways (e.g. nutrients, sediment and microorganisms) in order to maintain the environmental values of the downstream lakes. The focus of this report is on potential increases in nutrient loads entering the lakes as this could lead to degradation of clarity, colour and biological processes, as well as ecological, recreational and commercial values.

ECan is responsible for the management of water quality in the Waitaki lakes as part of its functions under the Resource Management Act (RMA). ECan commissioned this study to assist with those functions. The broad goal of this work was to increase understanding of the links between catchment land use changes, associated nutrient loads to the lakes, in-lake processes, lake water quality and associated values. The specific purpose of this study is to provide options for ECan’s regional plan provisions.

1.2 Previous assessments

Reports commissioned in 2005 and 2008 used a relatively simple, regression-based modelling approach to predict that moderate land-use intensification could result in algal blooms (i.e., intense phytoplankton accumulations and/or increased periphyton (benthic algae) accumulations) in the Upper Waitaki Basin’s rivers and lakes, with negative consequences for ecological, recreational and commercial values (GPF (White) 2005; Snelder et al. 2005). In short, these studies concluded that the rivers could be increasingly affected by filamentous algae accumulations and the values of the lakes could be altered as a consequence of increased nutrient loads.

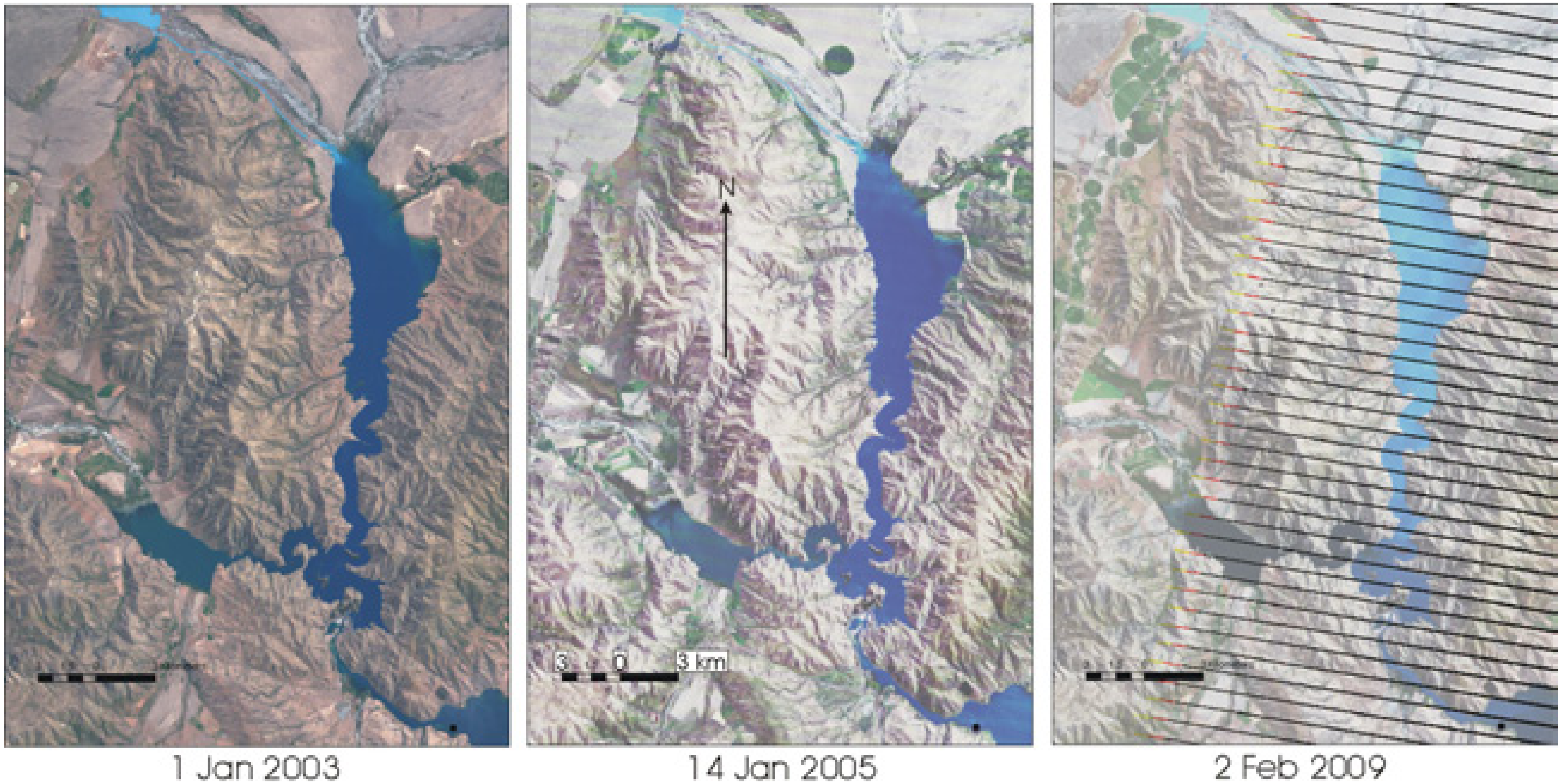

Figure 1: NASA images showing increased irrigated land (green areas) over time; note the progressive development in the northwest corner. Dates are NZST, 10:00. These are unprocessed images, no atmospheric or colour corrections have been made and differences in lake colour should not be interpreted as real. All images are from NASA Landsat Program and provided by US Geological Survey, Sioux Falls, South Dakota, USA; from left to right: Landsat ETM+ scene LE70750912002365EDC00, Landsat TM scene LT50750912005013HOA00, Landsat ETM+ scene L71075091_09120090201. Images were located and downloaded by Mathew Allan, University of Waikato.

Such changes would alter the natural character of the lakes, degrade aesthetics, reduce the quality of habitat for fish and other aquatic life, negatively affect recreational opportunities such as angling, swimming and boating, and have a number of negative impacts on hydro-generation operations. The earlier reports were of a scoping nature and acknowledged the need for more comprehensive information and considerations, requiring further study

1.3 Lessons from the central North Island experience

The impacts of increased nutrient loadings from intensive land-use have become apparent in a number of lakes in the North Island. Lake Taupo is an example of one such lake, where water quality is affected by land use intensification within the catchment. Hydrodynamic-ecosystem models were used to show that under current nutrient loadings, lake quality would continue to decline affecting lake ecosystem functions and resulting in a deterioration of lake values for some time into the future (Spigel et al. 2003). Environment Waikato (EW) responded to this situation by identifying objectives for Lake Taupo water quality (i.e., in terms of water clarity [Secchi disk depth], nitrogen, phosphorus and chlorophyll *a* (Chl *a*) concentrations) and by quantifying the nutrient load cap necessary to achieve those objectives. The water quality objectives, nutrient limits and associated land-use rules have been incorporated into the Waikato Region Plan Variation 5 – Lake Taupo Catchment (known colloquially as "RPV5"). The RPV5 provides a quantitative planning mechanism to regulate the contribution of nutrients from land-use activities in the Taupo catchment (http://www.ew.govt.nz/Policy-and-plans/Protecting-Lake-Taupo/).

Other North Island lakes affected by intensified land use include Lakes Rotorua, Rotoiti, Rotoehu, Okaro and Okareka which are all in the jurisdiction of Environment Bay of Plenty (EBOP). EBOP has developed measurable objectives for these lakes that are expressed in terms of a numeric lake Trophic Level Index (TLI) for each lake (TLI is a function of nitrogen and phosphorus concentrations, Secchi depth [clarity] and chlorophyll *a* concentration; Burns et al. 2000). EBOP has incorporated these objectives, along with several rules to manage nutrient loads from land-use within defined loading caps, into their Proposed Regional Water and Land Plan. The rules, collectively known as "Rule 11" effectively draw a "line in the sand" to cap existing nitrogen and phosphorus loss from land use activities in lakes where a predefined TLI value has been attained (http://www.envbop.govt.nz/Water/Lakes/Rule-11.asp).

An important aspect of both RPV5 and Rule 11 is that they are designed to manage a situation where lake water quality shows early signs of, or has already, declined. Both these planning instruments will restrict further land use development and it is possible that some areas of existing intensive land use will ultimately need to be retired in order to meet water quality objectives in some of these lakes. In other words Rule 11 and RPV5 are playing "catch-up" for lakes whose capacity to assimilate nutrients without further deterioration in water quality, has already been exceeded.

A lesson to be learned from the central North Island experience is that it would be more certain for environmental outcomes, fairer, less time-consuming and more cost effective, if appropriate water quality objectives and related nutrient load limits were established *before* the assimilative capacity of a lake (or a river system) is exceeded. This would make the ground rules for land developers clear before they make investment decisions. Measurable plan objectives and nutrient load caps would clearly quantify the sustainable capacity of the lakes in terms of catchment landuse.

1.4 Additional management considerations

Lakes tend to respond to nutrient increases in a non-linear manner. As nutrient loads increase, small changes may be observed initially in a lake, but at some tipping point large changes in trophic status can occur rapidly (Carpenter 2000). However trophic state may not revert back at the same rate when nutrient loads are reduced. A time lag of 10-15 years is commonly observed before lakes respond positively to substantial reductions in external loading (Jeppesen et al. 2005). This means that it is generally more effective to manage lake trophic state before nutrient thresholds are reached than to try and restore degraded lakes.

The modelling described in this report is based on the response of the lake to nutrient loads as they enter the lake at present; the existing response has been used to calibrate the model and serve as the baseline on which the scenarios for increased nutrient loads have been built. As pointed out in Section 1.1 with reference to Figure 1, land use changes have already occurred in the Lake Benmore catchment. It has not been possible in this study to account for time lags that occur between changes in catchment land use and changes in nutrient loads at the lake boundary, either in surface or groundwater inflows. These lags will need to be considered when making decisions regarding nutrient caps for proposed future land use changes (see also discussion at the end of Section 2.1).

Global climate change will impact a variety of drivers of algal proliferation. Of these, perhaps the least contentious is the effect of climate warming, which may encourage algal blooms in many lakes and may disproportionately favour nuisance (potentially toxic) cyanobacterial blooms (Paerl and Huisman 2008). Thus, climate change may exacerbate the adverse effects of increased nutrient loads from intensified landuse in future.

1.5 Possible implications for regional planning

Farm-scale models are now available to estimate the quantity of nutrients lost from land under specified landuses. Farm-scale models can be used to assist with allocating a catchment-based sustainable nutrient load cap amongst farm owners, provided that loads from other sources are also accounted for (e.g., natural background loads and point sources such as wastewater discharges). Once the full allocation has been made it would be clear that the only way to intensify existing land use would be to “free-up” some nutrient credit by employing nutrient reduction measures on some other existing land in the catchment (e.g., reduced fertiliser and/or stocking rates, riparian buffer strips, wetlands etc).

2 Purpose and overview

2.1 Purpose

The purpose of this report is to provide technical information and options for ECan to consider for:

(i) Measurable objectives for water quality in Lake Benmore; and

(ii) Nutrient load caps associated with achieving each of the objective options.

Because Lake Benmore is the critical downstream receiving basin for nutrients from both the Ahuriri Arm and Haldon Arm catchments, it is likely to be the first of the Waitaki Lakes to experience water quality changes. Lake Benmore is also likely to be more vulnerable to adverse effects from increased nutrient loads than downstream Lakes Aviemore and Waitaki. This is due primarily to the longer water residence time in Lake Benmore (57 days for Haldon Arm, 75 days for Ahuriri Arm) compared to Lake Aviemore (14 days) and Waitaki (1.1 days) (residence times from Snelder et al. 2005, Table 5, based on basin volumes and flows supplied by Meridian Energy Limited). Therefore, setting objectives and nutrient load caps for Lake Benmore could establish the “line in the sand” basis for effectively managing the cumulative effects of multiple activities in the Upper Waitaki catchment on water quality for the chain of lakes downstream of, and including, Lake Benmore, i.e., by protecting water quality in the lower Lake Benmore basin, water quality of Lakes Aviemore and Waitaki is also likely to be protected.

Establishing clear, measurable objectives and related water contaminant limits is consistent with the general approach described by Norton and Snelder (2003). This approach would also be consistent with Objective 3 of the Water section of the Canterbury Regional Policy Statement and the associated policies. The Proposed Natural Resources Regional Plan (PNRRP), for which the public hearing part of the plan development process finished in June 2009, includes specific lake water quality objectives but does not specify lake trophic indices or set nutrient load caps for lakes. PNRRP Objective WQL1.2 (3) is relevant for Lake Benmore and states:

“For artificial lakes, the water quality of the lake shall be maintained so that:

(a) it is suitable for the activities and uses for which the lake and its water is used; and

(b) it does not result in persistent seasonal stratification leading to oxygen depletion in the lake; and

(c) it does not result in toxic or nuisance algal blooms; and

(d) the average annual phytoplankton biomass does not exceed five milligrams of chlorophyll a *per cubic metre of lake water.”*

Specific lake trophic index objectives and nutrient load caps could be implemented via the regional plan change/variation process and the options for Lake Benmore outlined in this report would be available as a reference point for ECan and the wider community to consider.

It is important to note that this report provides options for objectives and nutrient load caps that would, if implemented through a regional plan, define the capacity of Lake Benmore to receive nutrients while still meeting a defined environmental objective. One way of thinking about this in terms of resource management and allocation, is that the nutrient load caps identified in this report (expressed in tonnes of nitrogen and phosphorus per year) describe the ‘total size of the pie’ to be allocated. Once the total size of the pie is defined, it is then necessary to determine how much of the pie is already allocated to natural background nutrient loads and loads from existing activities such as point and diffuse discharges. The piece of pie left over may then be considered for allocation to new proposed activities. The purpose of this report is only to identify the total size of the pie and the measured loads of nitrogen and phosphorus currently entering the lake. This report does not attempt to quantify the nutrient loads coming from various existing activities and does not predict nutrient loads associated with specific future proposed activities.

2.2 Aims

In order to achieve the above purpose this study has two aims

Aim 1 – Determine options for objectives

The TLI provides a convenient and pragmatic numeric scale describing lake condition. Choices for the desired environmental state of the lake (i.e. the level of attainment of the purposes for management defined above) can be equated with a numeric value on the TLI scale. EBOP established a precedent for using TLI in regional plan objectives for lakes in a Regional Water and Land Plan that became effective on 1 December 2008. At recent PNRRP hearings ECan officers (Hayward et al. 2009) proposed a TLI value of 3 as a regional planning objective for all Canterbury lakes in the “Artificial Lakes” management unit and this includes Lake Benmore. The first aim of this study is to provide a range of options around this TLI value and to describe the consequences of these options, specifically for Lake Benmore.

Aim 2 – Determine nutrient load caps associated with achieving objective options

A coupled hydrodynamic-ecosystem modelling approach has been used to determine how the components of the TLI and the TLI, itself, as well as some other indicators of lake condition, such as dissolved oxygen, would change in response to a range of future scenarios that involve increasing nutrient loads (i.e. nitrogen and phosphorus) entering the lake. This means describing the *relationship* between nutrient load and TLI. The second aim of this study involved presenting these technical relationships in graphic form so that ECan could select any particular TLI value as an objective and obtain the nutrient load cap needed to achieve the objective that was forecast. In other words, for any value of TLI the related nitrogen and phosphorus load caps could be read from graphs.

This method could be applied to any management objective in addition to the TLI and/or could also be applied to individual components of the TLI using the data generated in this study. However this study has adopted the TLI for use in objectives, following the approach used by ECan officers at recent PNRRP hearings (e.g., Hayward et al. 2009) and the precedent set by EBOP and EW.

2.3 Study steps

This study has followed a series of steps as follows:

- Obtain water quality monitoring data from lakes and river inflows
- Obtain other relevant environmental data including local climate including wind and temperature
- Develop a proposed modelling approach
- Hold “OPEN WORKSHOP No. 1” (4 March 2009) - present proposed method to stakeholders and receive feedback
- Set up model
- Test model and calibrate model outputs against existing measured data
- Assess whether model is running effectively and that simulations are sufficiently robust to make predictions.
- Develop an approach to defining a range of future scenarios for nutrient increases
- Hold “OPEN WORKSHOP No. 2” (21 May 2009) - present proposed method to stakeholders and receive feedback
- Define a range of future nutrient load scenarios and test the effect of nutrient increases (using the model)
- Present model results - effects of nutrient increases on TLI and other measures of lake condition
- Analyse model results – relate changes in lake condition (e.g., TLI) to potential effects on lake values and the ‘purposes for management’ listed in section 2.1.
- Describe a range of options to meet various TLI objectives (i.e., options for managing the lake in a range of possible environmental states)

- Use model relationships to provide the necessary nutrient load caps for each TLI objective option
- Hold “OPEN WORKSHOP No. 3” (29 July 2009) - present draft report and receive feedback
- Finalise report.

The key outputs that address the purpose outline in Section 2.1 (i.e. measurable TLI objectives and linked nutrient load caps) are provided in tables in the conclusions (Section 6) of this report.

3 Methods

3.1 The models

Three models that were used for this study were developed by the Centre for Water Research, University of Western Australia (CWR) – DYRESM (**DY**namic **RE**servoir **S**imulation **M**odel), ELCOM (**E**stuary **L**ake and **C**oastal **O**cean **M**odel) and CAEDYM (**C**omputational **A**quatic **E**cosystem **D**ynamics **M**odel). These are complex, process-based (mechanistic) models that simulate coupled hydrodynamic, water quality and biogeochemical cycles in aquatic ecosystems. They were used for this study because of their ability to explicitly represent the physical, chemical and biological processes that control trophic state, and to resolve these processes spatially and temporally to account for the complexity of the Lake Benmore ecosystem.

Executable codes for these models are available publicly by request through the CWR website: http://www.cwr.uwa.edu.au which contains documentation and support links. Complete documentation, including science manuals and user guides that describe the processes, governing equations and parameters in the models, and sample input files, can be downloaded from the website.

DYRESM is a one-dimensional (in the vertical) hydrodynamics model that simulates the vertical distribution of temperature, salinity and density as lake-wide averages. It is suitable for multi-year simulations and sensitivity testing to long term changes in environmental factors.

DYRESM models lake-wide-average temperature stratification, while ELCOM is a three-dimensional hydrodynamics model used for predicting the velocity, temperature and salinity distributions in lakes, reservoirs and coastal systems. ELCOM models hydrodynamic and thermodynamic variables in three dimensions in individual grid cells. For the Lake Benmore application, the ELCOM grid consisted of cells with horizontal dimensions of 400 m by 400 m and depths of 2.5 m (Figure 2); a time step of 300 seconds was used. Because of its extensive demands on computer processing and memory resources, ELCOM was originally designed for simulations on time scales of days to weeks, although it can be run for as long as one year.

CAEDYM is a complex ecological model that simulates the cycles of carbon, nitrogen, phosphorus, silicon and dissolved oxygen in freshwater or marine ecosystems; phytoplankton dynamics; and inorganic suspended solids. The model has been used largely for assessments of eutrophication, being of the 'N-P-Z' (nutrients – phytoplankton – zooplankton) model format, but it also includes several other state variables not used in the present application to Lake Benmore.

CAEDYM is not operated as a stand-alone model; it is usually coupled with a hydrodynamic model that provides a framework for basin structure and bathymetry, water movements (mixing and circulation), river inflows, outflows, and heat exchange with the atmosphere. In this study, CAEDYM was run coupled with both DYRESM and ELCOM. DYRESM-CAEDYM was used initially for long-term runs (1994-2009) to test the suitability and stability of the initial choice of CAEDYM's model parameters. However, it soon became evident that a three-dimensional modelling approach was necessary to account for the complex shape of Lake Benmore and the possible interactions between its basins. This was confirmed in simulations with ELCOM that showed extensive water exchange between the Haldon, Ahuriri and Lower Benmore basins. The main focus of the application was on 8-month runs (1 August 2008 through 8 April 2009) with ELCOM-CAEDYM for model calibration against measured data, and 1-year runs for scenario testing (1 August 2003 – 31 July 2004; further discussion relating to choice of a representative year in terms of flows and climate is given in Section 3.7). Computational demands of the three-dimensional model, together with the deadline specified for project completion, precluded the use of ELCOM-CAEDYM for long-term (multi-year) runs.

We used stable executable code downloaded from the CWR website for the coupled models ELCOM-CAEDYM (release 19 June 2006) and DYRESM-CAEDYM (release 24 March 2006).

DYRESM was originally developed the 1970's, DYRESMWQ, the precursor to DYRESM-CAEDYM, in the 1980's, and CAEDYM and ELCOM in the 1990's. The models have been, and continue to be, used extensively and actively developed both by CWR and the international modeling community. The quality and reputation of the models is reflected by the number of publications in the international scientific literature describing research that uses the models. A list of the more recent publications includes Antenucci et al. (2003), Bruce et al. (2006), Burger et al. (2008), Copetti et al. (2006), Hillmer and Imberger (2005), Hillmer et al (2008), Hipsey et al (2004), Romero and Imberger (2003), Romero et al. (2004), Spillman et al. (2007, 2008), and Trolle et al. (2008a,b); papers submitted or in press include Chung et al. (2009), Hipsey et al. (2009a,b), Makler-Pick et al. (2009), and Smith et al. (2009).

3.2 Input data

A large amount of input data is required to run the model (we will use the term "model" for the rest of this report to refer to either or both of the coupled models DYRESM-CAEDYM or ELCOM-CAEDYM, depending on the context). This includes daily inflow volumes, temperatures and nutrients, and hourly or daily climate data. Daily inflow data were supplied by Meridian Energy Limited and the National Hydrologic Archive for the period 1994 – 2008. Annual minimum, maximum, mean and median for these data are presented in Appendix D. Climate data were obtained from the Tara Hills climate station, with additional inputs from Lauder, where necessary (Figure 3). DYRESM required daily climate data and ELCOM required hourly data. Annual averages of climate data are presented in Appendix D. Site locations for climate data and flow data are presented in Figure 3.

Ungauged inflows, including any net groundwater inflow to the lake, were estimated as residuals from a lakewide water balance, and a single point source was included for ungauged inflows in each Arm (Figure 2). In order to decide how much of the total ungauged (residual) inflow should be apportioned to the Haldon Arm and how much to the Ahuriri Arm, we used the CLUES model (Woods et al. 2006) to estimate mean annual inflows to both Arms from all streams and rivers. Based on the relative proportions of the inflows predicted by CLUES, we decided to allocate 90% of the total ungauged (residual) inflow to the Haldon Arm and 10% to the Ahuriri Arm. Nutrient concentrations from the Tekapo-Pukaki River samples were assigned to the ungauged inflows to the Haldon Arm. Nutrient concentrations from the Ahuriri River samples were assigned to the unguaged inflows to the Ahuriri Arm.

No quantitative information was available to us concerning either the quantity or quality of existing groundwater inputs to Lake Benmore. Neither was any information available regarding how such inputs are likely to change with intensifying land use and irrigation. In the model all net groundwater inputs are incorporated in the ungauged inflows, and the water quality of groundwater is assumed to be that of the ungauged inflow for each basin. Further discussion on water quality, and how it is assumed to change in the model application with intensifying land use and irrigation, is given below and in subsequent sections (3.3, 3.6).

Nutrient data were obtained from an intensive fortnightly summer water quality monitoring programme in 2008-09 (Appendix A). Data for nitrate (NO_3), ammonium (NH_4), dissolved reactive phosphorus (DRP), dissolved organic nitrogen (DON), dissolved organic phosphorus (DOP), particulate organic nitrogen (PON), particulate organic phosphorus (POP), particulate inorganic nitrogen (PIN), particulate inorganic phosphorus (PIP), dissolved organic carbon (DOC) and particulate organic carbon (POC) obtained from six sampling occasions from 8 December 2008 – 17 February 2009 were used for modelling. Measured water quality variables and sampling locations are shown in Appendix A. Averages of the data obtained from each of the three main river inflow sites were used to construct daily input files for the models. Averages were used because of the small number of data points available for modelling (four points in time when modelling started, six points by the time modelling was completed). It is important to note that these averages have been used to specify daily inflow concentrations for all the model runs, both for calibration and for the scenarios. Although inflow volumes and temperatures incorporate daily and seasonal fluctuations, inflow nutrient concentrations do not. Data (not averaged) from the in-lake sampling sites were used for model calibration and performance assessment. Averaged in-lake data were used to help specify initial conditions for the model runs; otherwise in-lake sample data were not used as input to the model but were used as verification of model runs.

3.3 Water and nutrient loads to Lake Benmore

Before proceeding with a description of more detailed modelling strategy and results, it is worthwhile to present some information on the long-term average water balance and an estimated nutrient balance to get a general perspective of the relative sizes of water flows and nutrient loads to the lake.

Water flows to Lake Benmore mainly arise from the Ohau C Canal inflow, with a mean annual inflow of 260 m^3/s, while flows from river and ungauged inflows (obtained as residuals from water balance calculations; see Section 3.2) are substantially less at 45 m^3/s or less (Figure 4). Discharge from the Ahuriri inflow to the lake is assumed to be 1.21 x gauged flow at South Diadem (location shown in Figure 3, red triangle marking the Ahuriri River gauging site), also based on proportions of mean annual flows from CLUES modelling. Outflows from the Lake are almost exclusively through the power station (Figure 4). Long term means are shown in Figure 4 and in Table E1, Appendix E. The Tekapo-Pukaki inflow was split between lake spill and gauged flow in Mary Burn and Fork Stream for the purpose of preparing scenario loads; increases in TN and TP concentrations for the scenarios were assumed to occur only in the Mary Burn and Fork Stream, and not in the spill from lakes Tekapo and Pukaki. This assumption is not strictly correct, as there may be some irrigation development on the shores of Lake Tekapo and Lake Pukaki. However, the assumption is made for convenience only and avoids the difficulty of having to determine separate sets of load factors for inflows to the Haldon Arm. The assumption is of no consequence in terms of the effects on the lake of total increased nutrient loads to the Haldon Arm from model scenarios.

Existing annual total nitrogen (TN) and total phosphorus (TP) loads to Lake Benmore have been estimated for each inflow based on water quality measurements obtained from six sampling occasion during 8 December 2008 – 17 February 2009 (Figure 5). Average inflow concentrations of TN and TP from these water quality measurements are also shown (Figure 5). The TN and TP loads in Figure 5 have been calculated as the product of the mean annual flow for the period 1994-2008 times the average concentration from the six samples (see also Tables E1-E7 in Appendix E for further load estimates). The loads are therefore approximations only and ignore any effects of variations in concentration with season or with flow, or possible long-term trends or cycles. Ohau C Canal provides the highest load to the lake but the lowest concentrations per cubic metre. The Ohau C Canal’s high flow coupled with low concentrations helps to maintain low concentrations in the Haldon Arm, thereby providing a flushing effect for the Haldon Arm, and to lesser extent the Lower Benmore basin and the lower reaches of the Ahuriri Arm. In the scenarios, nutrient concentrations were not increased in the Ohau C Canal; as noted above, increased concentrations were assumed to occur only in river flows that do not originate from lakes Tekapo, Pukaki or Ohau. This assumption is not strictly correct, in that proposed irrigation development near the Wairepo Arm will discharge water and nutrients, some of which will eventually find their way into the Ohau C Canal. As noted above in discussion of the spill from Lakes Tekapo and Pukaki, the assumption is one ofconvenience only and avoids the difficulty of having to specify separate sets of load factors for inflows to the Haldon Arm, all of which enter the Arm in nearly adjacent cells in the model. The assumption is of no consequence in terms of the effects of total increased nutrient loads to the Haldon arm from model scenarios.

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T	U	V	W	X	Y	Z	AA	AB	AC	AD	AE	AF	AG	AH	AI	AJ	AK	AL	AM	AN	AO	AP	AQ	AR
1	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40	41	42	43	44
2	2																											Ungauged North			-0.1	-0.7	-0.4					Tekapo Pukaki (2,32)						
3	3																											Ohau		-0	-0.1	-3.4	-4	-1.7	-0			Spill Tekapo Pukaki (2,33)						
4	4																													-0	-4.4	-5	-5	-4.8				Wet depos N arm (3,34)						
5	5																														-8.1	-6.4	-7.1	-7.5				Dry depos N arm (4,34)						
6	6																														-9.9	-9.6	-9.9	-9.9										
7	7																														-10	-10	-10	-10	-0.4									
8	8																													-7.5	-15	-12	-12	-14	-1.6									
9	9																													-14	-16	-15	-15	-15	-12	-3	-0.8							
10	10																													-12	-15	-15	-15	-15	-7.9	-3.9	-2.4	-0.2						
11	11																													-9.5	-19	-20	-18	-18	-18	-7	-5.9	-1.8						
12	12																													-12	-20	-20	-20	-20	-19	-14	-5.3	-2.5	-0.2					
13	13																														-27	-23	-21	-20	-19	-9.2	-7.3	-5.7	-2.1					
14	14																														-10	-27	-23	-22	-21	-14	-11	-8.9	-7.1					
15	15																															-25	-25	-25	-24	-14	-11	-9.2	-8.5	-3.3	-0.4	-0.2	-0.1	
16	16																															-16	-26	-26	-26	-21	-15	-12	-12	-7.8	-2.7	-2.2	-1.4	
17	17																															-3.5	-29	-29	-30	-30	-20	-15	-14	-12	-7.4	-5.3	-4.5	
18	18																																-31	-30	-30	-31	-27	-15	-15	-13	-10	-7.8	-6.6	
19	19																																-26	-30	-31	-34	-34	-13	-14	-15	-13	-5.6	-3.1	
20	20																																-22	-35	-35	-35	-35	-20	-14	-17	-16	-0.6		
21	21																																-14	-38	-37	-35	-35	-38	-16	-17	-8.2			
22	22																																-24	-34	-35	-35	-36	-34	-18	-15	-2.4			
23	23																																-28	-34	-35	-35	-38	-40	-20	-17				
24	24																															-0.4	-29	-35	-36	-37	-39	-40	-22	-19	-2.3			
25	25																																-29	-38	-38	-39	-40	-40	-23	-2.1				
26	26																																-31	-38	-40	-40	-41	-11						
27	27																																-33	-36	-43	46	-40							
28	28																																-28	-41	-47	46.5								
29	29																															-0.4	-30	-45	47	47								
30	30																															-16	-27	-55	-54	-42								
31	31																															-19	-27	-55	-55	-52	-0.4							
32	32																															-14	-24	-48	-58	-50								
33	33																															-6	-22	-18	-60	-46	-8.2							
34	34																															-1.1	-14	-33	-59	-58								
35	35																																-8.7	-0.3	-58	-60								
36	36																																	-4.4	-60	-60								
37	37																																-0.6	-62	-60	-53								
38	38																																-63	-62	-2.3									
39	39																																-61	-61										
40	40																																-10	-61	-31									
41	41																														-12	-57	-60	-60	-33	-32								
42	42																														-68	-61	-27	-14	-51									
43	43																														-65	-59												
44	44				Ahuriri																										-59	-59												
45	45			Ungauged west arm																											-17	-60	-4.3											
46	46																															-60	-20											
47	47																															-61	-25											
48	48			-0.1	-0.1	-0.24																										-62												
49	49	-0.1	-1.1	-2	-2.1	-2.04	-0.8																								-3	-63	-20											
50	50	-1.1	-3.9	-3.9	-4	-4.14	-4.9	-0.3																								-64	-64											
51	51		-5	-5.8	-6.3	-6.96	-8.3	-8.9																		-5.6							-64	-48	-30	-2.7								
52	52		-0.1	-6.7	-8.8	-9.12	-9.1	-9.9	-11	-16	-7.7													-1.7	-4.2	-36	-39						-68	-39										
53	53				-11	-9.74	-11	-12	-13	-15	-15	-16												-46	-48	-50	-52					-48	-70											
54	54					-13.5	-15	-16	-14	-15	-17	-20	-22									-26		-44	-6.5		-55	-57		-29	-73	-1.4	-71	-72										
55	55					-0.08	-13	-16	-16	-17	-18	-20	-24	-25	-25	-30	-30	-29	-30	-29	-29	-34	-39	-42				-60	-9.4	-74	-74	-1.4		-73										
56	56			Dry depos W arm					-17	-17	-17	-19	-22	-23	-10	-26								-33	-34		-62	-60	-30	-66	-75	-75	-74	-73										
57	57			Wet depos W arm						-11	-9.8	-13	-15	-17										-31	-0.9		-65	-5.6	-30	-70	-76	-6												
58	58									-1.1	-3.9	-7.2	-8.9	-7.5													-70			-71	-77	-78	-78	-4.2										
59	59											-0.1	-2.7	-0.7													-70	-71	-71	-71	-77	-7.7	-79	-82	-44									
60	60																											-1.8	-3	-62	-35			-83	-81	-23								
61	61																																	-82	-82	-14								
62	62																																	-82	-76	-0.8								
63	63																															-13	-60	-82	-54	-6.2								
64	64																																-56	-10		-26								
65																																												

- Gauged inflows
- Ungauged inflows
- Islands
- Profile sites (field sampling and model output)
- Profile sites (model output only)
- Wet deposition
- Dry deposition
- Outflows
- Curtains
- Curtains join

Profile sites

Elcom set number	Profile number	Profile name	Location	
7	1	Ahuriri	55, 13	Field sampling site and model output
8	2	Haldon	33, 35	Field sampling site and model output
9	3	Benmore	62, 34	Field sampling site and model output
10	4	Ahu0	55, 28	Model output only
11	5	Ahu1	55, 23	Model output only
12	6	Ahu2	52, 7	Model output only
13	7	Hal0	53, 33	Model output only
14	8	Hal1	45, 32	Model output only
		Hal2	18, 35	Model output only
		Hal3	9, 33	Model output only

Figure 2: Schematic map of lake Benmore on a 44 x 64 cell model grid (400 m x 400 m cells) showing locations of inflows, outflows, curtain transects for model output and profiling sites (both for field sampling and model output). The absolute values of the numbers in the cells are depths (m).

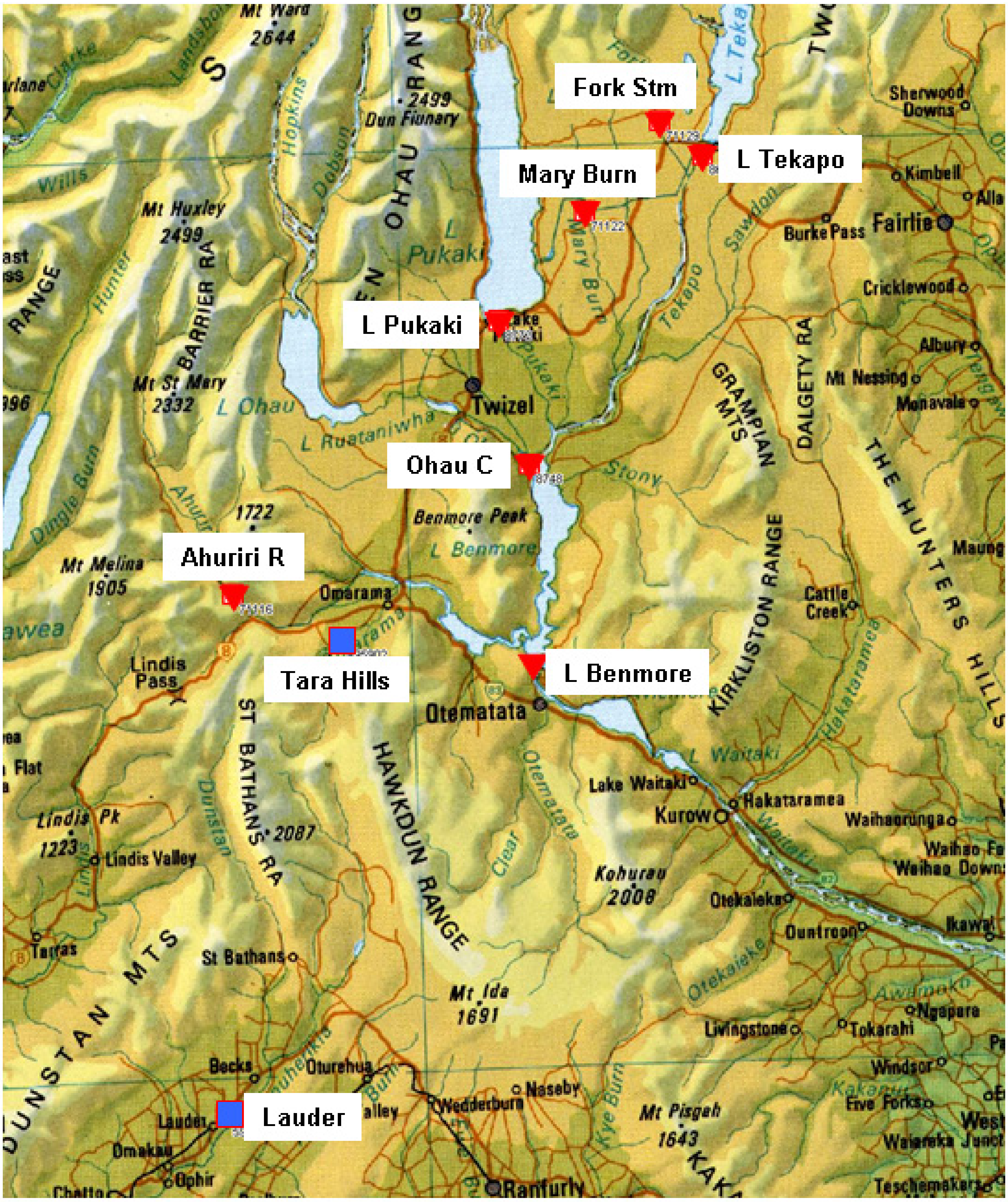

Figure 3: Locations of climate stations (blue squares) and inflow recorders (red triangles) where data was collected for input into the model.

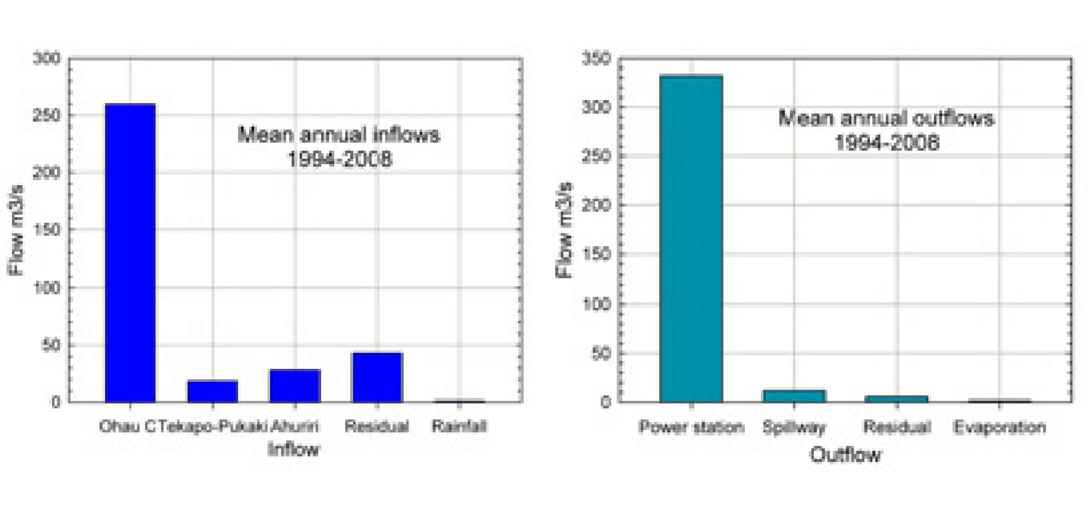

Figure 4: **Mean annual inflows and outflows for Lake Benmore over the periods 1994 – 2008; "residual" refers to ungauged flows, calculated as residuals from water balance.**

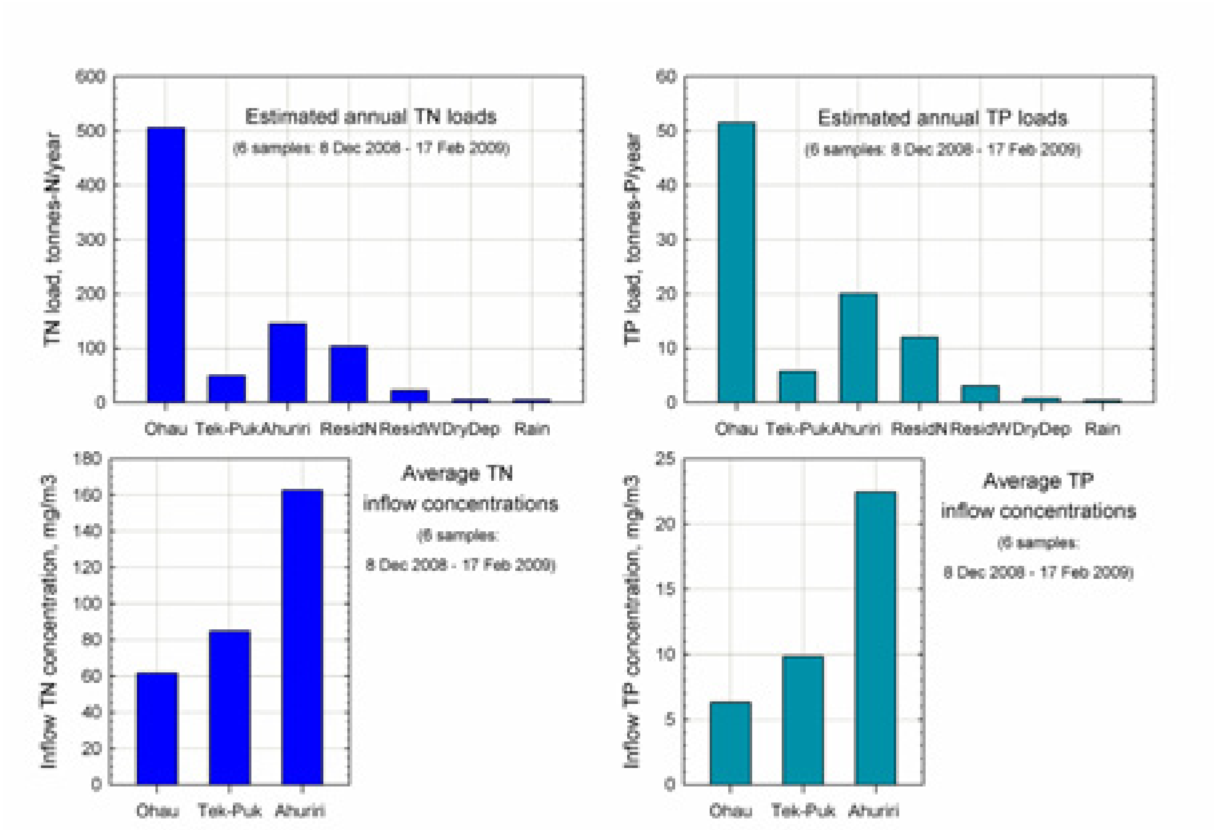

Figure 5: **Estimated annual total nitrogen (TN) and total phosphorus (TP) loads for each inflow into Lake Benmore. "Residual" refers to ungauged flows, calculated as residuals from water balance. Average inflow concentrations of TN and TP are also given for each inflow. Estimates are based on inflow measurements taken on six sampling occasions between 8 December 2008 and 17 February 2009, and the mean annual flows shown in Figure 4.**

3.4 Model configuration and calibration strategy

The model was calibrated against existing baseline conditions in Lake Benmore with model runs for the period 1 August 2008 – 8 April 2009. Although sample data were not available until early summer (8 December 2008), the model was initialized with fully-mixed winter profiles for temperature and nutrients, both because we had insufficient information to specify summer stratification values with adequate resolution, and because we wanted to check the model's ability to simulate conditions over an almost-complete season. We finished the calibration runs in early April, in the hope that data from an early April sampling trip would be available, but only data through mid-March was available to us when modeling was completed. The model was initialised with a uniform temperature of 6.5°C based on the temperature data of Pickrill and Irwin (1986), and uniform concentrations based on the averages of in-lake sample data available when modeling began.

This calibration period overlapped the water quality sampling period (8 December 2008 – 17 February 2009, Appendix A) and results from the modelled output could be compared against actual measured values of in-lake temperatures, dissolved oxygen, chl *a*, TN, TP concentrations etc. observed at three different sites within Lake Benmore (Appendices A and C).

ELCOM is generally free of calibration, as hydrodynamics, temperatures and water velocities are driven purely by the physics of conservation of mass, momentum and energy. CAEDYM, however, has adjustable parameters (see Appendix C for parameter values relevant to this application) and these must be adjusted in order to calibrate the model. CAEDYM was configured to include two different phytoplankton groups to account for some of the observed spatial and temporal variations from the measured phytoplankton species abundance corresponding to successional changes in Lake Benmore (Appendix A). The conceptual model for phytoplankton includes light-, nutrient- and temperature-dependent growth, sedimentation and loss of biomass due to respiration, mortality and excretion. Availability of light for photosynthesis is controlled by an optical sub-model within CAEDYM that attenuates solar radiation with depth. The sub-model includes light attenuation by pure water, dissolved organic matter, chlorophyll-a, and suspended sediment. Attenuation by all variables, except pure water, is concentration-dependent, and provides the chief means for feedback between biology and physics in the model, because of the dependence of thermal stratification on absorption of solar radiation in the water column. Zooplankton, and the subsequent loss of phytoplankton biomass through grazing, was not explicitly included in the biota model for Lake Benmore because of lack of information required to calibrate parameters that control zooplankton dynamics. Hence, phytoplankton parameters such as respiration rates were calibrated to compensate for any grazing pressure missing in the model. CAEDYM parameters were adjusted manually in stepwise fashion, within intervals derived from literature and applications to oligotrophic lakes (Hamilton and Schladow 1997, Schladow and Hamilton 1997), and that were consistent with limnological processes and previous modelling experience. The manual calibration included repeated trial and error adjustments of parameters until there was minimal improvement in the agreement between simulated variables and measurements.

The calibration focussed mainly on chl *a*, and secondarily on dissolved oxygen, TN and TP. Although the processes modelled by CAEDYM are strongly interrelated, so that changes in a parameter for one process affects results from several other processes, it has been found from experience that results are more sensitive to some model parameters than others (Hamilton and Schladow 1997, Schladow and Hamilton 1997, Spigel et al. 2003, Trolle 2009). The Lake Benmore application was no exception. For chl *a*, these parameters are the maximum potential specific growth rate, respiration rate, and parameters controlling the minimum internal storage capacities and uptake rates for phosphorus and nitrogen. For dissolved oxygen, TN and TP, sensitive model parameters are the static sediment oxygen demand, water-column mineralisation rates for organic nitrogen and phosphorus, and sediment release rates for ammonium and phosphate (although all of these processes are also temperature and concentration dependent). In-lake values of TN and TP were most strongly governed by inputs of TN and TP specified in the input files for the inflows, with only a minor influence from in-lake model processes and parameters. In summary, the values selected for CAEDYM parameters fit well within literature values, with some differences between the properties of the two different phytoplankton groups, as would be expected.

The influence of glacial flour on limnology of the Haldon Arm (and to lesser extent the Lower Benmore basin) was accounted for in the model mainly through the dependence of the extinction coefficient for solar radiation on the concentration of suspended sediment in the water column, and secondarily through the model parameters governing phytoplankton production as a function of light intensity. There is very little scientific literature on oligotrophic lakes influenced by glacial flour, and what literature we have found focuses on the effects of glacial flour on optical properties (Vincent et al. 1984, Sommaruga et al. 1999, Gallegos et al. 2008). It was beyond the scope of this project to revise the optical sub-model in CAEDYM (described briefly earlier in this section) to specifically account for the unique effects of glacial-flour on light transmission, absorption and scattering in the water column. However, it was possible to obtain a good calibration for light attenuation with the existing sub-model. As noted above, the sub-model accounts for the effects of suspended sediment and chlorophyll-*a* on light attenuation. Thus, the increasing importance of attenuation by chlorophyll-*a* as nutrient loads increase is accounted for in the model.

Due to the limited available data describing existing water quality variables in Lake Benmore, statistical fitting methods or goodness-of-fit tests were not used during calibration, which instead had to follow a subjective approach (hence, based on expert opinion) by visually comparing modelled output against observed values. To put this procedure in a broader perspective, we quote from a paper by Robson et el. (2008, p. 376), that discusses recommended practices, difficulties and methods for calibrating and evaluating performance of models like ELCOM-CAEDYM: "…it is not yet feasible to apply these techniques [statistical fitting or optimisation algorithms for model calibration] to coupled three-dimensional hydrodynamic and biogeochemical models at reasonable resolutions … In practice, complex biogeochemical simulation models are most commonly calibrated by trial and error: An expert modeller with an understanding of both the biophysics of the system and the structure of the model compares model results with field data either by eye or with the aid of some measure of goodness of fit, and adjusts parameter values by trial and error within literature ranges."

Ideally, model application includes separate steps for calibration and validation (Beck 1983a,b) – some field data are used for calibration, and a separate set are then used for an independent validation. A minimum length of record is typically one year of monthly sampling for each of these steps, though studies referred to in Section 5.1 in discussion of model uncertainty (Arhonditsis and Brett 2005a,b; Burger et al 2008; Trolle et al. 2008 a,b, Trolle 2009) use much longer records (see also notes to Table F-2, Appendix F). In the Benmore application, because of the highly restricted period of time for which spatially resolved measurements of water quality were available, all of the water quality data from the six samples collected between 8 December 2008 and 17 February 2009 were used for calibration. It was not possible to have an independent validation period to further test the performance of the model. To provide an estimate of model errors, however, absolute and normalised absolute errors were calculated after the calibration (see Table F.1, Appendix F; further discussion on model uncertainty is given in Section 5.1).

By the time calibration was completed, results from a further two sample trips made in March (3-4 March and 16-17 March) became available. These data were not used for calibration but have been included in the model calibration output plots in Appendix C (Figures C.1 – C.9), and in the calculation of root-mean-square-errors in Table F.1, Appendix F. They represent the only independent validation data available at the time of writing.

3.5 Mesocosms

In order to establish if phytoplankton in Lake Benmore will respond to changing nutrient (nitrogen and/or phosphorus) concentrations, an *in vitro* mesocosm experiment was conducted under conditions of increased nitrogen, increased phosphorus and increased nitrogen + phosphorus. The mesocosms provide a small-scale replication of the “real” system for the purposes of testing phytoplankton responses to changes in nutrient availability. Complete methodology and results from the mesocosm experiment are presented in Appendix B.

3.6 Scenario nutrient loads

Nutrient-load scenarios were run to examine the possible response of the lake to a range of nutrient loadings. Criteria used for selecting the scenario loads were:

- loads should cover the range from existing to maximum anticipated as a result of land-use intensification;
- loads should be as realistic as possible in terms of relative loadings between the Haldon and Ahuriri Arms of the lake, and in terms of relative proportions of different components of nitrogen and phosphorus.

At the time of model setup and running, the only sources of information available to us on possible future nitrogen and phosphorus loads were the 2004-2005 reports of GNS (White et al. 2004), HortResearch (Green, 2005), AgResearch (McDowell 2004) and NIWA (Snelder et al. 2005). These reports did not provide a consistent set of loads or concentrations that were suitable for the lake model. Even for the dry-land farming scenarios in the reports, corresponding to existing conditions, there were not nitrogen and phosphorus concentrations that were consistent with those measured during the monitoring programme that started in December 2008 for the present Lake Benmore study (Appendix A). The 2004-2005 reports did give an indication of the *relative* increases that could be expected over the range of scenarios, suggesting that a maximum of 12 times increase in current loads for both nitrogen and phosphorus could be possible. The relative increases in load showed no obvious pattern that differentiated the two arms of the lake.

The scenarios have been based on measured data (from December 2008 – February 2009) from the monitoring programme for the present Lake Benmore study. We have used the maximum increase in loads of 12 times current values of nitrogen and phosphorus to construct the range of scenarios for the lake model. Each scenario is associated with a single load multiplication factor that is applied to existing average concentrations of TN and TP measured in the Ahuriri River for the Ahuriri Arm, and in the Tekapo-Pukaki River for the Haldon Arm. The inflow volumes are not assumed to change significantly with increasing irrigation (Brian Ellwood, Meridian Energy Limited personal communications May 2009), so that the load multiplication factor can be interpreted as applying either to loads or concentrations. The multipliers were also applied to the ungauged inflows (obtained as residuals from water balance) for each arm, with Ahuriri River concentrations assigned to the ungauged inflow to the Ahuriri Arm, and Tekapo-Pukaki River concentrations assigned to the ungauged inflow to the Haldon Arm. No multipliers were applied to concentrations in either the Ohau C Canal or to the spill from Lake Tekapo or Lake Pukaki; concentrations in these inflows were assumed to remain at base (existing) levels for all scenarios, based on the assumption that there is little potential development around the shorelines of either of these lakes or in their upper catchments. Any nutrient input into these lakes is assumed to be sufficiently small that concentrations would not be measureable in the main bodies of the lakes above current concentrations. In this report, results are presented for scenarios with load multiplication factors of 2, 4, 6, 8, 10 and 12 for the Ahuriri River, the Tekapo-Puakaki River and ungauged inflows. Because the same multiplication factor is applied to both TN and TP for a given scenario, this means that existing TN:TP ratios in the inflows remain the same for all scenarios, equal to those for the base scenario of existing conditions. We note that in the 2004-2005 reports, ratios of nitrogen to phosphorus decreased as loading increased. However, from discussions with Dr R. Wilcock (NIWA) and Dr. R. McDowell (AgResearch), and from data presented in Monaghan (2008), nitrogen to phosphorus ratios would be expected to either remain roughly the same or to increase as load increased with intensifying land-use and irrigation.

It was pointed out at the second Environment Canterbury workshop (Waitaki Lakes Water Quality: a method to define sustainable thresholds for nutrient loading for regional planning; Development Workshop 2: 21 May 09) that as TN and TP loads increase with intensifying land-use and increasing irrigation in the catchment, the relative proportions of inorganic and organic components can be expected to change, with the dissolved inorganic fractions (NH_4, NO_3, DRP) becoming relatively larger than the dissolved organic fractions (also see Wilcock 2008). In an effort to include this effect, we have obtained unpublished data from the Best Practice Dairy Catchments Project (see NZ Dairy Research Institute 2001; Wilcock et al. 2008, MfE 2009) from R. Wilcock, for five monitored catchments heavily influenced by dairy farming. The catchments are spread throughout New Zealand, including one (Waikakahi) located in the Lower Waitaki River Valley. The catchments have been monitored monthly for from 8 to 14 years, with a more intensive fortnightly monitoring period of 2 years at startup. (Inchbonnie, on the West Coast of the South Island, has only been monitored for 5 years, but was not used in developing the relationships discussed below). These data are presented in Appendix: E - Nutrient Load Scenario Selection Data as Figures E.1 and E.2. Because the load factors will be applied to existing values of TN and TP in the inflows to Lake Benmore, TN and TP data from the Benmore inflows are included in the plots as well.

The rationale and method used to partition TN, TP into inorganic, organic, dissolved and particulate components that were used as inputs to the model for a given scenario are summarized as follows. It can be seen (Figures E1, E.2) that as TN and TP increase, the fractions made up by nitrate (NO_3) and dissolved reactive phosphorus (DRP) increase rapidly at first, and then appear to approach limiting values of around 70-80% for NO_3 and 50-60% for DRP. The fraction of total nitrogen (TN) made up of ammonium (NH_4) appears to either decrease slowly, or remain constant. It was decided to use this behaviour for the *fractions* of TN and TP made up by the dissolved inorganic components as the basis for determining the corresponding fractions in the scenario loads. To do this it was necessary to fit curves to the data that extended from the base values of TN, TP in the Lake Benmore inflows to the maximum values of TN, TP in the Best Practice Dairy Catchment data; this would include the total range required for the scenarios, and allow values to be interpolated within this range for individual scenarios. As a first step nonlinear regression curves were fit to the data, shown as blue curves in Figures E.3 and E.4. Points for Inchbonnie catchment (West Coast, South Island), and the Ohau C Canal were excluded, Inchbonnie because of its unusual and unique character (R. Wilcock, personal communication, June 2009), and Ohau C because its concentrations are not altered in the scenarios. The second step involved modifying the regression curves slightly by trial and error to give curves that passed through the Lake Benmore data for the Ahuriri and the Tekapo-Pukaki rivers. These curves, and the formulas for them, are also shown in Figures E.3 and E.4, and were used to interpolate for the fractions of NO_3, NH_4 and DRP for scenarios as functions of TN, TP. Values for concentrations of NO_3, NH_4 and DRP were obtained by multiplying the fraction by the concentration of TN or TP. To derive concentrations for the remaining nitrogen and phosphorus components, it was assumed that the concentrations of particulate components do not increase as TN and TP increase, but remain at their existing concentrations for all scenarios. This may underestimate particulate component concentrations, but the little data available for particulate nitrogen and phosphorus from the Best Practice Dairy Catchments show that these concentrations make up a very small percentage (approximately 1%) of total nitrogen and phosphorus in the monitored catchments (see further discussion in next paragraph). Dissolved organic components can then be derived by difference between the concentrations of TN and TP, minus the sum of dissolved inorganic components plus particulate components. The resulting scenario concentrations are shown graphically in Figures E.5 and E.6; and are tabulated for the TN, TP multipliers 2, 4, 6, 8, 10, 12 in Tables E.2 - 7. Also shown for illustrative purposes are annual loads estimated from these concentrations and the mean annual flows in Table E.1, for the scenario simulation year 1 August 2003 – 31 July 2004. Mean annual flows for the 15 year period 1994-2008 are also given in Table E.1.

Because the dissolved organic components are derived by difference, underestimating the particulate components of total nitrogen and phosphorus concentrations will overestimate the concentrations of dissolved organic components. This will have the effect of increasing the nutrients that are eventually available for phytoplankton uptake in the lake. However, this is a secondary effect in the model because the organic components, both dissolved and particulate, need to be mineralized before they are available for uptake, and not all of the organic components are mineralized. Furthermore, some of the fraction of any possible increase in particulate components that is organic will in fact find its way into the pool of available nutrients in the lake, subject to lability, mineralisation rate and sinking velocity.

Since the scenario input files were created and the model runs carried out, a new publication has become available from the Ministry for the Environment (MfE 2009a) that presents data from 14 dairy farming catchments in New Zealand from 2001-2007, with a focus on data collected in 2006-2007. The catchments include the five catchments that are monitored in the Best Practice Dairy Catchments project described above and referred to as "Tier 1 catchments" in the MfE (2009) report, plus a further nine predominantly dairying catchments referred to as "Tier 2 catchments" in the report. As with the Tier 1 catchments, the additional catchments are spread throughout New Zealand, but monitoring data generally extends over a much shorter period and with a lower frequency of data collection. The report does not present data that could be used to modify the method for partitioning TN and TP into components. Regarding the question of how particulate components change as dairying increases, the report does say that "four out of 13 catchments had median turbidity levels that were at, or in excess of ANZECC guidelines for ecosystem protection" (MfE 2009, p. viii), but does not provide details on the composition of suspended solids that presumably are responsible for the turbidity. The report points out that turbidity is governed by stock access to waterways and the erodibility and stability of the land, factors which vary from catchment to catchment in the survey.

3.7 Modelling strategy

The time available for completing this report did not allow us to run long-term simulations with ELCOM but it was necessary to run scenarios for a full year in order to calculate statistics for trophic level index (Burns et al 1999, Burns et al 2000). It was decided to model a year that best typified climate, inflow and outflow variables for the Upper Waitaki Basin. August 2003- July 2004 was chosen, based on examination of annual means for climate variables, inflows and outflows (see Appendix D for details).

Each scenario was run two times for a full year (1 August 2003 – 31 July 2004), to ensure that the simulation was depicting conditions that were as close as possible to a long-term equilibrium for the nutrient loading associated with that scenario, and not an evolving trend in water quality conditions. For the first run the initial conditions for in-lake nutrient concentrations and sediment oxygen demand were as for the existing, oligotrophic base conditions. As would be expected, at the end of the first run for a scenario with nitrogen and phosphorus loads increased above existing levels, the in-lake nutrient and chl *a* concentrations were higher at the end of the year than at the start of the year. For the second run, the initial conditions for nutrient concentrations were reset to in-lake values averaged over the lake volume and over the last 5 days of the first run.

The sediment oxygen demand parameter for CAEDYM was also recalculated using the relationship given by Schallenberg and Burns (1999), which relates areal hypolimnetic oxygen demand (AHOD) to summer mean epilimnion chl *a* concentration, euphotic depth, and lake mean depth. Chl *a* and euphotic depths from the first run were used to increase the sediment oxygen demand parameter by a factor based on the ratio of oxygen demands specified by the Shallenberg-Burns relationship. This relationship was developed using data from four New Zealand lakes with trophic levels ranging from ultra-oligotrophic to eutrophic, and "predicts the AHOD in New Zealand lakes remarkably well." (Schallenberg and Burns 1999, p. 16). Ideally the iteration procedure of running the model for a year, revising the initial conditions and sediment oxygen demand, and rerunning the model could be repeated until end-of-year results either exhibited minimal further change or agreed with initial conditions within some specified tolerance. However, initial testing indicated that convergence was rapid and one iteration was considered satisfactory. Modelled data output from the second run were used to extract the statistics and other results presented in Section 4 and Appendix F.

4 Results

Results are presented as summer and annual averages for each of the three main basins (Haldon (North) Arm, Ahuriri (West) Arm and Lower Benmore) and for the entire lake. While trophic level index (TLI) is traditionally calculated on annual averages, in cases where there are extremes between summer and winter productivity, such as in the modelled results for the Ahuriri Arm and Lower Benmore (see Figures 19-20), annual TLI scores may not well characterise the summer period. Protecting the summer TLI, or dissolved oxygen, should be regarded as fundamentally important for the management of Lake Benmore as this is the period most often associated with nuisance, and / or toxic algal blooms and is generally the period of highest recreational use. For the purpose of this report, we have defined the summer period from 15 November 2003 – 15 March 2004.

A full set of results is presented in Appendix F; those most relevant to establishing TLI objectives are presented and discussed in Sections 4.1 – 4.5.

With the exception of Section 4.5, most of the discussion in the remainder of this section focuses on means – average concentrations over a year or a summer. Standard deviations have also been calculated for the yearly model results for epilimnion chl *a*, TN and TP, and above-bottom dissolved oxygen. These results are presented in Figure F.29, Appendix F, as plots of coefficient of variation (standard deviation divided by the mean) versus increasing scenario load and are discussed further in Section 4.5.

In order to extract statistics from the model output to use in assessing some of the trophic level objectives, it was necessary to make a distinction between epilimnion and hypolimnion values when the water column was thermally stratified. The epilimnion was assumed to consist of water above 30 m depth, and the hypolimnion of water below 30 m. The use of 30 m as the depth of the main thermocline was based on examination of temperature profiles measured at the sampling sites in the Haldon Arm and the Lower Benmore basin.

4.1 Minimum Hypolimnion Dissolved Oxygen

Summer and annual minimum dissolved oxygen (DO) concentrations (mg/L) and percent saturations averaged through the hypolimnion are presented in Figures 6-9. There is no difference between summer and annual minimum DO concentrations and percent saturation. Minimum DO concentrations ranged from 9.9 – 7.7 mg/L in the Haldon Arm, 9.3 – 5.9 mg/L in the Ahuriri Arm, 10.4 – 8.4 mg/L in the Lower Benmore and 9.4 – 5.9 for the whole lake, from the current baseline (referred to as baseline from hereon) to the 12 x nutrient loading (Figure 6-7 and Appendix E). Minimum percent saturation of DO ranged from 89 – 56 % in the Ahuriri Arm across the full range of nutrient scenarios. Minimum percent saturation fell below 70% in the Ahuriri Arm as baseline nutrient loads were increased around 4–fold (Figures 8-9). Percent saturation in the Haldon Arm and Lower Benmore ranged from 90 – 72% and 90 - 77%, respectively (Figures 8-9). Threshold guidelines for dissolved oxygen for maintaining the health of aquatic life are 80% saturation (Third Schedule RMA) and 5 mg/L for protecting fish habitat (ANZECC 1992).

4.2 Minimum above lake-bed dissolved oxygen

The absolute minimum DO concentration and percent saturation occurred just above the lake bed (Figures 10-13). Minimum DO was the same in summer as for the whole year, i.e. lowest DO occurred during summer, with the exception of Lower Benmore, whose inflows arise from the Ahuriri and Haldon Arms. The reason for this difference may be the relatively smaller volume and greater isolation of the deepest water in the Lower Benmore basin, which is not reoxygenated by deep mixing or river inflows for a longer period than bottom water in either the Haldon or Ahuriri Arms. At all sites, as nutrient input increased under the different scenarios, minimum DO concentrations and percent saturations decreased. The most impacted basin was Ahuriri Arm, where DO concentrations and percent saturations decreased from 8.8 to 4 mg/L and 88% to 40%, respectively (Figures 10-13). DO concentration decreased below 5mg/L at around 6 x baseline nutrient loads in the Ahuriri Arm, and 8 x baseline loads in the Haldon Arm and Lower Benmore, while percent saturation at all sites decreased below 70% at, or before 4 x baseline loads (Figures 10-13).

4.3 Chlorophyll *a* (chl *a)*

Summer mean chl *a* concentrations in the epilimnion were higher than the annual means for all three basins. The highest chl *a* concentration predicted in the Haldon Arm was approximately 2.5 µg/L in the summer under the highest (12 x) nutrient loading scenario. The Haldon Arm shifted two trophic levels as defined by the TLI (from microtrophic to mesotrophic) between the 8 x and 10 x load scenarios during summer and one trophic level (from microtrophic to oligotrophic) across the whole year for nutrient loadings corresponding to 6 – 12 x baseline (Figures 14-15). Chlorophyll *a* concentration in the Ahuriri Arm reached eutrophic levels (>5 µg/L) between 2 x and 4 x baseline loads in summer and at 4 x baseline loads across the whole year (Figures 14-15). When nutrient loadings increased to 6 x baseline in the Ahuriri Arm, the model predicted that summer mean chl *a* would reach supertrophic levels (>12 µg/L), a shift of three trophic levels from the current baseline oligotrophic level. In the Lower Benmore, chl *a* concentrations reached eutrophic levels at 8 x baseline nutrient loads during the summer period (Figures 14-15). Chlorophyll *a* in the Lower Benmore moved up two trophic levels (from microtrophic to mesotrophic) when nutrient loads were increased to 4 x baseline loads in the summer, and 6 x baseline loads for the entire year. Averaged across the entire lake, chl *a* concentrations increased from microtrophic (<1 µg/L), at baseline, to eutrophic levels (5 µg/L, Table 1) between 4 and 6 x baseline nutrient loads during the summer period (Figures 14-15).

The modelled summer and annual chl *a* concentrations shown in Figures 14-15 are based on averaging concentrations over their respective time periods. However, within these time periods, very high chl *a* peaks were reached for short durations even under moderate nutrient loads. In the Ahuriri Arm, chl *a* concentrations peaked at 17 µg/L at 4 x baseline nutrient loads and 28 µg/L at 6 x baseline (see Figure 19); values greater than 30 µg/L (the maximum shown for the colorbar in Figure 19) occurred in all scenarios with loads of 8 x baseline and greater, with peak values > 40 µg/L predicted for the 12 x baseline scenario.

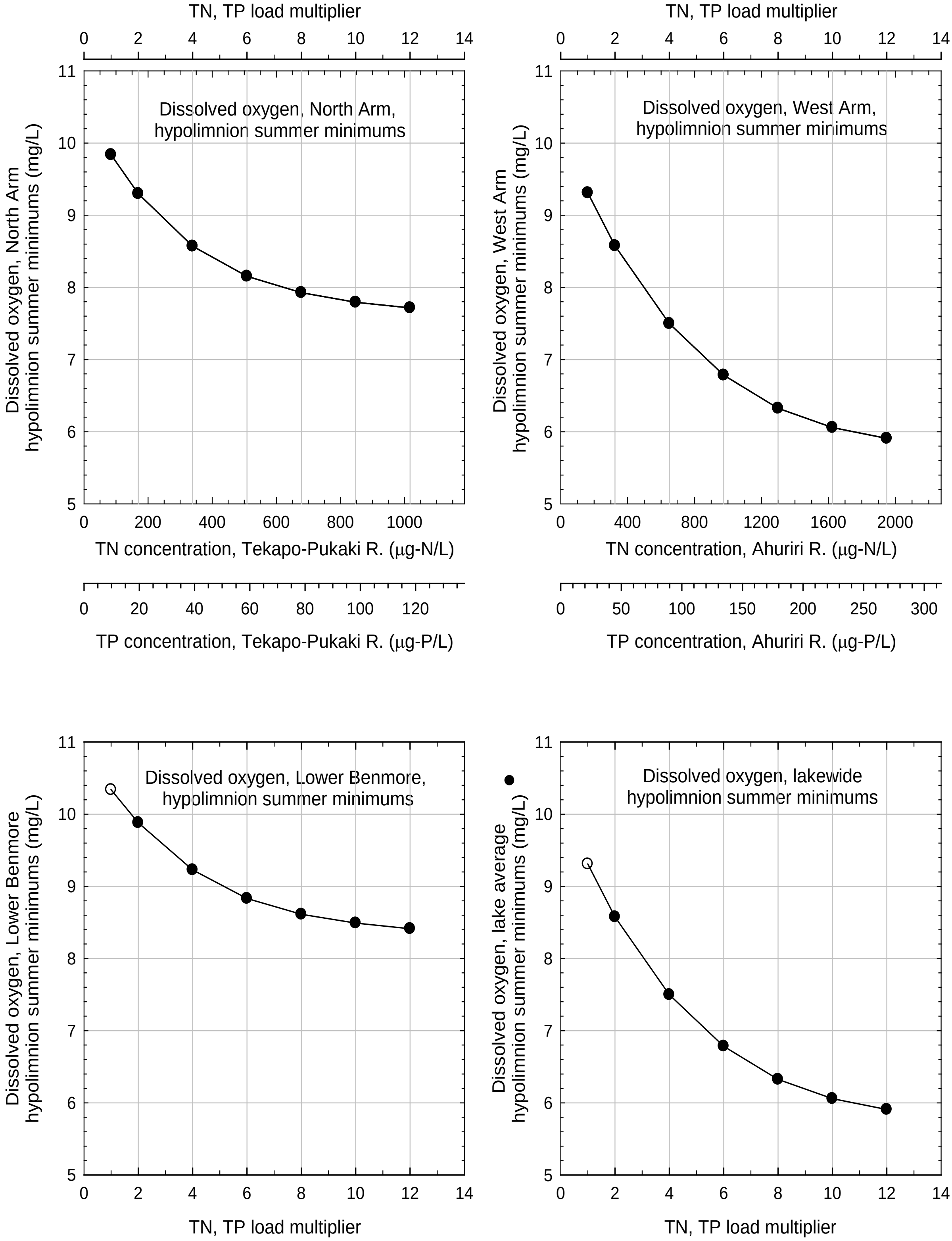

Figure 6: **Minimum dissolved oxygen concentrations (DO mg/L) for the hypolimnion during the summer period (15 Nov 2003 - 15 March 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

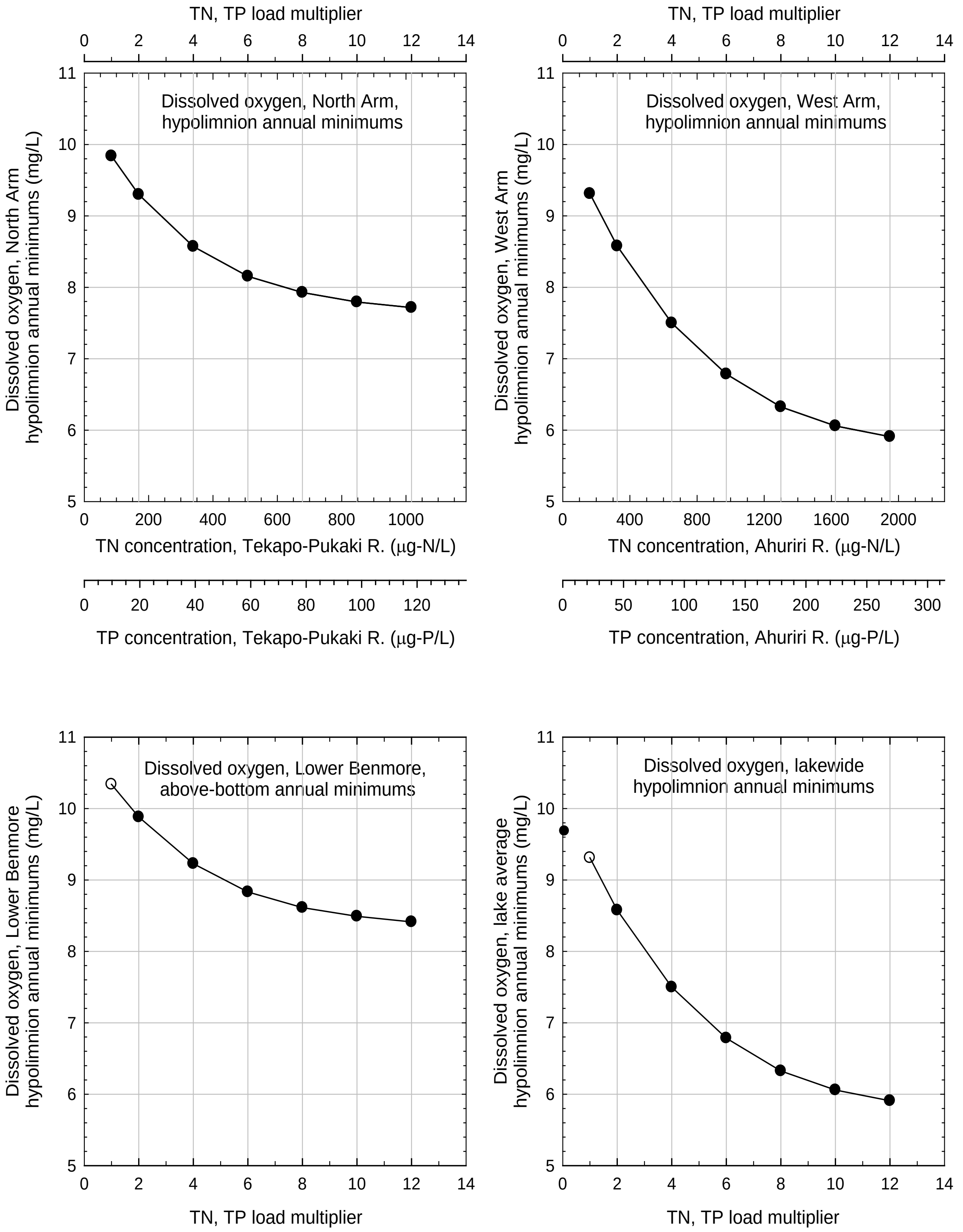

Figure 7: **Minimum dissolved oxygen concentrations (DO mg/L) for the hypolimnion for the year (1 Aug 2003 -31 Jul 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

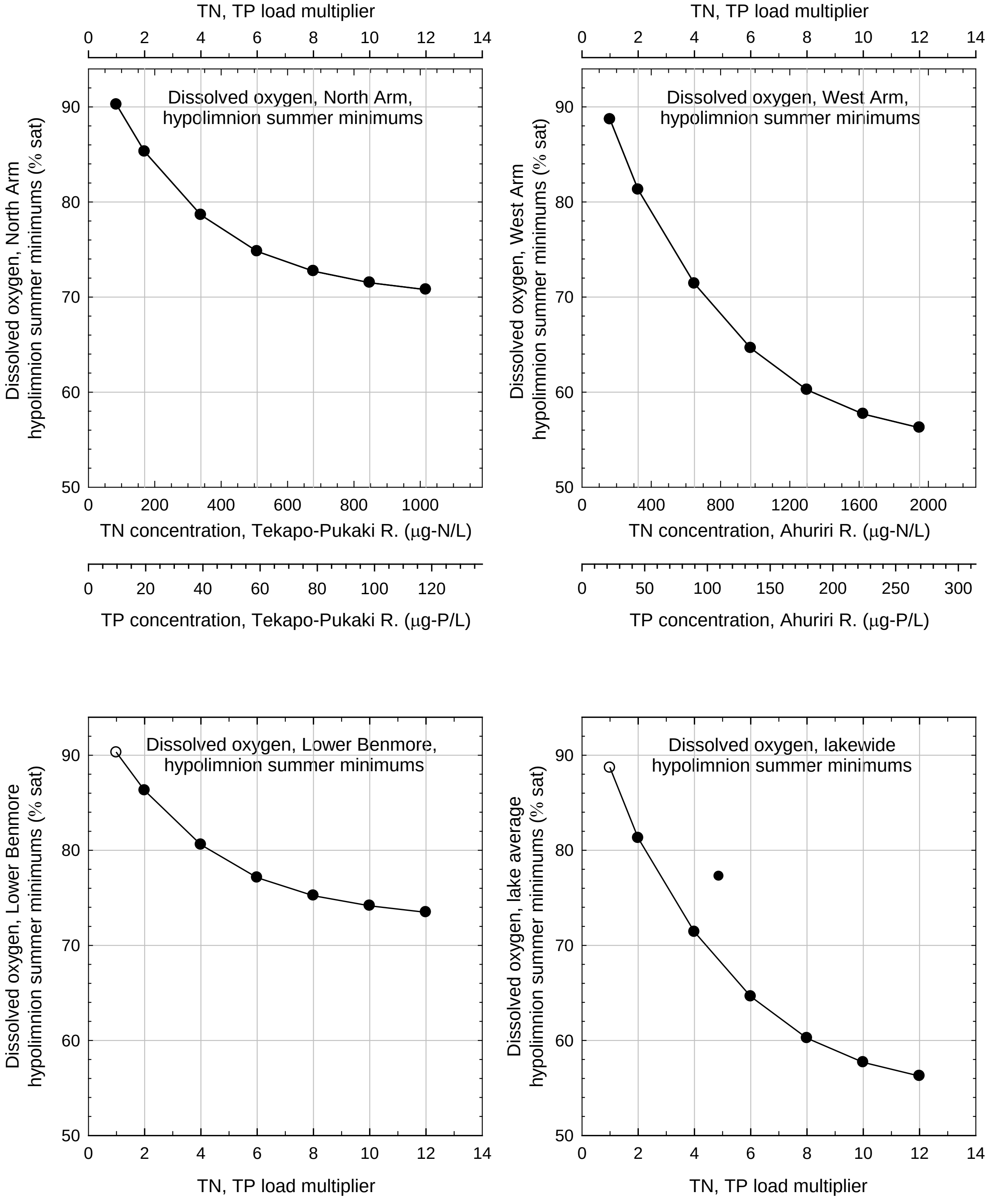

Figure 8: **Minimum percent saturation of dissolved oxygen for the hypolimnion during the summer period (15 Nov 2003 - 15 March 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

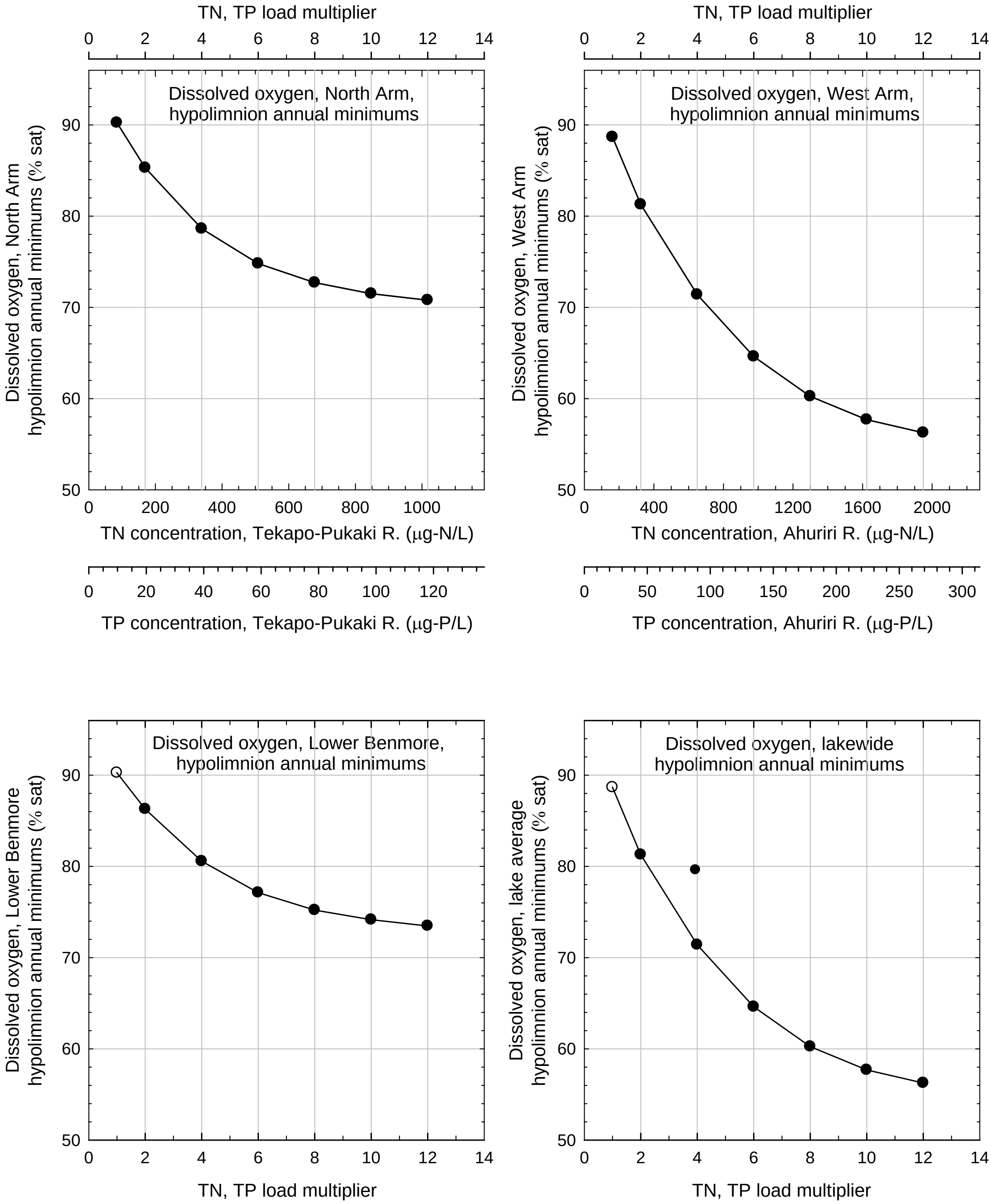

Figure 9: **Minimum percent saturation of dissolved oxygen for the hypolimnion for the year (1 Aug 2003 -31 Jul 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

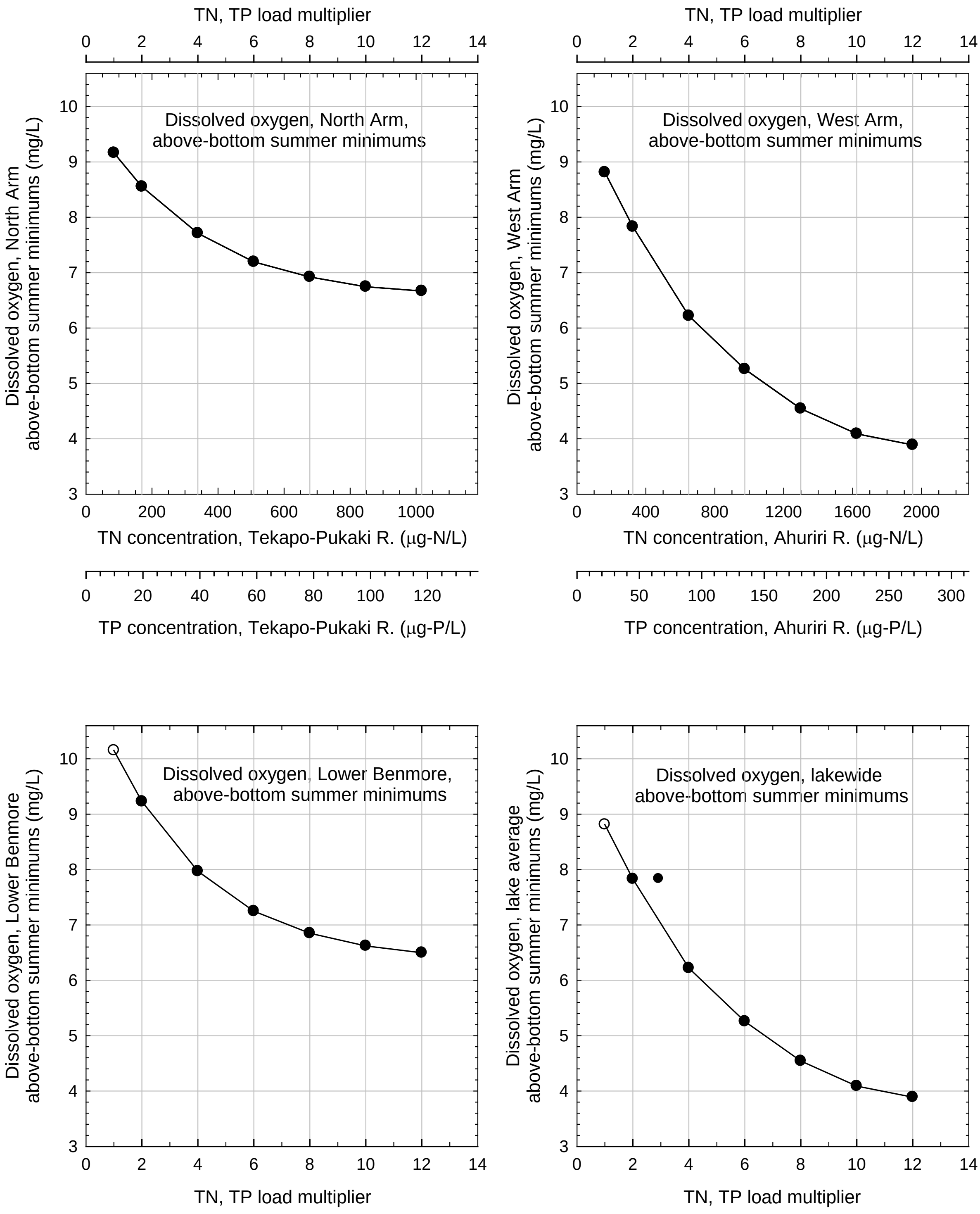

Figure 10: **Minimum dissolved oxygen concentrations (DO mg/L) 1.25 m above bottom during the summer period (15 Nov 2003 - 15 March 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

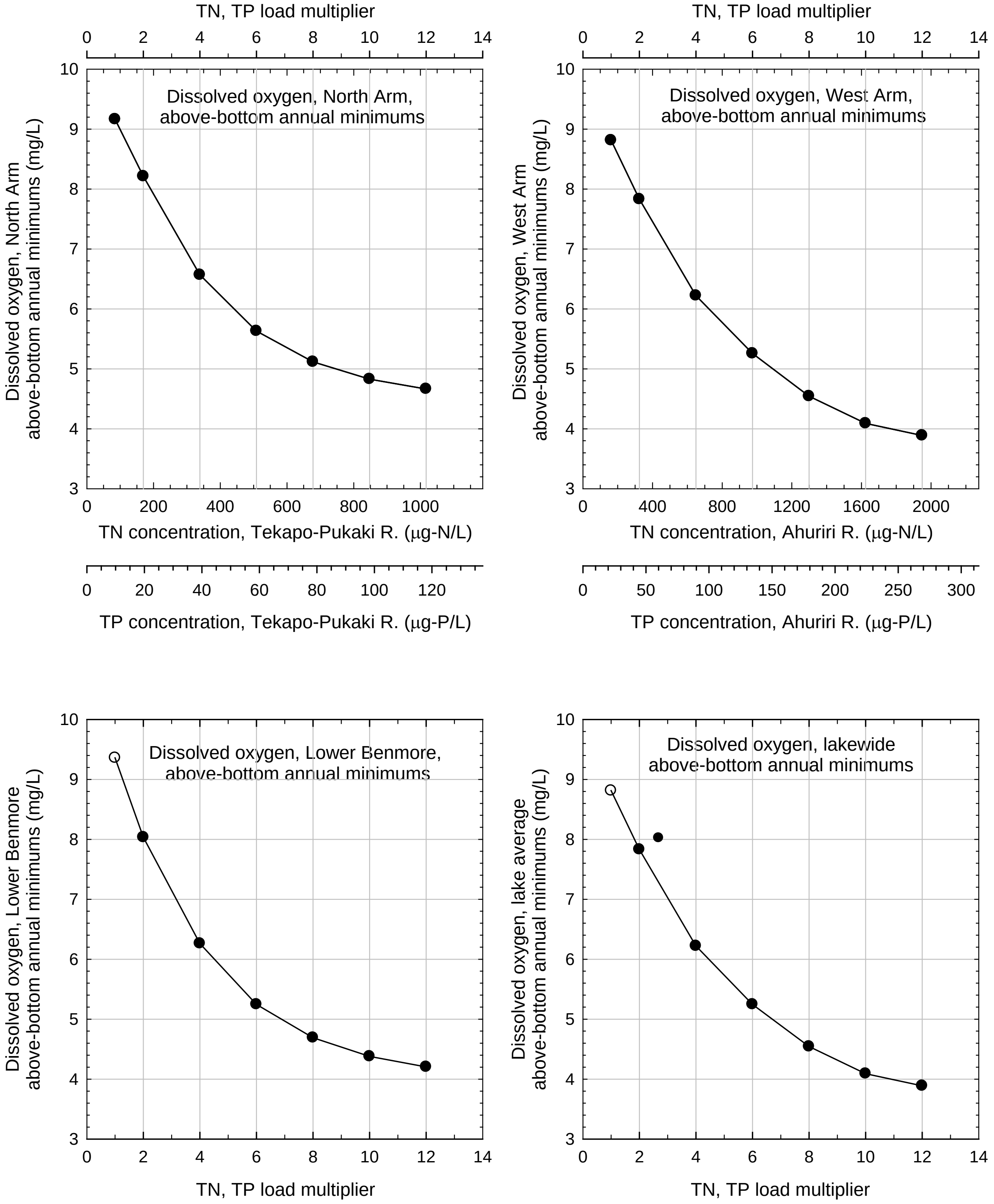

Figure 11: **Minimum dissolved oxygen concentrations (DO mg/L) 1.25 m above bottom for the year (1 Aug 2003 -31 Jul 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

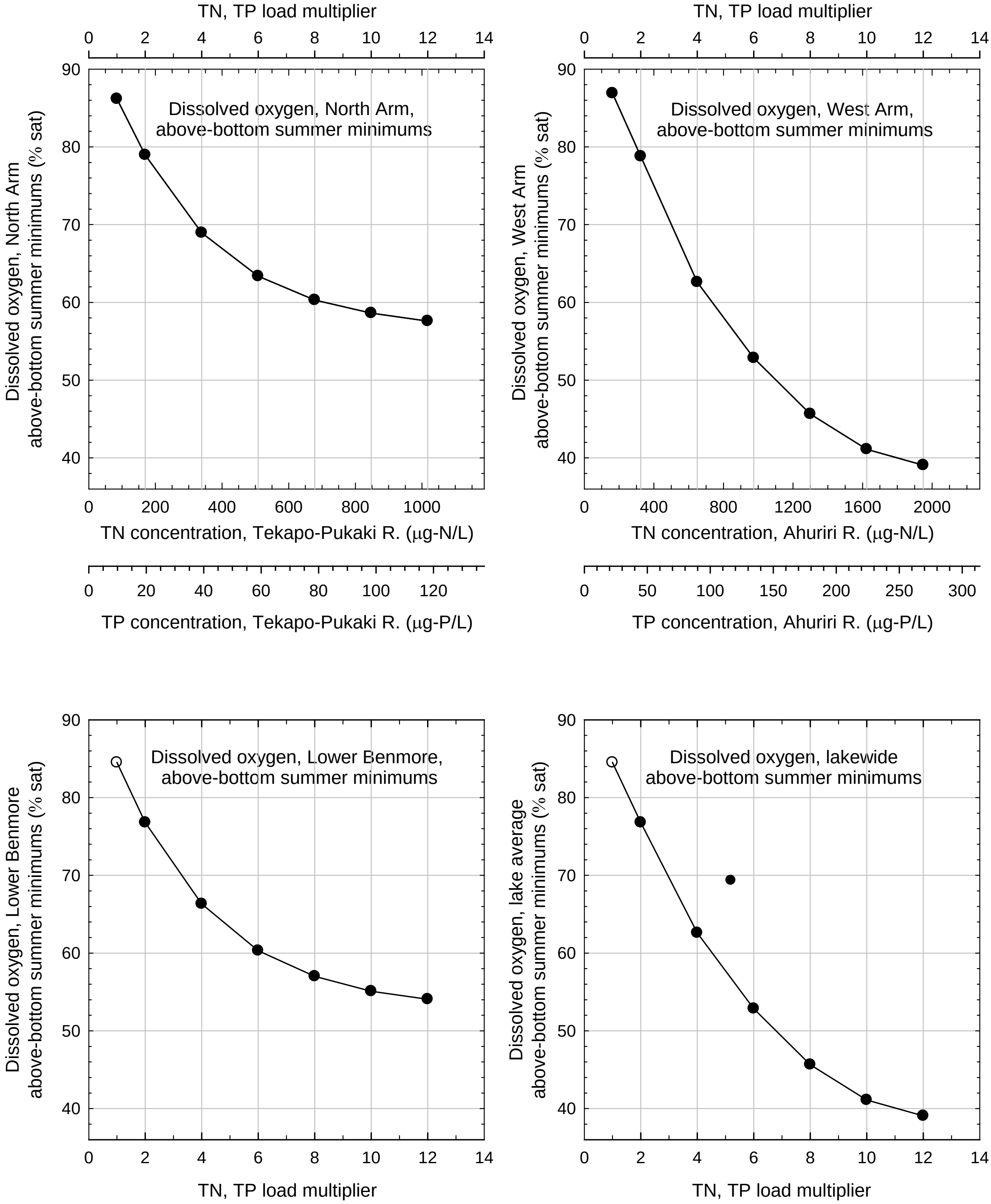

Figure 12: **Minimum percent saturation of dissolved oxygen 1.25 m above bottom during the summer period (15 Nov 2003 - 15 March 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

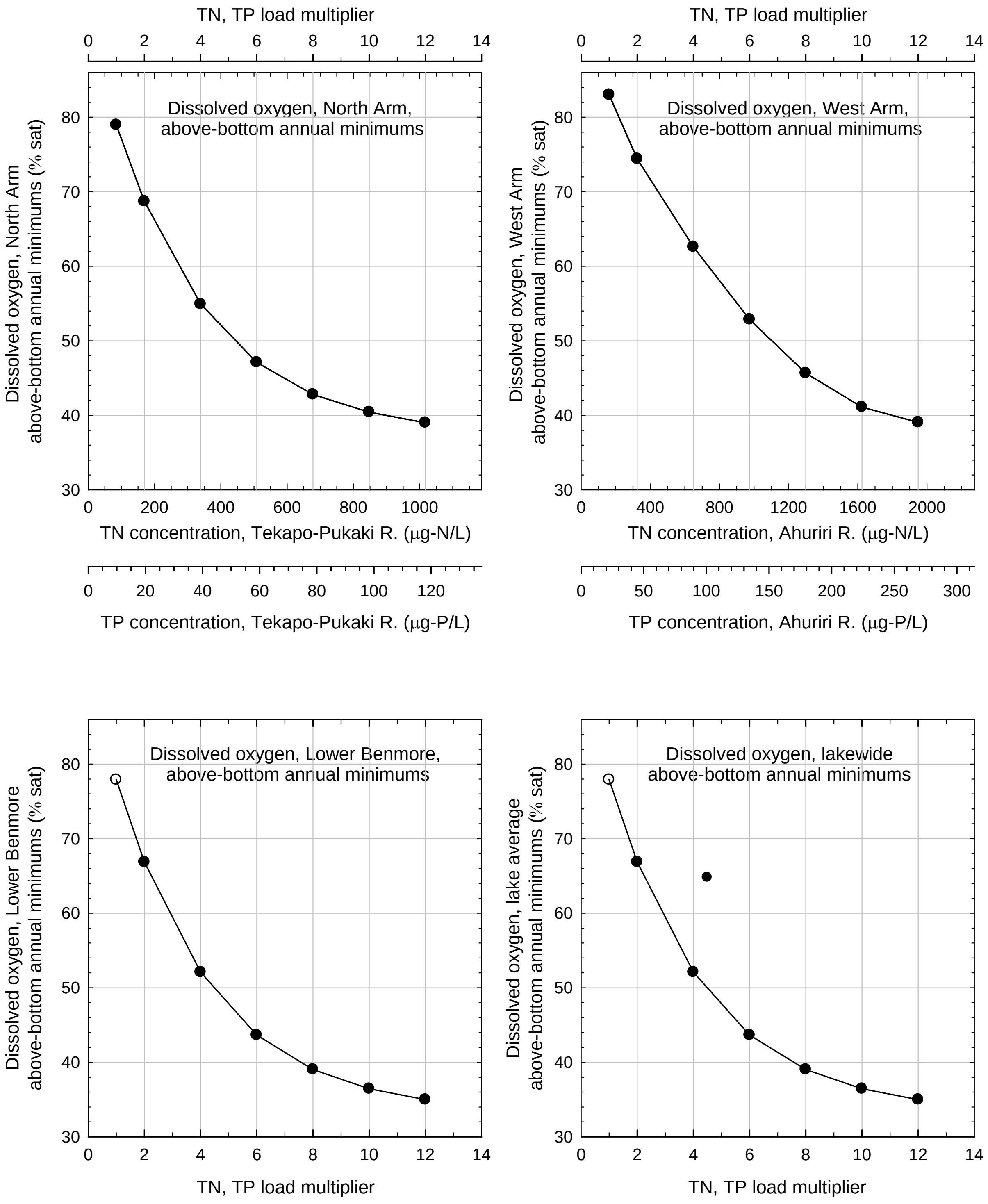

Figure 13: **Minimum percent saturation of dissolved oxygen 1.25 m above bottom for the year (1 Aug 2003 -31 Jul 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

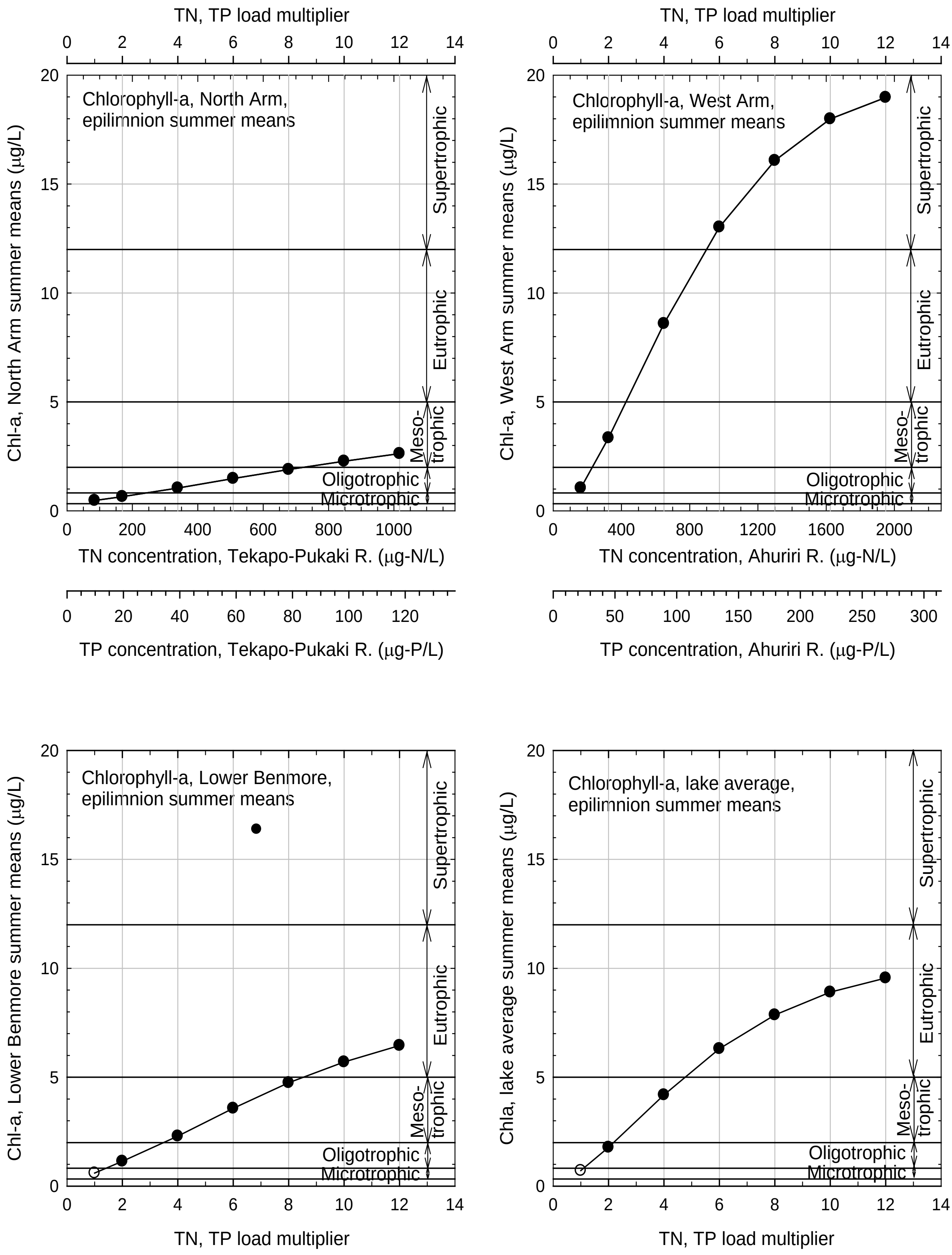

Figure 14: **Chlorophyll *a* concentrations (Chl *a* µg/L) averaged through the epilimnion during the summer period (15 Nov 2003 - 15 March 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

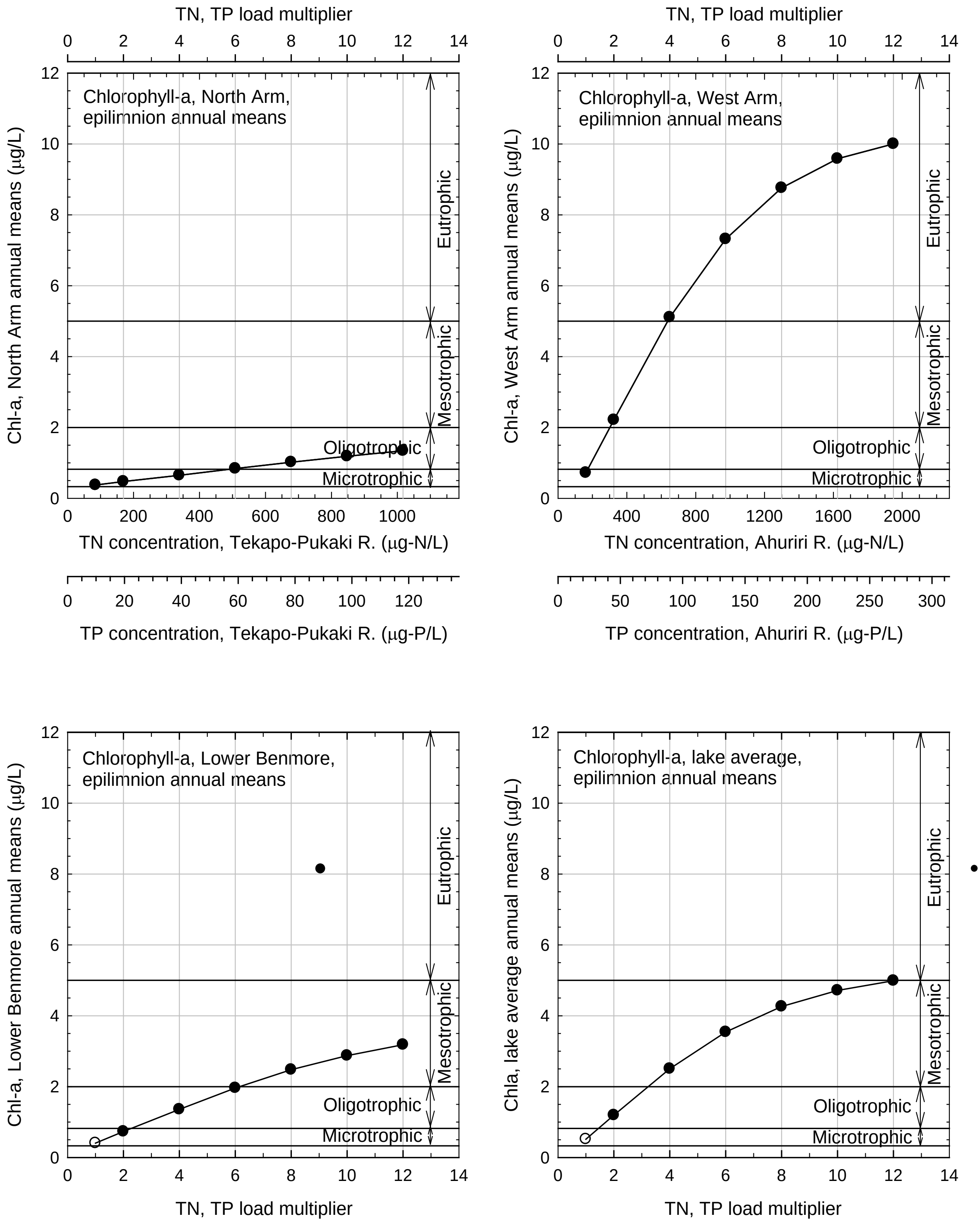

Figure 15: Chlorophyll *a* concentrations (Chl *a* µg/L) averaged through the epilimnion for the year (1 Aug 2003 -31 Jul 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).

4.4 Trophic Level Index (TLI)

While traditionally the trophic level index (TLI) is calculated using the four parameters, total nitrogen, total phosphorus, chl *a* and Secchi disk (clarity), Secchi disk was excluded from the TLI calculations for Lake Benmore. This was due to the high concentrations of glacial flour suspended in the water column in the Haldon Arm. The TLI assumes that, based on the wide range of mostly deeper lakes for which it was developed, that chl *a* is the dominant light-attenuating factor. That is not the case in Lake Benmore, where glacial flour creates a rather unique light field (Gallegos et al. 2008) and dominates light attenuation/transmission under the present oligotrophic comditions. This method of calculating TLI in Lake Benmore by excluding Secchi disk and obtaining an average for the remaining three variables is standard protocol used by Environment Canterbury (Meredith & Wilks, 2006). The ranges for each variable from which the TLI is calculated (as defined by Burns et al. 2000), excluding Secchi depth, are given in Table 1.

Table 1: Values of variables, excluding Secchi depth, which defines the boundaries of different trophic levels according to Burns et al. 2000.

Lake Type	TLI	Chl a, ug/L	TP, ug/L	TN, ug/L
Ultra-microtrophic	0 - 1	0.13 - 0.33	0.84 - 1.8	16 - 34
Microtrophic	1 - 2	0.33 - 0.82	1.8 -4.1	34 - 73
Oligotrophic	2 - 3	0.82 - 2.0	4.1 - 9.0	73 - 157
Mesotrophic	3 - 4	2.0 - 5.0	9.0 – 20	157 - 337
Eutrophic	4 - 5	5.0 - 12	20 – 43	337 - 725
Supertrophic	5 - 6	12 - 31	43 – 96	725 - 1558
Hypertrophic	6 - 7	> 31	> 96	> 1558

The TLI in the Haldon Arm ranged from <2 to 3.2 across the range of nutrient loading scenarios (i.e. existing to 12 x loading), based on annual averages (Figure 17). This represented a shift in two trophic levels, from microtrophic to mesotrophic, when nutrient loads increased to 10 x baseline. Summer mean TLI in the Haldon Arm reached mesotrophy at 6 x baseline loads (Figure 16). In the Ahuriri Arm, annual average TLI increased from 2.1 (oligotrophic), at baseline loads, to 4 (eutrophic) at 4 x baseline (Figure 17). TLI increased to supertrophic state (TLI >5) when nutrient loads increased to 10 x baseline. The summer mean TLI reached eutrophic and supertrophic levels at ~ 3 x and ~ 6 x baseline nutrient loads, respectively (Figure 16). In the Lower Benmore, TLI shifted two trophic levels from microtrophic (TLI = 1.8 at baseline) to mesotrophic (TLI = 3.3 at 6 x baseline) between 4 and 6 x baseline nutrient loads based on annual averages, and at 4 x baseline during the summer period (Figures 16-17). When calculated from lake-wide averages of chl *a*, TN and TP, the annual TLI ranged from 2 to 4.5 from baseline to 12 x nutrient loads. TLI increased to 4 (eutrophic) at 8 x baseline nutrient loads. In summer this occurred at just below 6 x loading (Figures 16-17).

4.5 Temporal variability

Plots of the modelled DO and chl *a* in the three basins of Lake Benmore over one year are shown in Figures 18-20. The period of highest chl *a* concentrations was between mid December and mid February in all three basins, regardless of the nutrient loading scenario. Epilimnion chl *a* concentrations throughout the year varied up to 5-fold in the Haldon Arm, up to 30-fold in the Ahuriri Arm and up to 12-fold in the Lower Benmore (Figures 18-20). Unlike chl *a*, periods of minimum DO were different between the three basins. In the Haldon Arm, predicted minimum DO occurred between late April and June, in the Ahuriri Arm, between February and late March, while in the Lower Benmore predicted minimum DO occurred late May to late June (Figures 18-20).

As noted at the beginning of Section 4, a further measure of variability has been calculated as the standard deviations of epilimnion chl *a*, TN and TP, and bottom DO. The results are presented as plots of coefficients of variation versus increasing scenario load in Figure F.30 Appendix F. The coefficients of variation increase with load for all four variables, but moreso for chl *a* than for TN, TP and DO, with some small differences among the three sites. The increases are rapid at first and increase more slowly for load factors $\geq$ 6. For example, the coefficient of variation for chl *a* in the Lower Benmore basin increases from 0.44 to 0.74 as the load factor increases from 1 to 4, and then from 0.81 to 0.87 as load factor increases from 6 to 12. Coefficients of variation are smaller for TN, TP and DO and increase more slowly (at the Lower Benmore site, the increases over the entire range of load factors of 1 to 12 is 0.06 to 0.22 for the coefficient of variation for epilimnion TN, 0.16 to 0.25 for epilimnion TP, and 0.08 to 0.30 for bottom DO).

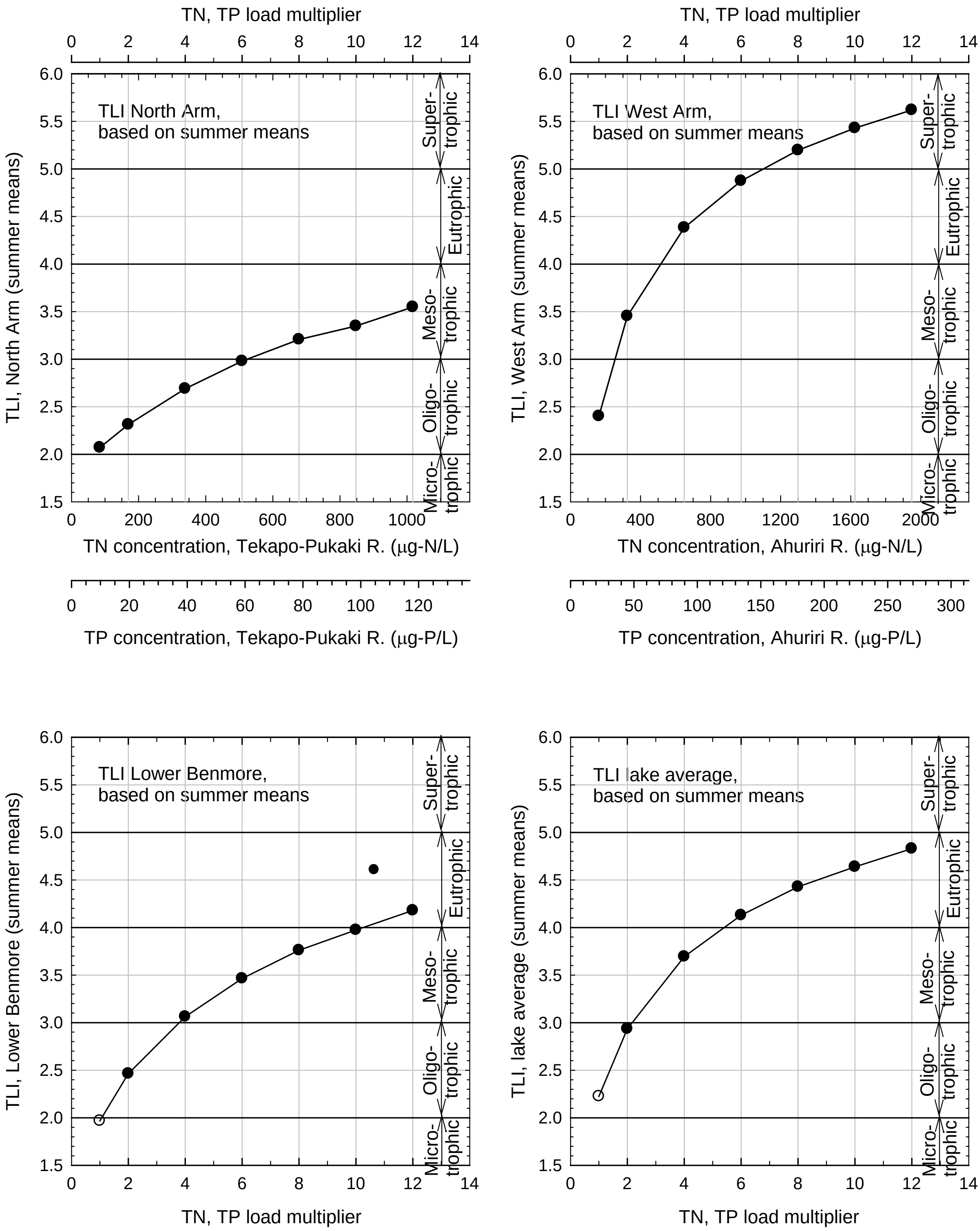

Figure 16: **Trophic Level Index (TLI) based on summer means (15 Nov 2003 - 15 March 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

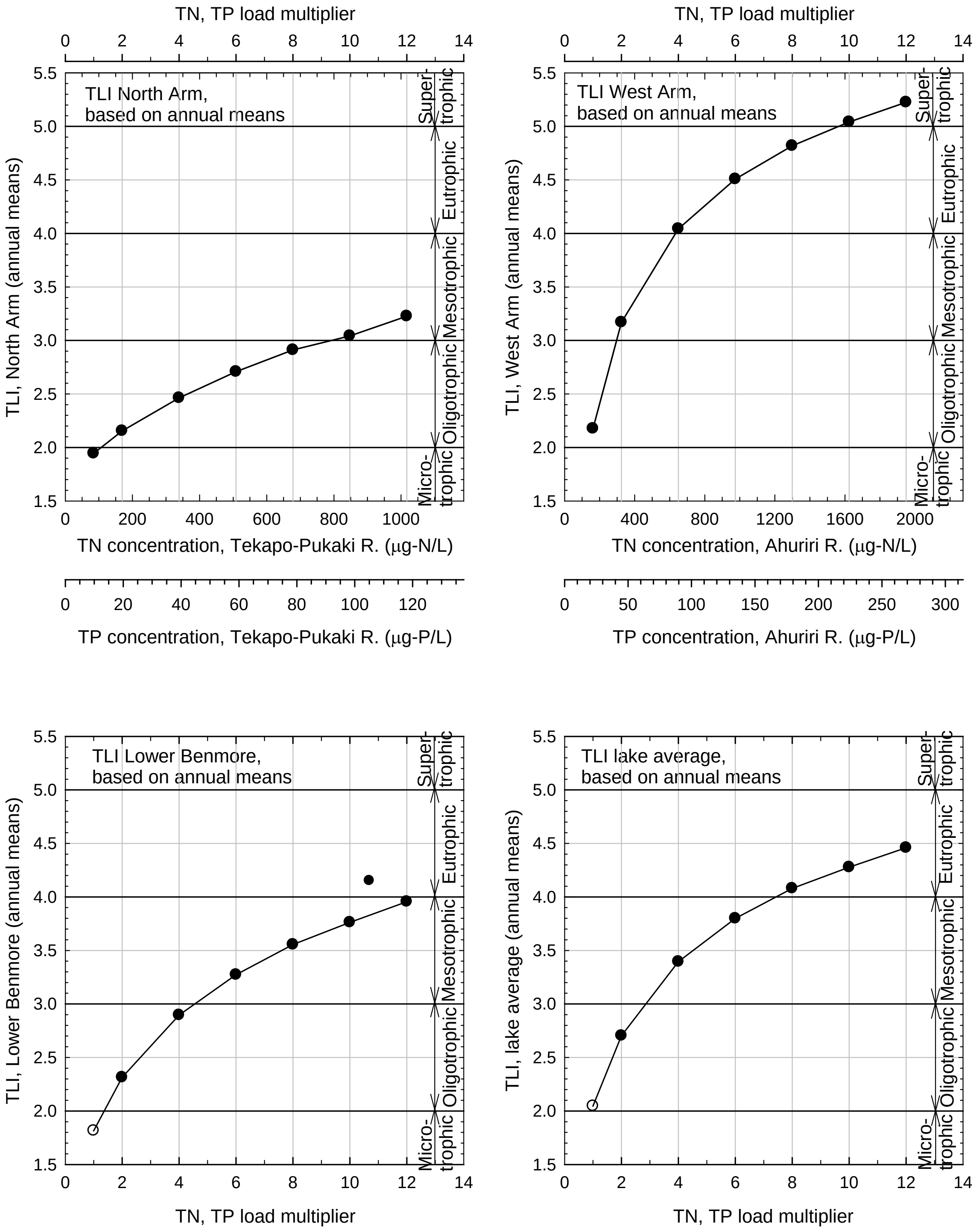

Figure 17: **Trophic Level Index (TLI) based on annual means (1 Aug 2003 -31 Jul 2004) in the Haldon (North) Arm (top left), Ahuriri (West) Arm (top right), Lower Benmore (bottom left) and the whole lake (bottom right).**

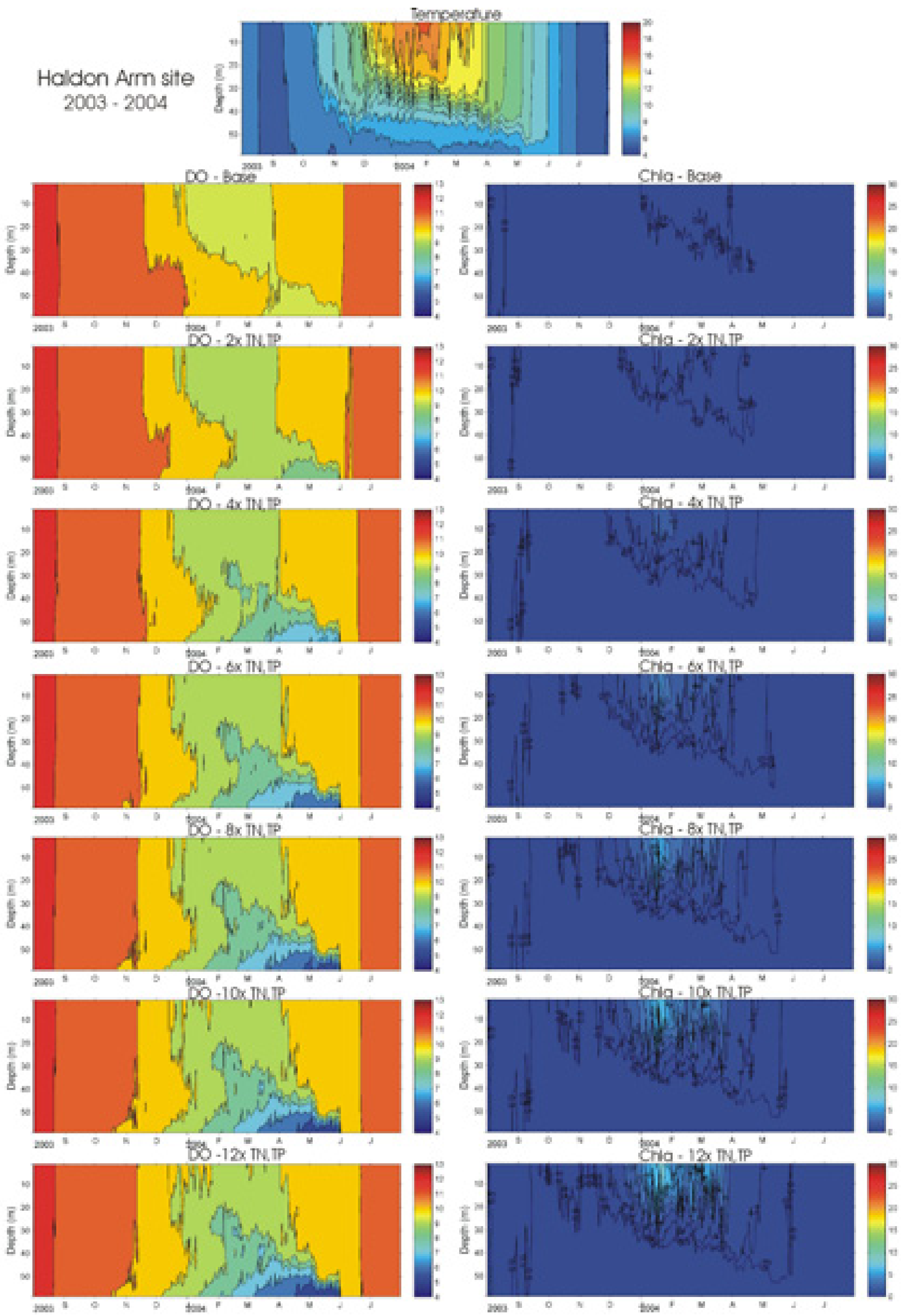

Figure 18: **Modelled contour plots of dissolved oxygen (DO mg/m^3) and chlorophyll *a* (Chl*a* mg/m^3, contour lines at 0.5, 1, 2, ..., 8 mg/m^3) from August 2003 to July 2004 under the different nutrient loading scenarios for the Haldon Arm. Temperature contour plot also included.**

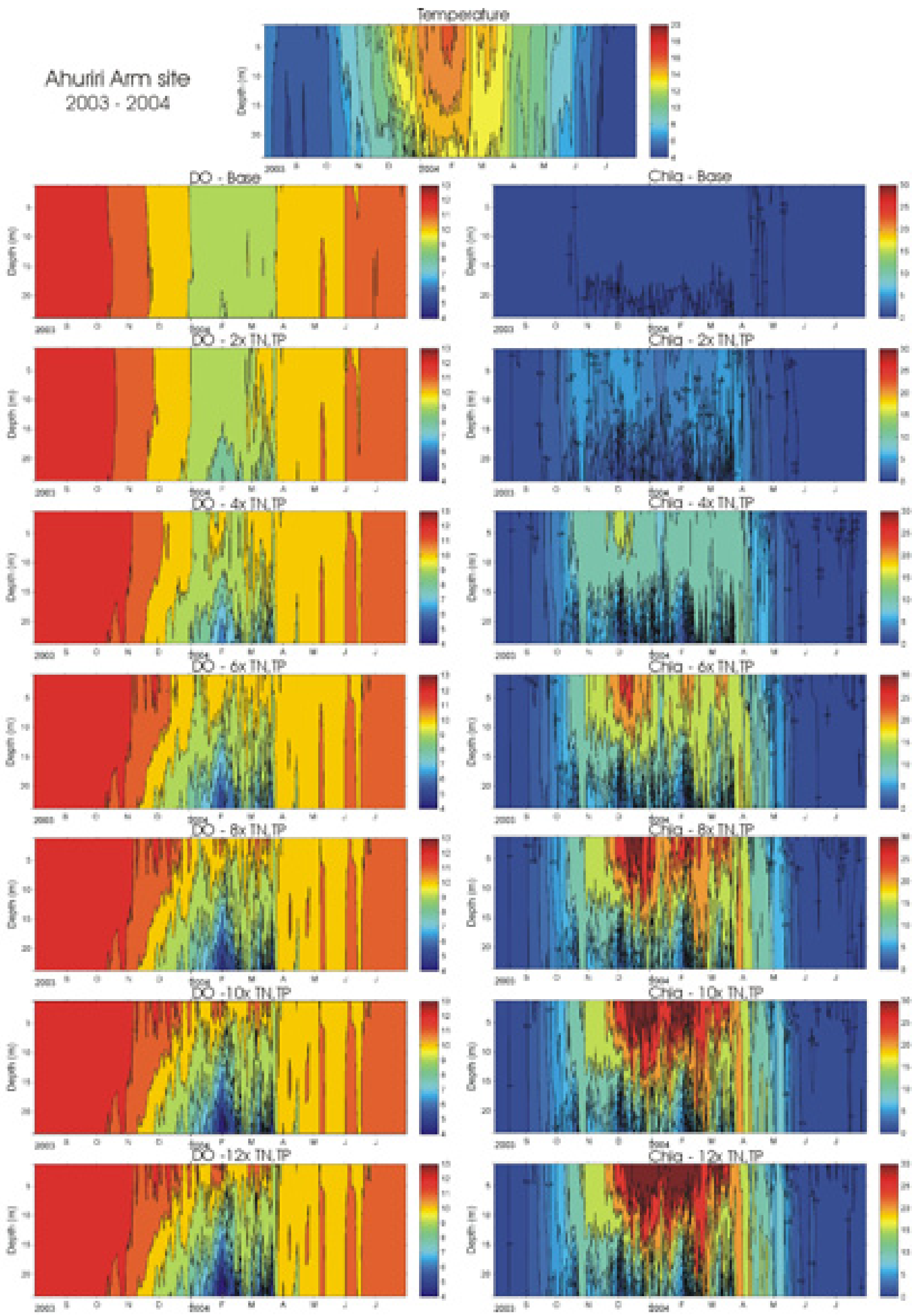

Figure 19: **Modelled contour plots of dissolved oxygen (DO mg/m^3) and chlorophyll *a* (Chla mg/m^3, contour lines at 1, 2, …, 10, 15, … , 30 mg/m^3) from August 2003 to July 2004 under the different nutrient loading scenarios for the Ahuriri Arm. Maximum predicted chlorophyll-a concentrations exceeded 30 mg/m^3 for scenarios with load factors of 8 and greater. Temperature contour plot also included.**

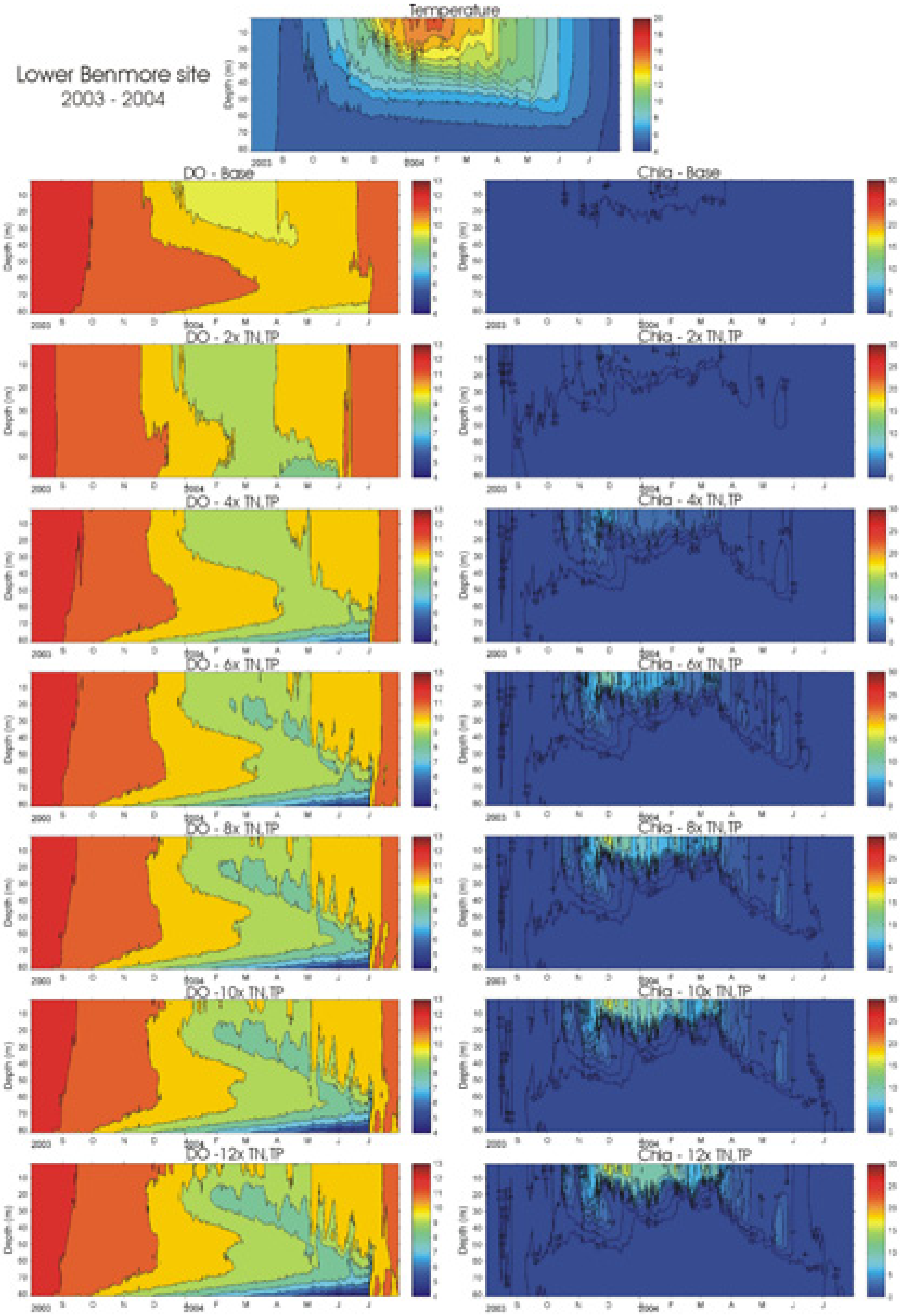

Figure 20: **Modelled contour plots of dissolved oxygen (DO mg/m^3) and chlorophyll *a* (Chl*a* mg/m^3, contour lines at 0.5, 1, 2, …, 10, 15, 20 mg/m^3) from August 2003 to July 2004 under the different nutrient loading scenarios for Lower Benmore. Temperature contour plot also included.**

5 Discussion

5.1 Model results and model uncertainty

As discussed in Section 3.4, ideally one would have enough data to calibrate the model over a full year, and preferably over a number of years that cover a range of conditions, in order to validate the calibration for a further set of years that were not included in the calibration process. That was not possible in this application because of the limited duration of the calibration data set. Nevertheless, we consider that the calibration has produced sufficiently robust results to give us confidence that the model is performing well, and that the scenario results are therefore the best that could be expected given the available data.

The profiles and time series shown in Figures C.1 – C.9 show similar qualitative agreement, by visual inspection, with similar profiles and time series from other complex hydrodynamic-ecosystem models for which results have been published in peer reviewed scientific literature for both longer studies (Omlin et al. 2001, Romero et al. 2004; both approximately 2 years) and shorter, event-based studies (Robson and Hamilton 2004, Hedger et al. 2004). However, none of these studies presents quantitative error measures for water quality parameters, or discusses separate evaluations for calibration and validation.

To provide a quantitative estimate of model errors and uncertainty, root-mean-square-errors, both absolute (RMSE) and normalised (NRMSE) by the variable sample mean, have been calculated for chl *a*, TN, TP, dissolved oxygen and temperature. These are given in Table F.1, Appendix F, and were calculated as the square root of the mean of the squared differences between modelled and measured values for eight samples collected between 8 December 2008 and 17 March 2009 (see Section 3.4, last paragraph, for explanation of two sets of samples collected in March 2009). For purposes of comparison Table F.2 gives NRMSE's extracted from published reports of four recent long-term (3 – 7 years) modelling studies (Arhonditsis and Brett 2005a,b; Burger et al 2008; Trolle et al. 2008a,b; Trolle 2009), all of which use one-dimensional (in the vertical) models and model lakes with simpler bathymetry than Lake Benmore.

The predictions of temperature, as shown in the profiles in Appendix C, Figures C.1-C.3, show excellent agreement for all three sampling sites in Lake Benmore. RMSE (Table F.1) are within the range 0.2-0.6°C (NRMSE 1.4-4.7%), except for the deepest site in the Lower Benmore basin, where a model prediction of cold temperatures in the shallow areas of the Ahuriri Arm in August results in an underflow that maintains too-cold deep bottom temperatures for the rest of the year in the Benmore basin (RMSE 0.9°C, NRMSE 13%). The predicted profiles of dissolved oxygen are similarly good (RMSE 0.3-0.6 mg/L in surface waters, NRMSE 3-7%; RMSE 0.7-1.4 mg/L in bottom waters, NRMSE 6-17%), although there is a tendency to overestimate dissolved oxygen concentrations. The match between the measured chl *a* and predicted profiles is not as close as for temperature and dissolved oxygen, but this is not unexpected given similar findings in other model applications and the complexity of the processes involved. RMSE values from the time series are 0.2-0.5 µg/L in surface waters (NRMSE 28-30%) and 0.08-0.61 µg/L in bottom waters (NRMSE 46-104%); there was no tendency to consistently overestimate or underestimate the concentrations.

There are no measured profiles for TN and TP; only the time-series plots are available to give visual assessments (Appendix C, Figures C.4 – C.9). RMSE for TN are 21-71 µg/L, NRMSE 28-67%, with a tendency for the model to underestimate concentrations at most sites and most depths. RMSE for TP are 3-19 µg/L, NRMSE 59-152%, with a tendency for the model to overestimate TP in the deeper waters at the Benmore and Haldon sites, but underestimate TP at all depths at the Ahuriri site.

Because these error measures are based on a small number of data points they are sensitive to the possibility of being biased by relatively extreme events. For example, on 21 January 2009, the sample collected from 15 m at the Ahuriri site was unusually turbid with visibly high concentrations of suspended solids. Analytical results returned very high values of TN and TP for the sample. If the result from this single sample is removed from the error analysis, the RMSE for TN at this site and depth is reduced from 54 to 31 µg/L (NRMSE from 43 to 24%) and for TP from 12 to 4 µg/L (NRMSE from 96 to 29%). In this regard it is also of interest to compare the RMSE values of all variables at all sites and depths with the naturally observed variability in measured values, a measure of which is the standard deviation. Standard deviations for the samples are shown in Table F.1, and they are of similar magnitudes to the RMSE values.

Examination of the time series for chl *a* and for nutrient species (Appendix C, Figures C.4 – C.9) also gives us reason to consider that the model is performing well in terms of predicting patterns for all these variables. The most notable discrepancy is that for dissolved organic carbon (DOC), for which the measured in-lake values are consistently higher than the modelled values. DOC concentrations were not increased in the inflows as part of the scenarios, and we deliberately avoided adjusting model parameters for in-lake DOC following the advice of Dr C Howard-Williams, who felt that further attention is needed to confirm the laboratory analyses of DOC in these waters laden with glacial flour. However, the discrepancy has only minor effects on the model's results for chl *a*, TN, TP (variables used to calculate TLI), or dissolved oxygen. The complete carbon cycle was not modelled in detail in the Lake Benmore application, because it is highly unlikely that availability of carbon limits phytoplankton production in the lake, either under present conditions or under the nutrient-enriched conditions of the scenarios (Schindler et al. 1972). Hence, there is little direct connection in the model, as configured for the Lake Benmore application, between DOC and the cycles of nitrogen and phosphorus. Moreover, the relatively low concentrations of DOC found in Lake Benmore are expected to have only minor effects on the lake's trophic state and productivity (Hanson et al., in preparation).

Our use of a complex, three-dimensional model is perhaps best justified by the very different physical, chemical and biological behaviours of the two basins, interpreted both from measured and simulated variables. At the same time, measured and modelled variables revealed hydrodynamic exchanges of water between the two basins that varied in strength and at varying depths at different times of the year; these exchanges were predicted by Pickrill and Irwin (1986) on the basis of temperature and suspended sediment data. The inherent simplifications of box models would not account for these complexities, particularly the different stratification and mixing patterns that occur within and between the basins, which have a marked effect on chemical and biological variables. It seems realistic that the predicted increases in trophic level in the Haldon Arm are small compared with those in the Ahuriri Arm as nutrient loads increase, given the flushing effects of the Ohau C Canal.

The error measures presented in Table F.1 and described above are the best estimates we can present as a guide to model uncertainty. They provide a better guide for the smaller load increases, and become less reliable at the larger load increases. They can be compared with normalised error measures from other studies as summarised in Table F.2, as noted earlier in this section. The normalised errors from the other longer-term studies are of roughly similar magnitude to those from this study, mostly smaller, but not always. Model uncertainty in general must increase at the extreme loads, and the results used with caution. At the same time, it is important to point out that while the model predictions span a wide range of trophic levels in the Ahuriri Arm, that the model does not predict dissolved oxygen concentrations in bottom waters that are sufficiently low to trigger a shift to a trophic state controlled by internal loading due to nutrient release from lake sediments. Such a shift would go beyond model limits. It is likely that flushing by water originating in the Haldon Arm and the Lower Benmore basin, as revealed in the hydrodynamic model output, assists in maintaining oxic conditions in the bottom waters of the main Ahuriri basin.

5.2 Consequences of increased nutrient loads to lake water quality

The model simulations predict that as nutrient loads into Lake Benmore increase there is degradation in the water quality, interpreted on the basis of chl *a* in the epilimnion and dissolved oxygen in the hypolimnion, as well as the TLI. The model suggests a shift of up to two levels in trophic status (i.e., from oligotrophic to eutrophic) for 4 x the baseline nutrient loads in the Ahuriri Arm; a similar shift is predicted in the Lower Benmore basin for 6 x baseline loads. The three-dimensional model simulations show strongly contrasting results between the Haldon and Ahuriri Arm for the same nutrient loads. This is most likely the result of the Haldon Arm flushing effects from the input of approximately 250 m^3/s of water from the Ohau C Canal. This water is primarily derived from lakes above the proposed development and, based on our current understandings, is not significantly impacted by increased nutrient loads in the catchment at present. However, the assumption that current catchment developments do not have a significant impact on nutrient concentrations in the Ohau C Canal or other surface waters would need to be investigated as part of any plan development process.

At the second Environment Canterbury workshop (Waitaki Lakes Water Quality: a method to define sustainable thresholds for nutrient loading for regional planning; Development Workshop 2: 21 May 09) we were informed that developments are proposed near the Wairepo Arm that would result in some nutrient runoff that eventually finds its way into Ohau C Canal. The amount was unspecified, so rather than attempt to derive separate load factors for the Ohau C Canal and the other inflows to the Haldon Arm, in the model scenarios nutrient load factors for the Haldon Arm were applied only to the

Tekapo-Pukaki Rivers and ungauged inflows (which incorporates groundwater), and no increases were applied to loads in the Ohau C Canal or the spill from Lake Tekapo and Lake Pukaki. As noted in Section 3.3, this is of no consequence in terms of assessing the effects *on the lake* of total increases in loads to the Haldon Arm. These effects will be the same in the model regardless of whether the increased loads enter the lake in the model only in the Tekapo-Pukaki River inflow and the ungauged Haldon Arm inflow, or whether some of the increase is allocated to adjacent model inflow cells for the Ohau C Canal and lake spill.

Seasonal variability is an important consideration in Lake Benmore and the dynamic model has contributed to a more complete understanding of its nature. The model clearly demonstrated the range that exists between summer maxima and winter minima of chl *a* in all three basins. This was particularly marked in the Ahuriri Arm and Lower Benmore and has an effect on the TLI for these basins. Comparisons between the summer averages and annual averages of TLI calculations showed that shifts in trophic levels were achieved at lower nutrient loads in the summer compared with averages across the year. We suggest that it is more appropriate to consider the modelled summer TLI when assessing nutrient loads as summer TLI will determine the impacts on the lake values as described in Hayward et al (2009).

Accelerated eutrophication is likely to increase the relative abundance of cyanobacteria but it may also decrease availability of light to deep-living algae which can aggregate into a layer known as the deep chlorophyll maximum (DCM) in deep, oligotrophic lakes. The DCM provides a buffer between relatively high nutrient concentrations of the hypolimnion and nutrient-depleted waters of the upper epilimnion. Disappearance of a DCM leads to conditions which favour more buoyant, surface-dwelling phytoplankton species (Pilati et al. 2003). The loss of a DCM is associated with greater biomass and horizontal variability of phytoplankton, as well as occurrence of algal blooms (Ryan et al., 2005). Transitions of this nature will increase as the TLI increases; they may be abrupt rather than gradual, signalling a regime shift, and requiring careful consideration in any proposed water quality management regime for Lake Benmore.

Grid size of the model (400 x 400 m Figure 2) was required to be sufficiently large to ensure that computational limits of the three-dimensional model did not limit the duration of simulations to less than one year. As a result, some localised effects may have been excluded. In the Haldon Arm, it has been observed during sampling visits that an area approximately 1 km^2 at the river mouth of the Tekapo-Pukaki Rivers does not mix with inflows received from the Ohau C Canal. The Tekapo-Pukaki Rivers are anticipated to receive the majority of increased nutrient loadings from the Haldon Arm catchment. It is plausible to suggest that within this 1 km^2 region algal blooms similar to the Ahuriri River inflow site could occur. This area is a popular location for fly fishing in Lake Benmore. Fish and Game report an estimated 57,000 angular days in Lake Benmore in 2007-08. There may be impacts on recreational fishing if fish begin to avoid foraging in this area as a result of algal blooms.

5.3 Comparison with proposed NRRP objectives

Numeric water quality objectives were proposed by ECan officers in a report presented at recent NRRP hearings (Hayward et al. 2009). The numeric water quality objectives proposed by Hayward et al. (2009) that would be relevant for Lake Benmore were as follows:

- Trophic Level Index (TLI) ≤ 3
- Hypolimnion dissolved oxygen (% saturation) > 70
- Epilimnion dissolved oxygen (% saturation) > 90

Modelled results presented here suggest that individual basins of the lake responded differently under the modelled scenarios. A TLI of≤ 3 in the Haldon Arm could be achievable under increased nutrient loads up to 6 x baseline (i.e., existing) concentrations, while in the Ahuriri Arm, could only be achieved if nutrient loads were kept below 1.5 x baseline loads.

5.4 Consequences for values

5.4.1. Ecological

Responses of the pelagic (open water) zone to increased nutrient loads were specifically modelled in this study for Lake Benmore. However, changes to the shoreline associated with increased nutrient loads can have complex and varied impacts on the biota and integrity of the littoral system. These changes can also occur more rapidly than detectable changes in the pelagic system. In some large, deep oligotrophic lakes, the nearshore littoral zone may provide the most important feeding and breeding habitat for organisms, in particular invertebrates and fish (e.g. Lake Waikaremoana – Howard-Williams et al. 1986). It is also the region subject to greatest recreational use and pressure, and is where macrophytes can proliferate.

Filamentous green algal growth has been associated with elevated nutrients in lakes in the USA (Jacoby et al 1991; Rosenberger et al, 2008; Lambert et al 2008) and along nutrient gradients in Danish lakes (Liboriussen & Jeppensen 2006). Changes in species composition to less desirable species, such as filamentous green algae and didymo are also likely to occur in Lake Benmore around shorelines in response to increased nutrient loads. Basic knowledge of didymo ecology and growth dynamics in New Zealand lakes is lacking. However, didymo is presently growing extensively along the shorelines of some South Island lakes, including Lake Ohau (Sutherland and Edwards 2009) and based on observations in New Zealand rivers and some overseas literature, didymo appears to perform exceptionally well with nutrient additions (Larned et al. 2007, Norton & Sorrell 2006, Kawecka & Sanecki 2003).

Presently, the only problematic weed in the Waitaki lakes is lagarosiphon (*Lagarosiphon major*), which forms dense stands in the Ahuriri Arm. Lagarosiphon biomass is likely to increase under moderate increased nutrient conditions. However, proliferation of phytoplankton blooms and epiphyte algae (attached to plants) will reduce the available light and therefore reduce growth rate and biomass of the plant. The more aggressively growing weeds egeria and hornwort grow well under moderate and high nutrient concentrations. Egeria and hornwort are two of New Zealand's worst aquatic weeds in terms of invasiveness, competitive ability and production of biomass, and have been shown to impact on hydropower stations in the Waikato system (Johnstone and Harding 1987). Both species are present in the South Island, but not yet in Lake Benmore, and do not appear to be limited by the lower South Island temperatures. As nutrient concentrations increase, the Waitaki lakes would become more favourable for the establishment of both egeria and hornwort.

The development of dense beds of aquatic weeds in conditions of nutrient enrichment would be likely to have the following effects:

- Increased likelihood of localised anoxia in the littoral zone.
- Increased recycling rates of nitrogen and phosphorus.
- Increased likelihood of macrophyte biomass uprooting in strong winds, and being washed up on beaches or affecting power intake structures.
- Decreased recreational opportunities in lake littoral zones.

5.4.2. Recreational and Aesthetic values

The value of water quality can also be defined as the suitability of water to sustain various uses or processes (Meybeck et al. 1996). Public perception of water quality is often based on observations of water clarity. Low water clarity is usually regarded by the public as indicative of poor water quality regardless of other water quality variables. Lake Benmore is used extensively for recreational purposes by both New Zealanders and tourists and the aesthetic values of this environment are highly regarded (Sutherland-Downing & Elley 2004). Degradation in water clarity, through increased phytoplankton growth, will impact on these values. A visual representation of increased chl *a* concentrations is given in Figure 21. For the purpose of recreational contact and maintenance of a high aesthetic value, the public is more likely to perceive a 'healthy' lake when chl *a* concentrations are low and the water does not appear green.

The general public has become increasingly aware of the risks associated with algal blooms. The frequent dominance of cyanobacteria in eutrophic waters is of additional concern to water quality considerations because several of these organisms can produce toxins (see MfE 2009b).

5.4.3. Hydro Power Generation

Large amounts of macrophyte and didymo drift material could impinge on the water intakes causing impedance of water flow to the turbines and incurring costs for removal and losses in generation output. Additional generation losses that could be caused by the material include:

- Extra water that must be passed through turbines to maintain generation when screens are partially blocked.
- Loss of water spilled to maintain flows to downstream systems when a station is by-passed for screen cleaning.
- Blockages that lead to partial and complete outages.

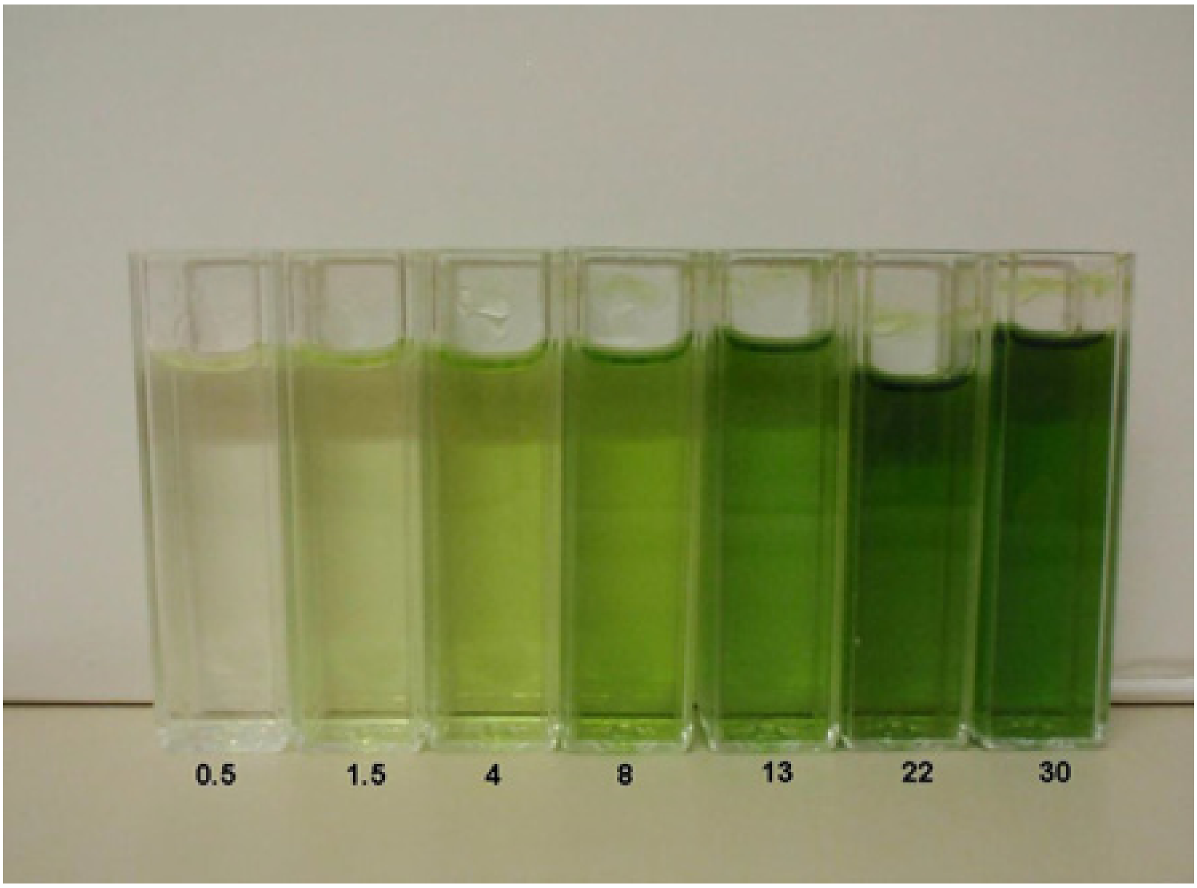

Figure 21: Visual display of extracted chlorophyll at concentrations (µg/L) ranging from current baseline average lake-wide for Lake Benmore (0.5 µg/L) to the near maximum peak summer chlorophyll value predicted in the Ahuriri Arm (30 µg/L). See Figures 18-20 for chl *a* predictions as a function of nutrient loadings over time.

Volvox aureus is a colonial green alga composed of numerous flagellate cells embedded in a hollow sphere comprised of a gelatinous extracellular matrix. It is commonly found amongst the phytoplankton assemblage of Lake Benmore. Blooms of this alga have, during the past 6 years, caused nuisance blockages in cooling water systems at the Benmore generator units. This has occurred in the six main generator units and the two auxiliary units at the same time, but with minor operational impact (M. Preston, Meridian, pers. comm). *Volvox* spp. respond to nutrient-rich conditions in stagnating water columns and are, therefore more frequent in small eutrophic lakes and during very stable phases in large river-fed basins and storage reservoirs (Reynolds et al 2002). Under increased nutrient loads, increase growth and prevalence of *V. aureus* and other mucilage producing algae and bacteria could potentially to lead to further operational issues for the Benmore Power Station.

5.5 On-going monitoring

On-going monitoring of the water quality in the inflows and the three basins of Lake Benmore will provide additional data to improve calibration and allow for independent validation of the model predictions if continued long-term. It would improve the statistical reliability of the error estimates of model predictions and would permit an assessment of intra- and inter-annual variability in water quality parameters. When evaluating the outcomes of this report there will be a need to make pragmatic decisions based on the best information available at the time and, importantly, manage unavoidable uncertainty in a risk-based manner. This is supported by providing for on-going monitoring, and for management flexibility to respond on the basis of the outcome of that monitoring.

6 Conclusions

We have used a coupled three-dimensional hydrodynamic-ecosystem model, ELCOM-CAEDYM, to predict changes in in-lake concentrations of chlorophyll-*a*, total nitrogen, total phosphorus and dissolved oxygen, and consequently in lake trophic state, that could occur from increases in nitrogen and phosphorus loads in the rivers entering Lake Benmore. We have presented errors between modelled and measured in-lake variables for the limited sample data available, to provide an estimate of model uncertainty, and have presented error measures from other similar model applications for comparison. Unfortunately we cannot point to a definitive publication that specifies a certain level of agreement between modelled and measured variables that is “acceptable”. A common theme that emerges from the growing body of literature that discusses the calibration, validation and assessment of uncertainty for complex biogeochemical models such as the ones used in this study, is that there are no universally accepted answers or standards for the questions surrounding these issues, that deciding what level of performance is acceptable entails some subjectivity, and that this decision must ultimately depend on the use that will be made of the model results (Robson et al. 2008, Arhonditsis and Brett 2005, Oreskes 1998, among others). A strength of the ELCOM-CAEDYM model is that it permits an understanding of the Lake Benmore ecosystem and allows us to make predictions of responses to change, in this case, increased nutrient loads.

The model simulations carried out in this study have demonstrated that the two main basins of Lake Benmore behave differently in response to increased nutrient loads; the model also showed linkages between the two Arms through water exchange (as predicted by Pickrill and Irwin, 1986, and noted in Section 5.1). The Ahuriri (West) Arm appears likely to be more sensitive to nutrient increases than the Haldon (North) Arm. There are two main reasons for this. First, the relatively large inflow of the Ohau Canal (c. 250 m^3/s) compared to Ahuriri inflow (c. 28 m^3/s) is presently little affected by catchment development and provides relatively larger dilution of nutrient loads than inflows to the Ahuriri Arm. Second, the longer residence time (75 days) in the Ahuriri Arm compared to the Haldon Arm (57 days) makes the Ahuriri Arm more susceptible to algal growth responses to increased nutrients.

In the model scenarios, nutrient load factors for the Haldon Arm were applied only to the Tekapo-Pukaki Rivers and ungauged inflows (which incorporates groundwater), and no increases were applied to loads in the Ohau C Canal or the spill from Lake Tekapo and Lake Pukaki. As noted in Section 3.3, this is of no consequence in terms of assessing the effects *on the lake* of total increases in loads to the Haldon Arm. These effects will be the same in the model regardless of whether the increased loads are allocated to only the Tekapo-Pukaki River inflow and the ungauged Haldon Arm inflow, or whether some of the increase is allocated to the Ohau C Canal and lake spill.

Because the Haldon and Ahuriri Arms behave differently, we recommend that they be considered separately in terms of setting management objectives and nutrient load caps. We suggest that ECan could set the same objectives (i.e. TLI value) for both arms or could choose different objectives for each arm. Under either option, the modelling results show that it would be appropriate to apply a separate set of nutrient load caps for each Arm. The Haldon Arm is likely to be able to tolerate a greater nutrient load than the Ahuriri Arm and therefore the amount of land-use development that is appropriate may differ between the two sub-catchments. However, if a decision is made to set a higher load cap for the Haldon Arm than for the Ahuriri Arm, it would be prudent to rerun the model with the specified loads, to check that hydrodynamic interactions between the Arms do not lead to changes in water quality that alter intended TLI outcomes.

Because the Ahuriri Arm appears to be more sensitive than the Haldon Arm, water quality objectives for both Arms based on the Ahuriri Arm would provide a high level of management for the whole lake and lakes further downstream. However, as suggested above, if a higher load cap is set for the Haldon Arm than for the Ahuriri Arm, it would be prudent to rerun the model with the specified loads to check that trophic level objectives would still be met in the Lower Benmore basin.

The two key outputs of this study are provided in Tables 2 and 3, which provide options for ECan to consider for:

(a) Measurable objectives for water quality in Lake Benmore in terms of TLI; and

(b) Nutrient load caps associated with achieving each of the objective options.

Table 3 is arranged so that options for objectives (i.e. TLI values) are given in the first column and these are linked to a description of the measurable environmental state that would be achieved for each option in the subsequent columns. Once an objective (TLI value) has been chosen, the related nutrient load caps (total nitrogen (TN) and total phosphorus (TP)) can be read from the table. We note that the TLI is a continuous scale and so any decimal number could be chosen (e.g., 2.9, 3, 3.1, 3.25, etc.) and the associated TN and TP caps calculated for that TLI value. For pragmatic reasons a selection of discrete TLI options is provided in Table 3. Generally, the higher the TLI number the greater the risk of associated degradation of environmental values.

It is not the purpose of this report to recommend any particular objective because that decision is ECan's responsibility and would be developed in the context of the RMA planning provisions. However several options and issues should be considered, as outlined below:

(i) One option for ECan would be to choose an objective TLI value of 3, as proposed by Hayward et al. (2009) for all Canterbury lakes in the "Artificial Lakes" management unit (which includes Lake Benmore) in the PNRRP. If this option is taken, the associated nutrient load caps that would apply are; for the Haldon Arm 1219 tonnes/year TN and 134 tonnes/year TP; and for the Ahuriri Arm 272 tonnes/year TN and 38 tonnes/year TP. These loads correspond to 6.2 x and 1.6 x the current loads calculated from measured concentrations in the inflows to the Haldon Arm and Ahuriri Arms respectively.

(ii) ECan could consider choosing an objective TLI value that is lower or higher than 3 (i.e. more or less environmentally conservative and respectively less or more enabling of land development). Tables 2 and 3 provide the associated nutrient load caps that would apply for other objectives. However the modelled scenarios, based on the nutrient loads applied as multiplicative factors of the current baseline, only allow consideration of results for TLI values in the range 2 to 3.5 for the Haldon Arm and 2.4 to 5.5 for the Ahuriri Arm.

(iii) Another option for ECan would be to use the existing PNRRP Objective WQL1.2(3)(d) which states that "*the average annual phytoplankton biomass does not exceed five milligrams of chlorophyll a per cubic metre of lake water.* [5 mg/m^3 = 5 µg/L]" This objective corresponds to managing the lake at the boundary between a mesotrophic and eutrophic state (see Table 1, Section 4.4). To achieve this objective the associated nutrient load caps can be identified using Figure 15 (annual average) and/or Figure 14 (summer average) to determine the appropriate load multiplier, then using Tables E8 (Haldon Arm) and E12 (Ahuriri Arm) to convert the multiplier into loads. To achieve 5 mg/m^3 chlorophyll *a* as an annual average for the Ahuriri Arm, Figure 15 (top right plot) shows that loads should be capped at 4 x the current loads – Table E12 shows that this corresponds to load caps of 693 tonnes/year TN and 96 tonnes/year TP in the Ahuriri Arm. Nutrient loads to the Haldon Arm could be increased by 12-fold and still achieve this objective as an annual average (Figure 15). To achieve 5 mg/m^3 chlorophyll *a* as a summer average, Figure 14 shows that loads should be capped at 2.5 x the current loads for the Ahuriri Arm. Table E12 shows that this corresponds to load caps of 433 tonnes/year TN and 60 tonnes/year TP for the Ahuriri Arm. Nutrient loads to the Haldon Arm could be increased by 12-fold, as for annual average chlorophyll *a* described above, and still achieve this objective.

(iv) The current measured nutrient concentrations in lake inflows may not fully reflect all existing irrigation development in the catchment because some development is very recent and there can be time lags for the transport of nutrients from land to lake. There can also be time lags associated with the changes in soil structure that occur over time when previously un-irrigated land is irrigated and landuse intensified. It is possible that nutrient losses from recently developed land will change with time and may increase sometime in the future. This has not been considered in the model used for this report and the response of nutrient loads to land use intensification could vary considerably from those assumed in this study. We reiterate comments made in Section 2.1 that this report provides options for ECan to define the 'total size of the pie' to be allocated (i.e., nutrient load caps). For the load caps to be useful, it will be necessary to define how much of the pie is already allocated to existing uses and to predict nutrient loads from future proposed activities. This will require an assessment of existing activities including recent land use changes. While this report has identified the measured loads of nitrogen and phosphorus currently entering the lake (i.e., natural loads plus existing activities), time lag effects will need to be considered when using models to predict the loads from recent and proposed future land use changes.

(v) While lakes tend to respond to nutrient increases in a non-linear way (i.e. as nutrient loads increase, little change may be observed initially in a lake, but at some tipping point large changes in trophic status can occur rapidly) identifying these thresholds or tipping points is very difficult. The modelling results presented graphically show a smoothed increasing response to increased nutrients but this is unlikely to be the case in reality. It is therefore important to appreciate that the higher the TLI value selected for an objective (and thus the greater nutrient load allowed) the greater the risk of rapid shift to a higher trophic status and associated degradation of values.

(vi) The results presented in this study should not necessarily be interpreted as providing a basis for the hypothesis that there is a linear response of lake trophic status to increases in nutrient loads. There are some identifiable thresholds for ecosystem effects that relate to dissolved oxygen. These are:

- Dissolved oxygen is important for healthy aquatic life including invertebrates and fish. Hayward et al. (2009) recommend a minimum threshold of 70% DO saturation for all Canterbury lakes in the "Artificial Lakes" management unit (which includes Lake Benmore). Schedule WQL1 of the PNRRP specifies a minimum water quality standard for class "Artificial" lakes of 80 % DO saturation. Schedule 3 of the RMA also defines a minimum standard of 80% DO saturation for the following water quality classes; AE (water managed for aquatic ecosystem purposes), Class F (fishery purposes) and Class FS (fish spawning purposes). Table 3 shows that; for the Haldon Arm minimum DO could fall below 80% at a TLI somewhere between 2.5 and 3 (and greater); and for the Ahuriri Arm minimum DO could fall below 80% at a TLI somewhere between 3.5 and 4 (and greater).

- Dissolved oxygen is also important at the lake bottom because nutrient releases from lake-bed sediments are greatly enhanced if conditions become anoxic (i.e. as DO approaches zero). This in turn can greatly accelerate eutrophication effects. Figure 13 shows that for both the Haldon and Ahuriri Arms the minimum DO saturation above the lake-bed was about 39% even under the highest nutrient loading scenario (12 x existing loads). Therefore the risk of anoxia is low and unlikely to be a critical factor in ECan's decision.

7 Acknowledgements

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Table 2: Summary of the modelled summer TLI ranges with associated nutrient load caps for total nitrogen (TN, tonnes N/year) and total phosphorus (TP, tonnes P/year) according to the model output for the Haldon and Ahuriri Arms. For purposes of specifying load caps in the table, the maximum load for a given trophic state corresponds to 0.1 TLI level below the boundary specified by Burns et al. (2000) for classification of trophic states (see Table 1, Section 4.4 in this report), or the maximum value modelled in this study. The minimum TLI shown for each range is the smaller of the trophic level boundary of Burns et al. (2000), or the lowest TLI modelled in this study; the baseline (existing) TLI values and loads are shown in blue.

	Haldon Arm			Ahuriri Arm		
		nutrient loads** (tonnes/yr)			nutrient loads** (tonnes/yr)	
Trophic State	**TLI***	**TN**	**TP**	**TLI ***	**TN**	**TP**
Oligotrophic	2.1 – 2.9	646 - 1140	68 - 125	2.4 – 2.9	173 - 256	24 - 35
Mesotrophic	3 – 3.6	1220 - 1860	134 - 209	3 – 3.9	272 -514	38 - 71
Eutrophic	***	***	***	4 – 4.9	551 - 1070	76 - 148
Supertrophic	***	***	***	5 – 5.6	1180 - 2080	162 - 287

Notes:

* - The TLI is a continuous scale; any decimal number could be chosen (e.g., 2.9, 3, 3.1, 3.25, etc.) and the associated TN and TP caps calculated for that TLI value. A selection of discrete TLI options is provided in Table 3, Section 6.

** - Total loads excluding aerial deposition; includes Ahuriri River + ungauged flows for the Ahuriri Arm (Table E.12, Appendix E), and Ohau C Canal + Tekapo-Pukaki Rivers (including spill from Lakes Tekapo and Pukaki) + ungauged flows for the Haldon Arm (Table E.8, Appendix E).

*** - Trophic state beyond model range.

Table 3. **Summary of relationships between options for TLI objectives and associated nutrient load caps for nitrogen and phosphorus in the Haldon and Ahuriri Arm. Nutrient load caps for each TLI value were obtained from model output plots for summer mean values (Figure 16) and converted to annual loads using Tables E.8 & 12. Loads are totals to each Arm (excluding aerial deposition): Ahuriri River + ungauged inflows, to Ahuriri Arm; Ohau C Canal + Tekapo-Pukaki Rivers (including spill from Lakes Tekapo and Pukaki) + ungauged inflow, to Haldon Arm. Dissolved oxygen (concentration and % saturation) was obtained from model output plots for hypolimnion annual minimums (Figures 7 and 9). * = trophic level below the boundary of modelled scenarios; ** = trophic level higher than the boundary of modelled scenarios. 1x, 2x etc refers to multiples of the current nutrient loads measured in lake inflows. Existing baseline nutrient loads and trophic state are highlighted in blue. Colour bar behind the TLI scale indicates extracted chl *a* as shown in Figure 21.**

General description of environmental characteristics for the specified trophic state	Options for Objective (TLI)	Haldon Arm Annual minimum DO (hypolimnion)		Haldon Arm Associated nutrient load caps for N & P (tonnes/yr)	Ahuriri Arm Annual minimum DO (hypolimnion)		Ahuriri Arm Associated nutrient load caps for N & P (tonnes/yr)
Microtrophic Clear water (visually appealing) No risk of visual phytoplankton (e.g. green colour) Low-moderate periphyton on bed & margins Healthy macrophyte beds No risk of toxic blooms Healthy invertebrate & fish communities High biodiversity value Very high contact recreation value Very high amenity value	1	>9.8mg/L	>90%	TN = * TP = *	>9.3 mg/L	>90%	TN = * TP = *
	1.5	>9.8mg/L	>90%	TN = * TP = *	>9.3 mg/L	>90%	TN = * TP = *
	1.9	>9.8mg/L	>90%	TN = * TP = *	>9.3 mg/L	>90%	TN = * TP = *

Oligotrophic Clear water (visually appealing) Very low risk visual phytoplankton (e.g. green colour) Low-moderate periphyton on bed & margins Healthy macrophyte beds No risk of toxic blooms Healthy invertebrate & fish communities High biodiversity value High contact recreation value High amenity value	**2.0**	>9.8 mg/L	>90%	TN = * TP = *	>9.3 mg/L	>90%	TN = * TP = *
	2.1	9.8 mg/L	90%	1.00xTN = 646.3 1.00xTP = 67.63	>9.3 mg/L	>90%	TN = * TP = *
	2.4	9.1 mg/L	84%	2.47xTN = 808 2.47xTP = 86.4	9.3 mg/L	89%	1.00xTN = 173.3 1.00xTP = 23.92
	2.5	9.0 mg/L	82%	2.99xTN = 866 2.99xTP = 93.2	9.1 mg/L	87%	1.09xTN = 190 1.09xTP = 26.2
	2.9	8.2 mg/L	75%	5.45xTN = 1138 5.45xTP = 125	9.0 mg/L	85%	1.47xTN = 256 1.47xTP = 35.3
Mesotrophic Clear tending green water (variable appeal) Moderate risk of phytoplankton blooms Moderate periphyton on bed & margins Increase stress to macrophyte beds Potential shift to phytoplankton dominated community Some risk of toxic blooms Increased invertebrate & fish productivity Good biodiversity value Good contact recreation value Good amenity value	**3.0**	8.1 mg/L	75%	6.18xTN = 1219 6.18xTP = 134	8.7 mg/L	85%	1.57xTN = 272 1.57xTP = 37.6
	3.5	7.8 mg/L	71%	11.5xTN = 1808 11.5xTP = 202	8.5 mg/L	81%	2.10xTN = 364 2.10xTP = 50.3
	3.6	7.7 mg/L	71%	12.0xTN = 1860 12.0xTP = 208	8.5 mg/L	80%	2.21xTN = 382 2.21xTP = 52.8
	3.9	**	**	TN = ** TP = **	8.0 mg/L	75%	2.96xTN = 513 2.96xTP = 70.9

<u>Eutrophic</u> Turbid green water (visually unappealing) High risk of sustained phytoplankton blooms Low-moderate periphyton on bed & margins High risk of macrophyte bed collapse Likely phytoplankton dominated system Moderate risk of toxic blooms Shifts to invertebrate & fish community composition Compromised biodiversity value Compromised contact recreation value Compromised amenity value	**4**	**	**	TN = ** TP = **	7.9 mg/L	75%	3.18xTN = 551 3.18xTP = 76.0
	4.5	**	**	TN = ** TP = **	7.3 mg/L	70%	4.48xTN = 776 4.48xTP = 107
	4.9	**	**	TN = ** TP = **	6.8 mg/L	63%	6.17xTN = 1070 6.17xTP = 148
<u>Supertrophic</u> Turbid green water (visually unappealing) Sustained phytoplankton blooms High risk of macrophyte bed collapse Likely phytoplankton dominated system High risk of toxic blooms Shifts to invertebrate & fish community composition Compromised biodiversity value Poor contact recreation value Poor amenity value	**5**	**	**	TN = ** TP = **	6.6 mg/L	63%	6.79xTN = 1180 6.79xTP = 162
	5.5	**	**	TN = ** TP = **	6.0 mg/L	57%	10.74xTN = 1860 10.74xTP = 257
	5.6	**	**	TN = ** TP = **	5.9 mg/L	56%	12.0xTN = 2080 12.0xTP = 287
	5.9	**	**	TN = ** TP = **	**	**	TN = ** TP = **

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