Gippsland Lakes Environmental Study

Integrated Model Development and Calibration

Technical Report

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Executive Summary

An integrated model of nutrient and sediment cycling, phytoplankton dynamics, sediment biogeochemistry and benthic plants has been developed and implemented for the Gippsland Lakes. The model has been designed to improve understanding of the function of the Lakes, and their response to catchment loads, and to allow managers to assess management scenarios. This report describes the formulation, implementation and calibration of the model, and the implications for system state and function. Management scenarios are presented in a separate report.

Physically, the model divides the Gippsland Lakes into 8 basins or sub-basins: L. Wellington, western, central and eastern L. Victoria, northern and southern L. King, Jones Bay and Reeves Channel. Two vertical layers represent L. Victoria, L. King and Reeves Channel, which are usually stratified.

The model is forced by physical exchanges derived from a high resolution hydrodynamic model of the Gippsland Lakes, and by estimated catchment flows and loads for the period July 1995 to June 1999.

The model representation of biophysical processes is based on the CSIRO Port Phillip Bay model, but this has been extended to include:

- improved representations of phytoplankton growth and zooplankton grazing;
- more flexible organic matter stoichiometry;
- improved phosphorus dynamics, including P adsorption and desorption;
- improved sediment biogeochemistry which allows explicitly for effects of oxygen on denitrification efficiencies and P desorption;
- additional phytoplankton functional groups to represent dinoflagellates and cyanobacteria (*Nodularia*).

The model has been calibrated against MAFRI and EPA time series data collected in the Lakes from July 1997 to June 1999. This period includes marked contrasts in flows and loads, with 12 months of drought followed by an extreme flood event in the eastern catchments. The process of model calibration and analysis resulted in the following key conclusions about the function of the Gippsland Lakes, and their response to nutrient and sediment loads.

The Lakes are subjected to very high nutrient loads. For all Lakes and all catchments combined, N loads expressed per unit area of the receiving water body are about twice those into Port Phillip Bay, but N loads expressed per unit volume of the receiving water body into all Lakes are about 5 times, and into L. Wellington about 12 times, those into Port Phillip Bay.

Under low flow conditions, in summer and autumn, or in drought years, flushing rates are very low, with flushing times of order 5 to 9 months. A large fraction of the load is retained and recycled within the Lakes, and internal sinks are as or more important than export for nutrient and sediment budgets.

Loads are highly variable in time. Peak events in L. King and L. Victoria flood surface waters with high nutrient concentrations, which result in large transient blooms. Base loads into L. Wellington are higher, and phytoplankton concentrations there are high much of the time.
L. Wellington is vertically well-mixed, and tends to behave like a chemostat, with high phytoplankton biomass, but low concentrations of dissolved inorganic nitrogen. Between major events, growth is supported by base loads and by recycling and release of ammonia and phosphate from sediments.

L. Victoria and L. King are stratified, except at the end of prolonged dry periods. This makes those basins particularly vulnerable to eutrophication. There is a positive feedback loop active in these Lakes, in which organic matter settling into bottom waters drives oxygen consumption and bottom-water hypoxia. This leads to shut down of denitrification and reduction of iron (Fe) in sediments, and in turn to high rates of ammonia and phosphate release, and increased organic matter production.

There is high light attenuation due to colour dissolved organic matter (CDOM) and suspended sediments, and at times to dense algal blooms. Low light intensities inhibit benthic plants (microalgae, macroalgae and seagrass), except in shallow areas along margins, and also inhibit phytoplankton growth in bottom layers in L. Victoria and L. King. This allows build-up of bottom water ammonia and phosphate, and of course prevents photosynthetic production of oxygen in bottom waters.

Between run-off events, continued phytoplankton growth in surface waters in L. Victoria and L. King is supported by limited vertical mixing of bottom nutrients into the surface loads, or by low base loads. Dinoflagellates, which can vertically migrate and thereby access bottom nutrients, are favoured in the model during these periods.

N:P ratios in loads are above Redfield, but observations suggest surface waters are N rather than P-limited. The model predicts N-limitation because denitrification is more effective overall than P burial. However, there is a delicate balance between N and P-limitation in the model, and brief episodes of P-limitation and high surface ammonia are predicted. There is some evidence for episodes of high surface ammonia in the long-term EPA record.

Periods of N-limitation, with elevated surface phosphate, favour cyanobacterial (*Nodularia*) blooms. These occur in late summer, and may be temperature limited at other times. Salinities are generally favourable for *Nodularia* growth, but may become too high at times in L. King.

Lack of knowledge or data for some key processes and parameters imposes intrinsic limits on the model’s capabilities. While the model is generally able to reproduce most aspects of the observed system behaviour, at least qualitatively, the calibration process revealed other limitations and questions. The key limitations and uncertainties are as follows.

We do not understand processes controlling long-term storage of P (and N) in sediments. The model is calibrated to reproduce the observed behaviour in 1997-99 when forced with repeated cycles of 1995-99 forcing. The longest time scale in the model is about 3 years, and the model is not able to address transients on longer time scales.

The model does not buffer water column phosphate concentrations within the observed range in L. Wellington, and the model may be too easily pushed into P-limitation.

Phytoplankton concentrations appear to decay too rapidly between run-off events in L. Wellington, and this may be due to over-prediction of denitrification efficiencies. On the other hand, the model over-predicts bottom accumulation of ammonia in western and central L. Victoria, suggesting it may under-predict denitrification efficiencies there.

The model predicts the magnitude of bottom water ammonia and phosphate build-up in L. King reasonably well, but the timing is wrong, with high concentrations predicted in spring rather than summer. This may be due to the model representation of dinoflagellate blooms,
which are delayed in the model by grazing interactions. More understanding of dinoflagellate
eco-physiology and population dynamics would be required to improve model representation
of phytoplankton – zooplankton interactions.

Similar limitations apply to the model representation of *Nodularia* dynamics. Temperature
and salinity limits on *Nodularia* in other locations have been taken from the literature, but
need to be validated for the Gippsland Lakes. Factors controlling akinete germination are not
represented, and mortality processes are represented by simple empirical loss rates. In the
model, *Nodularia* blooms respond to temperature, salinity, light and phosphate supply during
growth periods, but may not show appropriate interannual variation. Unless akinete
germination is artificially restricted to L. King, the model tends to predict maximum
*Nodularia* densities in L. Wellington. This is not consistent with observations, and suggests
some key limiting process may be missing.

The hydrodynamic and box model representation of overall flushing rates is considered to be
robustly calibrated against observed salinity. However, the model representation of vertical
mixing is less certain, and this does affect predictions of bottom water hypoxia and
consequently N and P cycling in L. Victoria and L. King.

The implications of these uncertainties for management are taken up in the Scenarios Report.

The model developed and calibrated for Gippsland Lakes represents a significant advance
over the Port Phillip Bay model in a number of areas, including its treatment of the interaction
between N and P dynamics, of the effects of hypoxia on sediment biogeochemistry, and its
incorporation of additional phytoplankton functional groups. The model provides new
insights into the system behaviour of the Gippsland Lakes, and their responses to existing
sediment and nutrient loads. Notwithstanding the limitations and uncertainties outlined above,
the model provides a sound and robust description of most of the key processes controlling
these responses.
1 Introduction

The CSIRO Gippsland Lakes Environmental Study is intended to help managers to understand the function of the Gippsland Lakes ecosystem and the factors underlying environmental issues such as water quality and algal blooms. It also aims to provide managers with the capacity to assess options to address these problems.

Managers need to predict the impacts of management actions that fall outside the range of past experience. We can only do this with confidence to the extent that our predictions are based on a solid understanding of the fundamental processes that determine the response of the Lakes to changes in circulation and nutrient and sediment loads. The development of a process-based model of the Gippsland Lakes, capable of predicting their response to catchment flows and loads and marine exchanges, is therefore a key part of the Study. This model, which we refer to as the integrated model, combines understanding of physical exchanges, biogeochemical processes, and some ecological responses. This final report describes the development and calibration of the integrated model for the Gippsland Lakes.

2 Model Description

The model we have implemented for the Gippsland Lakes is based on a successful model developed as part of the Port Phillip Bay Environmental Study (Harris et al., 1996). That model is described in detail elsewhere (Murray and Parslow, 1997). Rather than repeat those details, we provide here a summary overview of the model, with more detail on the changes to the model since Murray and Parslow (1997).

2.1 Physical Structure

The model divides the Lakes horizontally into a series of polygonal columns or boxes. The box model (Fig. 1) divides the Lakes into the major basins, with some further subdivision of L. Victoria, L. King, and the entrance, to improve the representation of circulation and flushing.

Vertically, the model divides the water column in the deeper boxes, which are typically stratified, into a surface and bottom layer. L. Wellington and Jones Bay are shallow and typically well-mixed, and the model has only one water-column layer in those boxes.

The model also includes a sediment layer underlying each box. The sediment layer includes both dissolved tracers in pore water and particulate tracers. The model allows exchanges of both dissolved and particulate tracers between the water column and the sediment. The basic physical unit corresponding to a specific box and layer is referred to as a cell. The model includes both water column and sediment cells.

The model uses a relatively coarse spatial resolution because individual simulations need to run over many years, and many simulations are required for both calibration and validation. By comparison, the hydrodynamic model has a much higher resolution, which is required to predict the circulation within the Lakes. However, it would not be feasible to run the integrated model at this spatial resolution, at least for many simulations over long time periods. Nor would we have the information to properly parameterise or test the model at this higher resolution. We expect that the box-model geometry will suffice to address the key time and space scales associated with the major ecological processes in the Lakes. In general this is
borne out below by calibration: we comment specifically on cases where the coarse resolution may limit model performance.

Figure 1. Bathymetry in the 8-box model geometry.

### 2.2 Physical transport.

The model represents time-varying horizontal and vertical exchanges of water and dissolved and suspended particulate tracers between water column cells. These exchanges are provided as forcing files: their derivation is discussed below under forcing.

The model allows exchanges of both dissolved and particulate tracers between the water column and the underlying sediment. The model includes both settling and resuspension of particulate materials. Settling fluxes of individual tracers are calculated from sinking rates and concentrations. Resuspension rates of bed sediments are controlled by values of bottom shear stress, which are provided as forcing files (see below).

### 2.3 Biogeochemical / ecological processes

The model represents the cycling of nitrogen, phosphorous and carbon through both pelagic and benthic ecosystems. The Port Phillip Bay Study showed convincingly that, in shallow coastal embayments and estuaries, representation of both pelagic and benthic systems, and the coupling between them, is critical to understanding and predicting the response to nutrient loads (Murray and Parslow, 1997).

The integrated model has three modules: water column, sediment, and epibenthos (or sediment surface). The water column module describes a simple planktonic food web. The model currently includes four phytoplankton functional groups: small phytoflagellates, larger diatoms, dinoflagellates and cyanobacteria (*Nodularia*). There are two size classes of zooplankton, which graze respectively on small flagellates, and on diatoms and
dinoflagellates. The model represents a range of forms of nonliving particulate and dissolved organic matter (discussed further below), as well as inorganic nutrient species, dissolved inorganic carbon (DIC) and dissolved oxygen.

The sediment module represents the breakdown of particulate and dissolved organic matter through microbial and detritivore activity, which consumes oxygen and releases DIC and inorganic nutrients. The module includes the processes of nitrification and denitrification, which have been shown to play a pivotal role in nitrogen cycling in coastal systems (Harris et al., 1996). The module also includes benthic microalgae (microphytobenthos or MPB), which have now been shown to make a major contribution to primary production in many coastal systems.

The epibenthic module represents two functional classes of attached macrophytes: macroalgae, which take up nutrients from the water column, and seagrass, which take up nutrients from the sediment pore water.

A schematic view of nitrogen cycling through water column, sediment and epibenthic components is shown in Fig. 2. This cycle is similar to that represented in the Port Phillip Bay model (Murray and Parslow, 1997). However, there have been a number of substantial improvements in the model since that study, and these are now discussed in more detail.

2.3.1 Phosphorus Dynamics.

The Port Phillip Bay model included phosphate as an explicit variable, but phosphorus participated in nutrient cycling as a “passive” tracer, always accompanying nitrogen at Redfield ratios. This was deemed adequate because phosphorus is always present at great excess in Port Phillip Bay.
The representation of phosphorus is substantially upgraded in the current model. First, phosphorus is now treated as a potentially limiting nutrient for plant growth along with nitrogen. The interaction between N and P in controlling plant growth is discussed further below. Note that silicate has been dropped from the model. Silicate is generally present at high concentrations, and is not thought to be limiting to diatom growth in the Gippsland Lakes (Longmore et al. 2001).

Second, adsorption-desorption interactions between dissolved inorganic P (DIP) and adsorbed particulate inorganic P (PIP) are now represented explicitly in both water column and sediments. Webster and Grace (2001) analysed data derived from P-adsorption experiments in the Gippsland Lakes. They concluded that the data fitted a non-linear power-law or Freundlich isotherm, rather than the Langmuir isotherm which is often assumed. According to the Freundlich isotherm:

\[ P_s = A P_d^B, \]

where \( P_s \) is the phosphorous adsorbed to sediment, \( P_d \) the dissolved phosphorous concentration, and \( A \) and \( B \) are constants.

Webster and Grace recommended, on the basis of their analyses, that the exponent \( B \) should be given a value of 0.34 for both water column and bed sediments in the Gippsland Lakes. (This was the average of the estimated exponents, which ranged from 0.28 to 0.45). They observed considerable variation in the value of \( A \). For bed sediments, they recommended a mean value of \( A=74 \), noting a factor of 3 variation. For suspended sediments, the variation in \( A \) among samples was extremely large, ranging from 58 up to 5101. They concluded that the measured high values might be due to uptake by algae during the experimental procedure, as opposed to adsorption, and suggested using a relatively low value of 107 for adsorption. We comment further on this under model calibration.

In the model, the adsorption-desorption reaction is represented as a dynamical balance between the processes of adsorption and desorption, so as to reproduce the recommended isotherms. The net adsorption minus desorption flux is given by:

\[ \frac{d \text{PIP}}{dt} = r_{\text{ads-P}} \cdot (P_{\text{ads-coeff}} \cdot \text{TSS} \cdot \text{DIP}_{\text{ads-exp}} - \text{PIP}). \]

Here \( r_{\text{ads-P}} \) (d⁻¹) controls the reaction rate, while \( P_{\text{ads-coeff}} \) (m³ kg⁻¹) and \( P_{\text{ads-exp}} \) correspond to Webster and Grace’s \( A \) and \( B \) parameters. TSS is the concentration of suspended inorganic sediment (kg m⁻³).

Finally, the model now allows for variation in the N:P stoichiometry of organic matter depending on its origin. This is discussed further under stoichiometry below.

### 2.3.2 Plant growth

The PPBES model represented plant growth in a fairly traditional way, using Monod equations and half-saturation constants \( K_N \) to describe nutrient-limitation of plant growth, and a bilinear function incorporating a light-saturation intensity \( K_l \) to describe light-limitation. The “Law of the Minimum” was used to describe interactions among multiple limiting nutrients, while a multiplicative model was used to describe the interaction between nutrient and light limitation. The latter is known to underestimate plant growth when nutrients and light are co-limiting.
The plant growth module has subsequently been replaced by the so-called Chemical Reaction (CR) scheme developed by Baird and Emsley (1999). This scheme has substantial advantages over the PPBES scheme. First, it uses bio-physical formulations as far as possible to compute uptake rates. For example, it computes potential nutrient uptake rates by phytoplankton as a function of far-field concentration and phytoplankton cell size and shape, assuming uptake is limited by the diffusive flux to the cell surface. Macroalgal potential nutrient uptake rates are computed using a model for the flux of nutrient across a turbulent boundary layer. Light capture by phytoplankton for photosynthesis is based on a model of optical cross-section, again based on cell size. The use of bio-physical formulations means that parameters such as $K_N$ and $K_I$ are derived from cell radius and fundamental physical laws, reducing the number of free parameters in the model.

Also, the CR scheme represents internal cell quotas for nitrogen, phosphorus and carbon, and assumes that growth rate depends on the product of factors proportional to internal cell quota. The scheme in principle allows cell quotas to be treated as dynamical variables and uptake to be decoupled from growth. However, in the current implementation, to reduce the number of model variables, quotas are assumed to adjust rapidly to changes in external concentration, and a quasi-steady-state relationship between external concentrations of DIN, DIP and light has been derived. This relationship has the substantial advantage over the multiplicative light-nutrient interaction used in PPBES that the initial slope of the growth rate vs concentration curve for one limiting factor is not reduced by other limiting factors.

A detailed description of the CR formulation has been prepared as a separate document (Baird et al., 2001).

### 2.3.3 O:C:N:P stoichiometry.

The model now explicitly represents the stoichiometry of all plant components, and distinguishes a Redfield (138:106:16:1) stoichiometry for phytoplankton and microphytobenthos from an Atkinson and Smith (1983) (716:550:30:1) stoichiometry for macrophytes. Zooplankton are assumed to have Redfield stoichiometry.

Allowing different stoichiometries complicates the treatment of detrital material. The model now includes two labile detrital pools: one of planktonic origin with Redfield stoichiometry (Lab_Det_Plank), and one of macrophyte origin with Atkinson stoichiometry (Lab_Det_Benth). It is assumed that by the time organic matter has been converted to refractory particulate detritus or refractory dissolved organic matter, its origin no longer affects breakdown rates. However, because there are different sources, with different stoichiometries, the model must separately represent DOC, DON and DOP, and Ref_Det_C, Ref_Det_N and Ref_Det_P, and allow these to vary independently, in order to conserve mass. This approach also provides much more flexibility in representing catchment load composition.

### 2.3.4 Zooplankton Encounter Rates.

Analogous to the bio-physical sub-models underpinning the CR scheme above, an encounter rate theory based on Jackson (1996) is now used to calculate zooplankton clearance rates, based on zooplankton and prey radius and zooplankton swimming speeds. While one can use a standard relationship between swimming speed and cell size, zooplankton swimming speeds and therefore maximum clearance rates are adjustable parameters.
2.3.5 Temperature dependence.

In the bio-physical sub-models discussed above, the temperature dependence of key physical and biochemical processes is accounted for. A $Q_{10}$ of 2 (i.e. a doubling of rates for an increase in temperature by 10°C) is generally applied to biochemical or physiological rates (e.g. maximum growth rates), while the temperature and salinity dependence of physical constants such as viscosities and diffusivities is based on physical law.

2.3.6 Light attenuation.

The PPBES model accounted for the specific attenuation due to phytoplankton and dissolved and particulate organic matter. The current model explicitly includes the attenuation due to suspended sediments (TSS), which play a key role in the Gippsland Lakes. In some Australian catchments, the runoff is highly coloured due to high concentrations of humic material. This coloured dissolved organic matter (CDOM) can dominate light attenuation in estuaries, and cannot be accounted for simply in terms of total DOC or DON. The model assumes that CDOM is conservative in estuaries: under this assumption, its concentration can be computed from salinity. The attenuation due to CDOM in freshwater, $K_{C_{\text{fw}}}$, is prescribed as a model parameter.

In contrast to the PPBES model, the model now deals explicitly with both self-shading and light interactions among benthic plants. The model assumes a specific light absorption cross-section per unit biomass of macroalgae and seagrass. Self-shading is taken into account in computing light-limited growth rates, and this sets a bio-physical upper limit to seagrass and macroalgal standing stock. Further, it is assumed that macroalgae can overgrow seagrass, so light attenuation by macroalgae reduces light available to seagrass. The result is that the model predicts in a simple but effective way the loss of seagrass due to overgrowth by filamentous macroalgae under eutrophic conditions. Both seagrass and macroalgae are assumed to shade microphytobenthos.

2.3.7 Flocculation.

Catchment runoff may contain loads of fine clay particles, which settle very slowly. Changes in surface chemistry associated with increasing salinity in estuaries lead to flocculation and more rapid settling of these fine particles. This process is represented in the model by distinguishing two classes of fine inorganic sediment: unflocculated clay particles (TSS_unfloc), and flocculated particles (TSS_floc) with higher sinking rates. The model fixes two salinity thresholds, 6 and 10 PSU, and the conversion rate from TSS_unfloc to TSS_floc increases linearly from zero at the lower threshold to a maximum rate $r_{\text{floc}}$ at the upper threshold.

Because these sediment fractions settle at different rates, the adsorbed inorganic P pool PIP is also divided into a PIP_unfloc and PIP_floc pool. Adsorption-desorption interactions of these fractions with DIP are modelled separately.
2.3.8 Sediment Biogeochemistry.

The sediment biogeochemistry module developed for Port Phillip Bay represented breakdown of organic matter and oxygen consumption in the sediment, but aerobic respiration, nitrification and denitrification were not explicitly dependent on oxygen concentrations in pore waters. These dependencies were instead represented implicitly, by making nitrification and denitrification efficiencies depend directly on sediment respiration rates.

This formulation worked quite well in Port Phillip Bay, but it cannot be applied to stratified systems such as the Gippsland Lakes. In Port Phillip Bay, water column oxygen concentrations were almost always high, so sediment oxic status was controlled primarily by sediment respiration rates. However, in the Gippsland Lakes, oxygen is often depleted in bottom waters, and this interacts with sediment respiration rates to determine sediment oxic status. If the model is to simulate bottom water oxygen and denitrification efficiencies under stratified conditions, it needs to explicitly represent the dependence of sediment processes on oxygen. This essentially required a rewrite of the sediment biogeochemistry module.

A key issue in formulating the new model is the representation of the physical structure of the sediments. Sediments can be characterised by very sharp vertical gradients: for example, at high sediment respiration rates, the oxic layer can be only 0.5 mm or less. In the presence of burrowing macrofauna, these gradients extend over the surface of burrows, forming a complex 3-D interface. Resolving these fine gradients can result in model pools with very short turnover times, demanding correspondingly short model time steps, and unacceptable increases in computational requirements.

After considering a number of options, we have decided to retain the single sediment layer used in Port Phillip Bay. We have used a sediment layer thickness of 4 cm, corresponding to about 40 years of average sediment deposition in the Gippsland Lakes. This choice involves a number of compromises. We have to accept that it is not possible to reproduce variations in tracer concentrations and process reaction rates within the sediment. Instead, we must adopt bulk process formulations based on average concentrations which simulate as closely as possible the expected pools and fluxes, especially those associated with exchanges between the sediment and the overlying water column. We discuss these formulations below for each of the key processes.

a. Respiration.

As in Port Phillip Bay, the model computes a total sediment respiration rate $\text{Tot}_{\text{resp}}$ based on the breakdown rates of the various organic matter pools within the sediment. However, rates of aerobic respiration and oxygen consumption ($\text{Aer}_{\text{resp}}$) are limited by the oxygen concentration within the sediment:

$$\text{Aer}_{\text{resp}} = \frac{\text{Tot}_{\text{resp}} \times \text{Oxygen}}{(K_{O_{aer}} + \text{Oxygen})},$$

where $K_{O_{aer}}$ is a half-saturation constant for aerobic respiration. We assume that the remaining respiration demand is met anaerobically.

In the model, the physical exchange of pore water and dissolved tracers between the sediment bed and the overlying water column is controlled by an exchange velocity $V$ (m s$^{-1}$). This sets an upper limit to the physical rate at which oxygen can be supplied to the sediment, given by $V_{Ow}$, where $Ow$ is the water column oxygen concentration. The exchange velocity must be chosen so as to allow maximum oxygen consumption rates to match those observed in the field. A typical upper bound for measured sediment respiration rates is 100 mmol $O_2$ m$^{-2}$ d$^{-1}$. 
For oxygen saturation concentrations of around 250 mmol m\(^{-3}\), this requires an exchange velocity of about 0.4 m d\(^{-1}\), or about 4E-6 m s\(^{-1}\).

If it were due to molecular diffusion alone, such a high exchange velocity would imply a diffusive length scale of about 0.25 mm. We expect very high oxygen consumption rates to correspond to very thin oxic layers, much less than the 4 cm thickness of the sediment layer represented in the model. By adopting a high value for V for oxygen, we are implicitly assuming that a thin oxic layer is established in the sediment.

Under this simple bulk formulation, we can distinguish two extremes in sediment oxygen dynamics. If the oxygen demand due to total respiration is much less than the physical supply rate V.Ow, then it will be met primarily aerobically, and oxygen concentrations in the sediment will be relatively high, and given approximately by Ow – Tot_resp / V. If the oxygen demand is much larger than the maximum physical supply rate, then a substantial fraction will be met anaerobically, and the oxygen concentration in the sediment is given approximately by Ow . KO_aer . V / Tot_res. If the overlying water column is well-oxygenated, this second situation can occur only in the presence of very high total respiration rates. However, if bottom water oxygen concentrations are depleted to low levels, sediment oxygen can be depleted even at modest sediment respiration rates.

For V = 0.4 m d\(^{-1}\), the oxygen pool in the sediment turns over 10 times per day due just to physical exchange. This is much faster than most other pools in the model, so that the oxygen pool in the sediment tends to come rapidly to equilibrium with supply and demand. This will be the case in any sediment with a thin oxic layer. The parameter KO_aer effectively determines the oxygen pool size and turnover under conditions of high respiration demand and/or low supply. The value for this parameter has been set to 500 mg O m\(^{-3}\), on pragmatic grounds. If this parameter is set much lower, then the turnover time of the oxygen pool under conditions of high demand becomes so short as to increase model integration times to unacceptable levels. Even at the adopted level, the model time step must be reduced from 1 h to 0.2 h. Using a relatively high value for KO_aer has little influence on oxygen dynamics. The oxygen pool would turn over even more rapidly if a lower value of KO_aer were used, but it is already effectively in steady-state balance with pools and fluxes which change much more slowly. The value of KO_aer does affect the parameterization and formulation of other oxygen dependent processes discussed below.

b. Nitrification and Denitrification.

In the Port Phillip Bay model, the nitrogen produced by benthic respiration was allocated directly to pore water ammonia, nitrate or N\(_2\) gas according to nitrification and denitrification efficiencies, which depended directly on the instantaneous sediment respiration rate. In the new formulation, all inorganic nitrogen produced by benthic respiration is added directly to the pore water ammonia pool. Pore water ammonia is then converted to nitrate via nitrification at a rate that depends on the ammonia and the oxygen concentration. Nitrate is then converted to N\(_2\) gas at a rate that depends on the nitrate concentration and the oxygen concentration.

Under most conditions, nitrification is the rate-limiting step. There is competition between physical transport of ammonia out of the sediment, and conversion to nitrate. The nitrification efficiency is essentially given by the ratio nitrification / (nitrification + transport). For example, to achieve a nitrification efficiency of 80%, it is necessary that the nitrification flux be 4 times the physical transport flux. However, under conditions of high sediment respiration rates, and low sediment oxygen, this position must be reversed, so that most of the ammonia is released from the sediment.

We first considered a formulation for nitrification of the form:
Nitrification = \( r_{\text{nit_sed}} \times \text{NH}_4 \times \text{Oxygen} / (K_{\text{Oit}} + \text{Oxygen}) \).

If \( K_{\text{Oer}} \) is small, \( r_{\text{nit_sed}} \) is sufficiently large, and \( K_{\text{Oit}} \gg K_{\text{Oer}} \), this formulation produces the desired dependence of nitrification efficiency on sediment respiration rate and bottom-water oxygen. However, for the high values of \( K_{\text{Oer}} \) adopted here (for computational efficiency), a modified formulation was found to be necessary:

\[
\text{Nitrification} = r_{\text{nit_sed}} \times \text{NH}_4 \times \text{Oxygen}^2 / (K_{\text{Oit}}^2 + \text{Oxygen}^2).
\]

This produces the desired behaviour for \( r_{\text{nit_sed}} = 0.5 \text{ d}^{-1} \) and \( K_{\text{Oit}} = 10000 \text{ mg m}^{-3} \). Note that we have adopted a much lower physical exchange velocity \( V_N \) for ammonia (and other dissolved pore water nutrients) of about 0.005 m d\(^{-1}\), corresponding to a diffusive length scale of about 2 cm. This reflects the fact that gradients in ammonia extend well beyond the oxic layer.

The formulation for denitrification is given by

\[
\text{Denitrification} = r_{\text{den}} \times \text{NO}_3 \times K_{\text{Den}} / (K_{\text{Den}} + \text{Oxygen}).
\]

Again, there is competition between denitrification and physical exchange of nitrate with the overlying water column. To achieve high denitrification efficiencies, \( r_{\text{den}} \) must be much greater than the physical exchange rate constant, given by \( V_N/0.004 \) (d\(^{-1}\)). Under conditions favouring denitrification, release of nitrate is low compared with release of ammonia, which implies that \( r_{\text{den}} \gg r_{\text{nit_sed}} \). We have adopted a value for \( r_{\text{den}} \) of 5.0 d\(^{-1}\). The adopted value of \( K_{\text{Den}} \) is 500 mg m\(^{-3}\). However, this produces only a small decrease in denitrification efficiency at very low respiration rates, when oxygen is close to saturation. It is difficult in this single layer bulk model to distinguish between a situation in which respiration rates are moderate, the oxic layer is thin, and denitrification efficiencies are high, and an oligotrophic situation in which respiration rates are very low, oxic layers are deep, and denitrification rates are low, with high release rates of nitrate. However, oligotrophic conditions are unlikely to occur in the Gippsland Lakes, at least under current or anticipated loads.

c. P adsorption, TSS and oxygen.
As noted above, the adsorption-desorption kinetics for phosphorus in sediments are based on the isotherms recommended by Webster and Grace (2001). However, these kinetics must bemodified in the bulk sediment formulation to allow for the effects of oxic state on P adsorption, and for vertical gradients in oxygen within the sediment. P adsorbs to ferric oxides and hydroxides within oxic sediments. Under anoxic conditions, the iron is reduced, releasing adsorbed P. This is represented in the model by making the P adsorption coefficient dependent on the oxygen concentration:

\[
P_{\text{ads_coeff}}^* = P_{\text{ads_coeff}} \times \text{Oxygen} / (K_{\text{Pads}} + \text{Oxygen}).
\]

In the model, \( K_{\text{Pads}} \) is given a value of 2000 mg O m\(^{-3}\) which is found to result in efficient P binding for oxic sediments, and P release under anoxic conditions.

However, this raises another issue concerning the magnitude of the TSS pool available for P adsorption in the sediment. As discussed above, the bulk sediment model implicitly represents
a thin oxic layer, about 0.5 mm thick. It is only the sediment within this oxic layer that is available for P adsorption. If all of the bed sediment in the 4 cm layer is made available for P adsorption, this allows the accumulation of a very large PIP pool, and, under conditions of alternating oxic and anoxic conditions, results in the release of very large amounts of DIP, much larger than those observed. In order to prevent this from occurring, the TSS concentration in the sediment must be scaled back roughly by the ratio of the oxic layer thickness to the full depth of the layer, or about 80:1.

d. P and TSS burial.
Budgets for P and TSS in L. Wellington indicate that a significant fraction of the P and TSS loads into the lake are sequestered in the sediment (see Calibration for further detail). The model allows a fraction of the adsorbed P and TSS pools in the sediment to be buried or immobilized in each year. This burial process is not well understood, especially for P, and should be regarded as a useful empirical assumption. It is discussed further under Calibration.

To sum up, this new sediment biogeochemistry significantly extends the Port Phillip Bay model to allow explicitly for effects of variation in oxygen supply and oxic status in the sediments. It is limited both in its spatial representation of the sediment, and in process detail. The lack of vertical resolution is largely a pragmatic decision driven by the high computational cost of a finely resolved depth-structured model. The process representation is also pragmatic, and has been designed (through a process of offline analysis) to ensure that changes in bottom-water oxygen and sediment oxygen demand have the expected effects on nitrification and denitrification efficiencies and P exchange with bed sediments. The formulation includes a number of uncertain empirical parameters, which are adjusted in model calibration.

It is important that this new formulation be able to represent the changes in sediment biogeochemistry which might be expected under proposed scenario changes in loads and exchanges, as well as under existing conditions. There are good reasons to be confident that this is the case. Because of temporal and spatial variability in loads and mixing, observations during the calibration period encompass a wide range of conditions, from low sediment respiration rates and well-oxygenated bottom waters, to anoxic bottom waters. Changes in loads and exchanges would be expected to lead primarily to changes in the relative frequency and intensity of these conditions.

2.3.9 Dinoflagellates and Cyanobacteria
The Port Phillip Bay model contained two phytoplankton functional groups, small flagellates and diatoms. Small flagellates are capable of outcompeting diatoms for nutrients under oligotrophic conditions, but their biomass tends to be grazer controlled under eutrophic conditions. Diatoms are capable of rapid growth and bloom formation under high nutrient conditions, and are subject to loss rates due to grazing by mesozooplankton (which respond much more slowly), and to sinking. Two other phytoplankton functional groups, dinoflagellates and cyanobacteria, are observed to play an important role in the Gippsland Lakes algal blooms.

In the Gippsland Lakes and in other systems, dinoflagellate blooms tend to occur following diatom blooms, under conditions of strong stratification, high light attenuation and elevated bottom water nutrients. In the Huon Estuary, dinoflagellates were observed to undertake strong and regular diel vertical migration, and it was hypothesized that this allowed dinoflagellates to take up nutrients in bottom water at night, and fix carbon in surface waters in the day (CSIRO Huon Study Team, 2000).
A dinoflagellate functional group has been introduced into the model. The biophysical representation of light absorption and nutrient uptake by dinoflagellates is similar to that for diatoms and flagellates, and follows the CR formulation described earlier. The key difference is that dinoflagellates are allowed to vertically migrate in stratified water columns, moving between the surface and bottom layers on a diel cycle. However, in order to allow dinoflagellates to take advantage of vertical migration, it is necessary to substantially modify the calculation of their growth rate.

While the CR formulation potentially allows carbon and nutrient quotas to vary independently, the implementation used for diatoms and flagellates adopts a fixed stoichiometry, and estimates a “steady-state” growth rate which balances the potential instantaneous supply rates of light and nutrients. Under stratified conditions, this formulation does not allow dinoflagellates to capture the benefits of vertical migration. They are unable to grow in surface waters (where they are nutrient limited) or in bottom waters (where they are light limited). The model was therefore modified to allow the dinoflagellate carbon and nitrogen content to vary independently, within fixed bounds. Dinoflagellates can take up nutrients in the dark in bottom layers until the C:N ratio falls below Redfield, and fix carbon in surface waters until the C:N ratio exceeds a maximum value. The ratio of maximum to minimum C:N ratios is an adjustable parameter, which, under strongly stratified conditions, sets an upper bound to the daily growth rate.

Dinoflagellates are generally considered to be slow-growing large cells, and are therefore out-competed by both diatoms and small flagellates under non-stratified conditions. The model assumes that dinoflagellates and diatoms are grazed by the same large zooplankton functional group. This assumption is discussed further under calibration.

High biomass cyanobacterial (Nodularia) blooms occur in the Gippsland Lakes, usually in summer and autumn. These blooms are N-fixing, and so can escape nitrogen limitation. In some years they are a major source of nitrogen, resulting in TN concentrations much higher than those produced by catchment runoff.

A cyanobacteria functional group has been added to the model. This group is modelled explicitly on Nodularia. The uptake of light and nutrients again follows the CR scheme, with the following modifications to allow for N-fixation. It is assumed, following Sanz-Alferez and del Campo (1994), that Nodularia is unable to utilize nitrate, but will take up ammonia in preference to fixing N. The model computes two growth rates for Nodularia, one based on ammonia, DIP and light, and one based solely on DIP and light. It is assumed that Nodularia is always able to achieve the DIP/light limited growth rate, and fixes sufficient N to make up the difference between this rate and the ammonium-limited rate.

Studies in the Peel-Harvey system in West Australia have shown Nodularia to be inhibited by low temperatures and high salinities. It has been suggested that temperatures greater than 17 °C and possibly greater than 20 °C are required for blooms to occur, and that blooms don’t occur at salinities above 30 PSU (Lukatelich and McComb, 1986). A temperature requirement of 17 °C was also noted by Sellner (1997) for the Baltic Sea. In the model, the maximum growth rate of Nodularia is modified by special salinity and temperature dependent terms. Growth rates are assumed to fall off exponentially for temperatures below a minimum temperature threshold, TempgrowBG, and to fall off exponentially for salinities above a maximum value, SaltgrowBG. These values were initially set to 17 °C and 25 °C respectively. Maximum growth rates for Nodularia are about 0.8 d⁻¹ at 28 °C (Sanz-Alferez and Campo, 1994), corresponding, for a Q₁₀ of 2, to 0.4 d⁻¹ at 18 °C.

Nodularia forms resting stages (akinetes) in shallow sediments over winter. It has been suggested that akinetes require low salinities (< 15 PSU) for germination. Longmore (2000) cites a number of other requirements for germination proposed by Huber (1985), but did not
find these to be consistent with the timing and location of *Nodularia* akinete germination in the Gippsland Lakes. In the absence of detailed autecological studies of *Nodularia* strains in the Gippsland Lakes, we have assumed instead that there is a period of akinete germination and associated supply of *Nodularia* to waters in L. King only in summer each year. Thus, the biomass of *Nodularia* predicted by the model depends on factors controlling growth and losses of cells in the water column, rather than akinete germination.

*Nodularia* can be positively buoyant, and Sellner (1997) reports that Baltic Sea populations are able to control buoyancy and may control position in the water column. However, we have not provided for diel vertical migration by *Nodularia* in the model. They are currently treated as neutrally buoyant in the model: in the 2-layer model, it makes little difference whether they are treated as neutrally or positively buoyant, as they grow almost entirely in the surface layer.

We know little about loss processes for *Nodularia*. Sellner (1997) reports that they are not grazed by most mesozooplankton in the Baltic, although there may be specialist grazers. It seems unreasonable therefore to assume that they are grazed by the same large zooplankton functional group as diatoms and dinoflagellates. Rather than introduce another (hypothetical) grazer, we have assigned a trophic closure mortality term, allowing both linear and quadratic mortality.

## 3 Calibration Data Sets

The model has been calibrated for the period July 1997 to June 1999. This corresponds to a field study conducted by MAFRI (Longmore, 2000), which has provided the key data sets for model calibration.

The MAFRI Study measured dissolved inorganic nutrients (NH$_3$, NO$_x$, PO$_4$), TN, TP, and chlorophyll a in surface and bottom waters fortnightly at 4 sites in L. Victoria and L. King (Fig. 3: Sites 4, 8, 18 and 19). Profiles of dissolved oxygen (DO), T and S were also obtained at these sites and a number of other sites shown in Fig. 3. The DO data at sites 4, 8, 18 and 19 are used here for model calibration, and salinity data from all sites have been used in calibrating the hydrodynamic model and in estimating box model exchanges using inverse techniques (see below).
To supplement the MAFRI observations, we have used data from EPA monitoring sites 2306, 2311, 2314, 2316 and 2322. The EPA sites are only monitored at 2-monthly intervals, but they include suspended sediment (NFR) and Secchi depth as well as the chemical data collected by MAFRI. With these additional variables, and the fact that site 2306 lies in L. Wellington, the EPA data make a valuable contribution to model calibration and testing.

The data sets for the calibration period are presented and discussed in this interim report. A much longer time series of EPA data exists, dating back to 1986, again mostly collected at 1 to 2 month intervals.

4 Forcing Data Sets

The key forcing data sets required by the model are the physical exchanges which control advection and mixing of water column tracers, and the catchment loads of nutrients and sediments. Other forcing data include temperature, surface irradiance, and bottom shear stress which controls resuspension. Boundary conditions are applied at the Bass Strait boundary.

4.1 Physical Exchanges

Two sets of physical exchanges have been computed for the calibration period July 1997 to June 1999. The first set is based on the application of an inverse technique to salinity data collected by MAFRI and EPA through the calibration period. The inverse technique uses a constrained minimisation to estimate (at each time step) the set of physical exchanges that
allows the box model, driven by observed river discharges, to best reproduce the observed salinity time series. The technique exploits the fact that the first-order representation of mixing and advection used in the box model is a linear operation that can be inverted by applying Quadratic Programming techniques.

The approach is described in more detail in the Ecological Model Interim Report. It yields good agreement between predicted and observed salinities (Fig. 4). The mismatch in predicted and observed salinities in the first four months in Fig. 4 arises because the model predictions shown represent the last two years of a 10 year run, with the 1997-99 forcing repeated 5 times. The initial salinities at the start of the last two years are higher than observed, presumably because runoff preceding July 1997 was higher than runoff preceding July 1999.

A second set of exchanges has been computed from outputs of the hydrodynamic model developed by CSIRO for the Gippsland Lakes Environmental Study (Walker and Andrewartha, 2000). These exchanges are also estimated using an inverse technique. In addition to salinity, a number of other conservative tracers with contrasting source-sink patterns are simulated by the hydrodynamic model. These tracers are then averaged over box-model cells at each output time step. The inverse procedure then estimates a set of constrained box-model exchanges at each time step which allow the box model to best reproduce the time series of tracers predicted by the hydrodynamic model. Again, the technique involves the minimisation of a cost function based on a sum of weighted squared deviations between the box model and hydrodynamic model predictions, subject to various constraints.

The introduction of additional conservative tracers changes the estimation problem from an under-constrained to an over-constrained problem, and thereby leads to more robust exchange estimates. This method of stepping from the fine-resolution hydrodynamic model to the coarse-resolution box model differs from the particle-tracking method used in Port Phillip Bay. The inverse method has certain advantages. Perhaps most important, it automatically incorporates and allows for the numerical diffusion implicit in the coarse box-model resolution.

Although the initial calibration was based on the exchanges derived from observed salinities (see the Ecological Model Interim Report), the final calibration has been based on exchanges derived from the hydrodynamic model runs. The two methods have different strengths and weaknesses. The exchanges based on observed salinities may include errors arising from the need to interpolate between fortnightly observations, and from the under-constrained nature of the inverse problem. (The solution is effectively a minimum energy solution, but may not be the right solution.) The exchanges based on the hydrodynamic model are subject to any limitations of the hydrodynamic model itself. As one might expect, the two sets of exchanges lead to qualitatively similar results.
There are two striking aspects of circulation in the Lakes demonstrated by the analysis of the data and the hydrodynamic model results. First, during periods of low runoff, exchanges between the Lakes and Bass Strait, and between L. Wellington and L. Victoria, are very small and flushing times are very long. Secondly, tidal amplitude is extremely attenuated inside the Lakes, and tidal circulation contributes little to flushing.

Partly as a result of these small tidal currents, and partly due to incursions of Bass Strait water along the bottom, associated with sea-level changes in Bass Strait (which vary over time scales of days to weeks), L. Victoria and L. King remain strongly stratified even after many months of negligible runoff. While a relatively deep (5 m) surface mixed layer is quickly established by wind mixing after runoff events, exchange between the surface and bottom layers is limited. Both the long flushing times and persistent stratification have important implications for nutrient cycling which are discussed in the section on calibration.

Fig. 4. Comparison of observed salinities (*) and box-model predictions (line) using exchanges from the inverse method applied to the measurements as described in text. For L. King and L. Victoria, red lines and asterisks correspond to lower layer, blue lines and asterisks to upper layer.
The calibrated model will be used to address a number of scenarios involving changes in river flows and/or loads, and changes in exchanges with Bass Strait. Those scenarios involving changes in flows and exchanges cause changes in circulation patterns in the Lakes, and the hydrodynamic model has been rerun for each of these scenarios. Box-model exchanges for these scenarios can only be derived from the hydrodynamic model output. It is partly for this reason that we have used exchanges based on the hydrodynamic model for the final model calibration.

While we have focused on the sampling period from July 1997 to June 1999 for model calibration, it was recognised that this was an unusual period, with extremely low catchment flows and loads in the first 12 months, followed by an unusually intense flood in June 1998 in the eastern catchments. It was agreed relatively late in the study that for the scenarios, we would extend the forcing to include the period July 1995 to June 1997, which includes two more “typical” wet seasons. While estimates of catchment flows and loads are available for this period, it was not practical in the time available to extend some of the other hydrodynamic model forcing back to the 1995-97 period. We therefore decided to adopt a compromise, in which the hydrodynamic model was run for the 1995-97 period with 1995-97 flows, but other forcing from 1997-99. The resulting simulations will not reproduce day-to-day sea levels and exchanges in this period, but are expected to reproduce the dominant effects of river flows on salinities and exchanges.

Box-model exchanges for the 1995-99 period were estimated from these extended hydrodynamic model runs. These physical exchanges, along with estimated catchment flows and loads, were used to drive the box model for the period July 1995 to June 1999 for the calibration and scenario runs. Although EPA monitoring of the Lakes did occur in the 1995-97 period, it was not possible to estimate exchanges from salinity observations in this period, as the sampling frequency was too low.

4.2 Catchment Loads

The principal rivers are the Latrobe, Avon and Thomson discharging into L. Wellington, and the Mitchell, Nicholson and Tambo discharging into northern L. King and Jones Bay. Loads of total nitrogen (TN), total phosphorus (TP) and total suspended sediment (TSS) from these catchments have been estimated by University of Melbourne based on a statistical analysis of runoff and concentration data from the rivers. Load estimates for the Macalister Irrigation District (MID) are included. A detailed account of the derivation of loads can be found in the final report by Grayson et al. (2001). In the model, loads from the Avon, Latrobe and Thomson are discharged into L. Wellington (model box 6), loads from the Nicholson and Mitchell into Jones Bay (model box 8 - see Hydrodynamic Model Report, Walker and Andrewartha, 2000), and loads from the Tambo into L. King north (model box 7). Grayson et al. have also estimated loads from small coastal catchments into model boxes 2, 3, 4 and 5, corresponding to L. King south and L. Victoria.

The TN and TP loads need to be subdivided further into the inorganic and organic fractions represented in the model. For nitrogen, the relevant components are potentially NOx, NH4, labile detrital N, dissolved organic N (DON) and refractory detrital N. For phosphorus, the relevant components are potentially dissolved inorganic P (DIP), adsorbed inorganic P (PIP_unfloc), labile detrital P, dissolved organic P (DOP) and refractory detrital P.

We have based the allocation of TN and TP among these components on data where possible. In particular, EPA has analysed the relationship between concentrations of NOx, NH4 and DIP and river discharge for each catchment, and summarised these in the form of log-log regressions. We have used these log-log regressions to compute the ratios NOx/TN, NH4/TN
and DIP/TP as a function of flow for each catchment, and applied these fractions to the TN and TP loads provided by Grayson et al. (2001).

However, the allocation of the remaining organic / particulate fractions among model components is somewhat arbitrary. We have assumed that, after subtracting NOx and NH3, the remaining TN is all organic, and have allocated 50% to DON and 50% to labile detrital N. The labile detritus in catchment loads is assumed to have Atkinson (macrophyte) stoichiometry, which is used to calculate the labile detrital P (LDP). The excess TP not accounted for as DIP or LDP is assumed to be either adsorbed inorganic P or DOP. We have used the P-adsorption isotherms recommended by Webster and Grace (2001) to estimate the adsorbed P (PIP) concentration that would be in equilibrium with the estimated TSS and DIP concentrations. If this concentration is less than the (TP–DIP-LDP) pool, the remainder is assumed to be DOP. Otherwise, DOP is set to zero, and all the excess TP is assigned to the PIP pool.

In terms of their role in nutrient cycling and impact on the Lakes, these different fractions of TN and TP differ primarily in the time scales on which they become available within the Lakes. The refractory components break down on time scales that are comparable to or longer than flushing times: increases in their relative contribution to loads will increase export, and reduce and delay impacts in the Lakes. Conversely, labile detrital fractions break down on time scales of 10 days, and so contribute to algal biomass and production with a lag of this order. The dissolved inorganic nutrient pools, and the particulate inorganic P pool, are available almost immediately to support algal blooms (PIP equilibrates with DIP on a time-scale of 1 day).

From the point of view of the model behaviour, it is the total loads into each of L. Wellington and L. King that matter, not the individual river loads. The daily TN loads into L. King vary by over 5 orders of magnitude through the calibration period, while those into L. Wellington vary by over 2 orders of magnitude. This makes plots of daily load hard to interpret. We have found plots of cumulative load over the 1995-99 period to be more useful.

Fig. 5, 6 and 7 show cumulative loads of the various N and P fractions into L. Wellington, L. King and L. Victoria over the 1995-99 period. Note that because the organic N is split equally between DON and Labile Detrital N (LDN), the latter is equal to and masked by the DON in these plots.

Over the 4-year period, the cumulative loads of N and P fractions into L. Wellington are about 2 to 3 times those into L. King. Loads into the Gippsland Lakes occur primarily in the winter and spring of each year. The loads into L. Wellington are dominated by the winter-spring wet seasons of 1995 and 1996, with a very significant contribution from a single flood event in 1995. The loads into L. King are dominated by the June 1998 flood event, whereas this event provides a relatively small proportion of total N and P loads into L. Wellington. The wet season in 1997 failed to deliver appreciable runoff and loads to either system. The estimated coastal catchment loads into L. Victoria are an order of magnitude lower than those from the major river catchments into L. Wellington and L. King.

Nitrate represents about 40% of the cumulative TN load into L. Wellington, but only about 20% of the cumulative TN load into L. King. Nitrate does constitute a larger fraction of the TN load into L. King in the June 1998 flood event. PIP loads in L. King are also disproportionately associated with this flood event. DIP loads make up about 25% of the TP load into L. Wellington, but only about 15% of the TP load into L. King. The persistent background loads of N and P into L. Wellington are associated primarily with the Latrobe and Thomson Rivers, whereas the Avon River, which is unregulated, makes a much larger contribution to the flood event loads in 1995 and 1998. TSS loads occur disproportionately in the major flood events, especially in L. King.
The TN:TP ratio over the 4 years is 8.7 (by weight) or 19:1 by moles in L. Wellington, and 11.1 by weight, or 24:1 by moles in L. King, compared with a Redfield ratio of 16:1 by moles for phytoplankton. The TP:TSS ratio is about 1.6 mg P / g TSS in L. King, and 1.5 mg P / g TSS in L. Wellington.

**Fig. 5, 6 & 7. Cumulative N, P and TSS loads into L. Wellington, L.King, and L.Victoria from July 1995 to June 1999.**
4.3 Surface Irradiance

For ease rather than using local solar irradiance data, we have used data collected in Port Phillip Bay and used to drive the Port Phillip Bay model (Fig. 8). As Port Phillip Bay and the Gippsland Lakes are at similar latitudes and have similar climate patterns, we expect the seasonal pattern and day-to-day variability to be similar.

![Fig. 8. Plot of mean daily surface irradiance (W m⁻²) used for model calibration.](image)

4.4 Temperature

For the calibration runs, water temperature throughout the Lakes is assumed to vary seasonally between 10 and 20 °C (Fig. 9), with a minimum in late June. This provides a good approximation to the observed seasonal cycle in temperature (see Hydrodynamic Model Report).

![Fig. 9. Seasonal variation in temperature used in the calibration runs.](image)
4.5 Bottom Stress

We have no model-based estimates of bottom stress for the Gippsland Lakes. We can expect bottom stress over much of L. King and L. Victoria to be very weak, due to weak tidal currents and strong density stratification, except during periods of high wave energy. In the model runs presented here, a constant normalised excess bottom stress (nxsbs) has been imposed over time across all boxes, and its value tuned to reproduce observed suspended sediment loads. This is discussed further under calibration of TSS.

4.6 Bass Strait Boundary Conditions

The Bass Strait boundary concentrations for model variables are shown in Table 1. With the exception of DON, boundary concentrations are generally low compared with concentrations in the Lakes, and have little effect on model behaviour, especially given that exchanges between the Lakes and Bass Strait are very low for much of the time.

Table 1. Bass Strait boundary concentrations for model variables.

<table>
<thead>
<tr>
<th>Model Variable</th>
<th>Bass Strait Value.</th>
</tr>
</thead>
<tbody>
<tr>
<td>TSS_unfloc</td>
<td>0.0 kg m(^{-3})</td>
</tr>
<tr>
<td>TSS_floc</td>
<td>0.0 kg m(^{-3})</td>
</tr>
<tr>
<td>Large_Phyto_N</td>
<td>2.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>Nano_Phyto_N</td>
<td>3.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>Dinoflag_N</td>
<td>0.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>Nodularia_N</td>
<td>0.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>Zoo_N</td>
<td>7.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>MicroZoo_N</td>
<td>2.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>NH(_3)</td>
<td>5.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>NO(_3)</td>
<td>5.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>DIP</td>
<td>15.0 mg P m(^{-3})</td>
</tr>
<tr>
<td>PIP_unfloc</td>
<td>1.0 mg P m(^{-3})</td>
</tr>
<tr>
<td>PIP_floc</td>
<td>1.0 mg P m(^{-3})</td>
</tr>
<tr>
<td>DIC</td>
<td>24000 mg C m(^{-3})</td>
</tr>
<tr>
<td>DOC</td>
<td>0.0 mg C m(^{-3})</td>
</tr>
<tr>
<td>DON</td>
<td>150 mg N m(^{-3})</td>
</tr>
<tr>
<td>DOP</td>
<td>0.0 mg P m(^{-3})</td>
</tr>
<tr>
<td>Ref_det_C</td>
<td>0.0 mg C m(^{-3})</td>
</tr>
<tr>
<td>Ref_det_N</td>
<td>0.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>Ref_det_P</td>
<td>0.0 mg P m(^{-3})</td>
</tr>
<tr>
<td>Lab_det_Plank</td>
<td>0.0 mg N m(^{-3})</td>
</tr>
<tr>
<td>Lab_det_Benth</td>
<td>0.0 mg N m(^{-3})</td>
</tr>
</tbody>
</table>
5 Model Calibration

As discussed above, recent modifications to the model have been designed to reduce the number of tunable ecological parameters, although there are still a number of parameters, especially those related to zooplankton dynamics, which are semi-empirical system properties. The parameter values used as a starting point for calibration of the Lakes model were identified in the Port Phillip Bay Environmental Study (PPBES) (where there was a very comprehensive set of process studies), and have since been applied successfully to a number of other estuaries. This has boosted confidence in both the model and parameter values, and the model with this parameter set is currently being used as the basis for a broad analysis of Australian estuaries as part of the National Land and Water Resources Audit.

Nonetheless, there are aspects of the Gippsland Lakes that differ from other test cases, and have required adjustment of model parameters and processes, and the introduction of some new processes. The model simulations used for calibration have been carried out for a period of 12 years, with the physical exchanges, runoff and loads for the period July 1995 to June 1999 repeated three times in succession. Model simulations require the specification of initial conditions for all model variables, some of which are poorly known. Some sediment variables change on long time scales, and their initial conditions can influence model behaviour for some years. Running the model over 12 years reduces the effect of initial conditions on the model behaviour. Model predictions for the last two years of these runs are compared with observations in the calibration period July 1997 to June 1999.

In order to provide a context for the calibration, we begin with an overview of the behaviour of the Lakes in the calibration period, based on the MAFRI and EPA data sets. We then discuss the calibration of the key model processes. Both sections refer to a complete set of comparison plots of model predictions and observations at MAFRI and EPA sites over the calibration period (Fig. 10 - 48).

(Note that we have compared the observations at sites 2316/18 in northern L. King with our northern L. King model box 7, even though it lies within our main L. King box 2. The model predictions for box 2 are shown in comparison plots against sites 2314/19 in southern L. King. The coarse model resolution does not allow exact correspondence between model boxes and sites.)

5.1 Observed response of the Lakes to nutrient and sediment loads during the calibration period

As discussed above, the calibration period follows two relatively “normal” years with slightly above average wet season flows. The wet season failed in 1997, and the period July 1997 to May 1998 is characterised by unusually low runoff and loads in all catchments. There are intense runoff events in June, September and November 1998, the June flood event in the eastern catchments being unusually large. These runoff events are followed by a typical dry summer and autumn in 1999. These strong temporal contrasts, along with the physical contrasts among the Lakes, provide a particularly powerful data set for calibrating the dynamical response of the Lakes to catchment loads. To set the scene for the model calibration, we first summarise the biogeochemical and ecological response of L. Wellington, L. Victoria and L. King, as reflected in the MAFRI and EPA data sets.
5.1.1 L. Wellington

Exchanges between L. Wellington and L. Victoria are very small except during high runoff events, when the flow is from L. Wellington into L. Victoria (see Hydrodynamic Model Report, Walker and Andrewartha, 2000). As a result, the behaviour of L. Wellington can be treated as a first approximation as independent of the other Lakes.

Runoff and loads reach L. Wellington from the Latrobe, Thompson and Avon Rivers. The Latrobe has higher base flow than other rivers into the Gippsland Lakes, but lower peak loads. It dominates total nutrient loads into L. Wellington over the 1997-99 period. Because of the Latrobe, loads into L. Wellington in the dry period July 1997 to May 1998 are still substantial, about 400 t TN and 50 t TP (Fig. 5). The Avon contributes most of the peak flow and load into L. Wellington in the June 1998 runoff event.

L. Wellington is relatively shallow (mean depth 2.6 m) and well-mixed, so it does not show the effects of vertical stratification which dominate the other Lakes (see below). Surface and bottom concentrations are virtually indistinguishable (Fig. 10-12).

Mean NOx concentrations in runoff into L. Wellington over the calibration period are around 220 mg m$^{-3}$, and peak concentrations exceed 1000 mg m$^{-3}$, but the NOx and NH$_3$ concentrations in surface waters remain very low, around 1 or 2 mg m$^{-3}$ (Fig. 10). This is almost certainly due to uptake of nitrate by phytoplankton. Even at the time of the June 1998 runoff event, NOx is only briefly elevated and reaches much lower peak concentrations than in L. King or L. Victoria.

DIP levels in L. Wellington are typically around 5 to 20 mg m$^{-3}$, and rarely appear low enough to be limiting to phytoplankton growth (Fig. 10). If anything, DIP concentrations are reduced following the June runoff event. The maintenance of comparatively constant moderate levels of DIP in the face of phytoplankton uptake and varying runoff strongly suggests that DIP levels are buffered, presumably by exchange with suspended and bed sediments (see below).

Chl a is persistently high in L. Wellington (Fig. 11). It is lower, around 10 mg m$^{-3}$, in the dry period prior to June 1998, and reaches 50 mg m$^{-3}$ in the spring and early summer following the event, before declining back to around 10 mg m$^{-3}$ in the following autumn.

The background levels of organic N (ie TN - DIN) (Fig. 11) are around 500 mg m$^{-3}$ prior to the runoff event, but peak at around 1200 mg m$^{-3}$ following the runoff. This increase parallels the increase in Chl a, but the increased phytoplankton biomass accounts for only about 30% of the increase in organic N. Background levels of non-PO$_4$ P (ie TP - DIP) are around 50 mg m$^{-3}$, but increase to around 180 mg m$^{-3}$ in association with the phytoplankton bloom (Fig. 11). The non-PO$_4$ P pool includes both adsorbed PIP and organic P.

Dissolved oxygen remains close to saturation in L. Wellington (Fig. 12), presumably because of strong vertical mixing and exchange with the atmosphere.

L. Wellington is highly turbid. Background suspended solids (NFR) decline to about 10 g m$^{-3}$ prior to the June runoff event, but a peak level of 100 g m$^{-3}$ is observed following the runoff event (Fig. 12). Secchi depths are very low, typically about 0.5 m, and dropping to 0.1 m following the runoff events (Fig. 12). For such high levels of light attenuation, we can approximate the mean light in the surface layer by I$_0$/Kd.z, where I$_0$ is surface irradiance, z is depth and Kd is the light attenuation coefficient. For the Secchi depths typically observed in L. Wellington, and a 3 m water column, the mean water column irradiance is 2 to 10% of surface irradiance. This may reduce but will not prevent phytoplankton growth. On the other hand, bottom light levels, given by I$_0$.exp(-Kd.z), will be less than 0.01% of surface irradiance at 3 m, preventing growth by benthic plants.
To summarise, L. Wellington appears to behave like a large phytoplankton chemostat, with high concentrations of nitrate in the inflow being immediately taken up by phytoplankton. The increased load of nutrients in the winter and spring of 1998 is reflected primarily in elevated phytoplankton biomass. The Lake is dominated by phytoplankton rather than macroalgae because of the high turbidity and resulting light attenuation. The mean TN concentration in the inflow is about 900 mg m$^{-3}$, but the mean TN concentration in the Lake is only about 600 mg m$^{-3}$. It appears that a substantial fraction of the TN load is trapped within the Lake and lost either to burial in sediments or to denitrification. The mean TP concentration in the inflow is around 130 mg m$^{-3}$, but the mean TP concentration in the water column is around 70 mg m$^{-3}$, suggesting that almost half of the TP load is sequestered within the Lake.

5.1.2 L. King

L. King receives inputs from three rivers (Mitchell, Nicholson and Tambo), with the Nicholson being a relatively minor source. The L. King catchments are less modified, and flow is relatively unregulated, so runoff and loads are concentrated in large events, with low base flow (Fig. 6). During and in the months following these events, it appears that water quality in L. King is predominantly controlled by these local inputs, although it nominally lies downstream of L. Wellington and L. Victoria.

The MAFRI 1997-99 nutrient stations include one station in northern L. King (site 18), and one in southern L. King (site 19). Observations from these two sites are quite similar. The EPA sites 2316 and 2314 correspond to MAFRI sites 18 and 19 respectively. EPA also sampled at site 2322 off Shaving Point, which lies in the area treated by the model as the Entrance box. We have no nutrient or chlorophyll data in Jones Bay, which is shallow, cut-off by the Mitchell promontory, and might be expected to behave rather differently from the main body of L. King. Observations from sites 2314/19 and 2316/18 are shown in Fig. 31-36, and 37-42 respectively.

During the calibration period, runoff is low prior to June 1998: only 10% of the 2-year discharge occurs in that period. About 30% of the 2-year discharge occurs in the June 1998 runoff event, with about equal contributions in that event from the Tambo and Mitchell. Over the two years, the Mitchell contributes around 70% of the discharge, with the Nicholson only about 5%. Northern L. King is stratified throughout most of the calibration period, although salinity differences between surface and bottom layers approach zero in March 1998, following the long dry period. Southern L. King is more strongly and persistently stratified.

Only about 5% of the cumulative TN and TP load occurs prior to June 1998 (Fig. 6). About 50% of the cumulative TN load, and 60% of the TP, DIP and NOx loads, occur in the June 1998 runoff event. A further 10% of the TN and TP loads occur in a September 1998 runoff event. Only about 5% of TN and TP loads occur after October 1998, so about 90% of loads into L. King in the calibration period occur in the period from June to October 1998.

Surface concentrations of nitrate and ammonia in L. King are typically very low, around 1 mg m$^{-3}$ (Fig. 31, 37), quite low enough to limit phytoplankton growth. Very large transient peaks in surface nitrate, up to 800 mg/m$^3$, occur during the runoff event in June 1998. However, these concentrations are quickly reduced to low background levels.

Bottom ammonia concentrations are mostly relatively low, up to 20 mg m$^{-3}$, during the dry period prior to the run-off event, but reach very high levels, up to 200 mg m$^{-3}$, immediately following the event, and even higher levels, up to 400 mg m$^{-3}$, in the summer and autumn of 1999 (Fig. 32, 38).
The ammonia in bottom waters must be produced by regeneration either within bottom waters or in sediments. Nitrate in bottom waters remains very low, suggesting that nitrification in bottom waters and sediments is minimal, or is coupled directly to denitrification (Fig. 32, 38).

Phosphate surface values are generally moderate, around 10 to 20 mg m\(^{-3}\), and bottom values are similar prior to June 1998, and high (up to 200 mg m\(^{-3}\)) after June 1998 (Fig. 31, 32, 37, 38). As in L. Wellington, surface DIP concentrations seem less likely to limit phytoplankton growth than DIN. High DIP concentrations in bottom waters coincide with periods of oxygen depletion, strongly suggesting that oxygen controls DIP release from sediments.

Prior to June 1998, chlorophyll is generally low, around 1 to 3 mg m\(^{-3}\) (Fig. 33, 39). Following the June runoff event, a phytoplankton bloom develops over several weeks, consuming surface nitrate and reaching a peak of 80 mg Chl a m\(^{-3}\) in southern L. King, but only 30 mg m\(^{-3}\) in northern L. King. This bloom decays quite rapidly, although it is followed in the spring by smaller secondary blooms in the 10 to 20 mg m\(^{-3}\) range. Chlorophyll peaks of around 15 to 30 mg m\(^{-3}\) in March 1999 are associated with a \textit{Nodularia} bloom.

Background organic N levels in L. King are around 300 to 400 mg m\(^{-3}\) (Fig. 33, 39). Organic N increases following the June 1998 runoff event to about 800 mg m\(^{-3}\), but declines to background levels by December. During the \textit{Nodularia} bloom in summer/autumn 1999, there are very large concentrations of organic N in surface waters, up to 1000 mg m\(^{-3}\).

In surface waters, non-PO\(_4\) P is relatively constant at around 10 mg m\(^{-3}\), with some higher values following the June 1998 runoff event (Fig. 33, 39). Values in bottom waters are somewhat higher, around 30 mg m\(^{-3}\) (Fig. 34, 40). However, increases in non-PO\(_4\) P in anoxic bottom water are relatively small when compared with the very large increases in bottom water DIP.

MAFRI measured sediment nutrient concentrations (in the top 2 cm) at sites 18 and 19 in L. King. NH\(_3\) in pore waters was lower (around 5 mM) prior to June 1998, but increased substantially after June 1998 to peak in April 1999 at around 12 mM. The phosphate concentration was low (< 0.5mM) up until January 1999, but increased substantially to 2 mM in March 99. Phosphate concentrations actually decreased during the period immediately following the June 1998 flood event.

MAFRI also measured benthic fluxes at site 19. DO consumption in dark chambers was typically around 20 to 40 mmol m\(^{-2}\) d\(^{-1}\). NH\(_3\) release was low, around 0 to 2 mmol m\(^{-2}\) d\(^{-1}\), prior to June 1998, but increased to around 6 mmol m\(^{-2}\) d\(^{-1}\) after the flood event. NO\(_x\) fluxes were generally close to zero. The PO\(_4\) flux was very low up until June 1998, but large effluxes, around 1 mmol m\(^{-2}\) d\(^{-1}\), were observed in November 1998 and February 1999. Denitrification efficiencies ranged from 23 to 57\%, with minimum values in winter. Overall, there was no obvious relationship between denitrification efficiency and sediment respiration rates. At the observed efflux rates, it would take about 30 days to build up the maximum observed ammonia and phosphate concentrations in bottom waters in August 1998 and March 1999.
L. King is substantially less turbid than L. Wellington. Prior to the June 1998 runoff event, observed suspended matter (NFR) concentrations in surface waters were low, around 3 to 4 g m$^{-3}$, but increased to around 10 g m$^{-3}$ after the event (Fig. 35, 41). Values in surface and bottom waters were quite similar (Fig. 35, 36, 41, 42). NFR in L. King is 3 to 10 times lower than in L. Wellington (Fig. 12).

In the calibration period, Secchi depths were around 3 to 4 m prior to June 1998, but decreased to 0.65 m in July after the event, and remained around 1.2 m up until April 1999 (Fig. 35, 41). For the Secchi depths observed under “dry” conditions prior to June 1998, the 1% light level is around 7 to 8 m, close to bottom depths. This means microphytobenthic growth may be possible on the bottom, and phytoplankton growth should be possible in both surface and bottom layers. For Secchi depths of around 1 m, the 1% light depth is around 2 m, phytoplankton growth will be severely light limited in the bottom layer, and growth by benthic plants below 4 m virtually eliminated.

To summarise, the behaviour of L. King in the calibration period is dominated by two factors: loads are primarily delivered in intense runoff events, and the estuary remains strongly stratified most of the time. The intense runoff events produce large transient nitrate peaks and algal blooms. Subsequent remineralization results in high ammonia levels in bottom layers, but high light attenuation prevents phytoplankton growth there, and phytoplankton in surface layers are dependent on physical or biological mechanisms to transport ammonia to the surface layer.

As in L. Wellington, DIP in surface waters appears to be strongly buffered, but this buffering appears to break down when bottom waters become anoxic following runoff events, and DIP accumulates to high levels there. The N-fixing cyanobacterial (*Nodularia*) bloom in the summer and autumn of 1999 produces very high organic nitrogen concentrations in the water column, and results in high ammonia and DIP concentrations in bottom waters. There is indirect evidence that, under prolonged dry conditions, increased light penetration may allow algal blooms in bottom waters.

5.1.3 L. Victoria

The model now includes estimated loads from coastal catchments into L. Victoria (Fig. 7). Although these are small compared with loads into L. Wellington and L. King, they may have significant impacts, especially during periods of lower runoff when western L. Victoria at least is poorly flushed. During high flow events, western L. Victoria receives the outflow from L. Wellington. There is one EPA site in central L. Victoria (2311) and 2 MAFRI sites: a deep site 4 in eastern L. Victoria, and a shallow site 8 (around 4 m deep) in western L. Victoria, close to McLenann Strait. Site 4 in eastern L. Victoria behaves very similarly to sites 18 and 19 in L. King (Fig. 25-30). It appears that exchange between L. King and eastern L. Victoria is sufficiently rapid so that they behave more or less as one water body.

Site 8 in western L. Victoria behaves rather differently from Site 4, perhaps because of its proximity to L. Wellington, but most likely because it is shallow and less strongly stratified than eastern L. Victoria and L. King. Nitrate and ammonia levels in surface waters are again very low, and dominated by the transient peak associated with the June 1998 runoff (Fig. 13). However, chlorophyll values in western L. Victoria are generally higher, both before and after the runoff event (Fig. 15). This probably reflects a more efficient transfer of regenerated nutrients from sediments to the surface layer. Bottom ammonia concentrations are also much lower at this site, again suggesting increased vertical exchange or phytoplankton uptake (Fig. 14). Bottom oxygen depletion and DIP build-up are also less pronounced (Fig. 14, 18).
Secchi depths at site 2311 in central L. Victoria are around 2 m prior to the flood, and around 1 m afterwards (Fig. 23). Even for values around 2 m, phytoplankton growth in the bottom layer is likely to be severely light-limited.

To summarise, eastern L. Victoria behaves very similarly to L. King, and most of the comments made above about L. King also apply there. Western L. Victoria is influenced by runoff into both L. King and L. Wellington. Because it is shallow and weakly stratified, it also behaves dynamically as an intermediate between L. Wellington and L. King.

5.2 Calibration of Key Model Processes

Although we are implementing and calibrating an integrated physical-biogeochemical-ecological model, it is (fortunately) true that not all components of the model interact equally strongly with all other components. In many cases, it is possible to identify processes or components of the model that can be treated and calibrated almost in isolation. The calibration process is most straightforward when we have process studies or observations that deliver information directly and independently about model processes and parameters. Examples of this include the experiments and analysis reported by Webster and Grace (2001) on P adsorption isotherms, discussed earlier, and the statistical analysis of light attenuation and optical constituents presented below. However, unlike the Port Phillip Bay Environmental Study, the Gippsland Lakes Study to date has been primarily a modelling study. There has been relatively little effort expended on focused process studies in the Lakes, so that calibration of the model against the MAFRI and EPA time series assumes particular importance.

This approach does not treat model calibration as a purely statistical problem, or the model as a large “black box”, with a large number of free parameters. Rather, the model system is treated as a set of interacting subsystems, whose behaviour and dependence on parameters can often be approximated by simple relationships. The role of numerical simulation and experiment in the calibration process is partly to confirm that these simple approximate descriptions of subsystem behaviour (or assumptions that interactions among subsystems can be neglected) are reasonable. Thus the calibration process has the dual function of improving understanding of the model system behaviour and dependence on forcing and parameters, and identifying a model parameterization that reproduces the observations.

Accordingly, we have structured the discussion of model calibration by processes or subsystems, emphasizing the key parameters and relationships that determine behaviour within subsystems, and the interactions among subsystems.

5.2.1 Light Attenuation

The observations show strong light attenuation (low Secchi depths), especially in L. Wellington (Fig. 12). Turbidity due to suspended sediments, and high chlorophyll concentrations, clearly contributes to light attenuation. However, even in quite turbid inland waters, coloured dissolved organic matter (CDOM) can also play a major role. Kirk (1983) has measured high attenuation due to coloured dissolved organic matter (CDOM) in the Gippsland Lakes.

Direct bio-optical measurements in the lakes would separate contributions to attenuation from dissolved and particulate fractions, and phytoplankton pigments. In the absence of such measurements, we have turned to the long-term EPA data set (1986 to 1999), which includes simultaneous measurements of Secchi depth, chlorophyll, suspended sediment (NFR), turbidity (NTU) and organic N (TKN - NH₃). The EPA measurements did not separate...
dissolved and particulate organic nitrogen, and in any case DON might not be a good measure of CDOM. However, if CDOM enters in freshwater from rivers, and is relatively refractory (both reasonable assumptions), then we might expect CDOM concentration to be proportional to the freshwater fraction (FW), computed as 1 - S/35, where S is salinity, and 35 is taken as marine salinity.

We assume that the bulk attenuation coefficient for downwelling irradiance, $K_d$, depends linearly on the concentrations of constituents contributing to attenuation. We have estimated $K_d$ as $2/($Secchi depth$)$, and carried out multiple linear regressions of $K_d$ on various combinations of Chl a, NFR, Turbidity, Organic N, and FW. The intercept was fixed at zero (background $K_d$ values due to pure water are negligible compared with values observed in the Gippsland Lakes). We have treated NFR and Turbidity (NTU) as alternative measures of suspended particulates, and have included one but not both in regressions.

For all lakes, there are 401 observations where $K_d$, NFR, Chl a, FW and organic N are available. A regression of $K_d$ on all four independent variables yielded an $R^2$ of 0.68, but the coefficient for Chl a was negative. When Chl a was dropped from the regression, the coefficient for organic N became negative. It appears that, for the Lakes as a whole, suspended sediment and CDOM dominate light attenuation to such an extent that contributions from Chl a and organic matter cannot be distinguished (despite the high Chl a concentrations observed at times). The coefficients for NFR and FW/CDOM were stable and highly significant at about 0.07 (m$^2$ g$^{-1}$) and 2.0 (m$^{-1}$ in freshwater) respectively.

L. Wellington is typically much more turbid than the other Lakes, and is dominated by fine suspended sediment. One might expect specific attenuation coefficients for NFR to differ between L. Wellington and the other lakes. One might also hope that, with a less dominant contribution from sediment, it might be possible to estimate specific attenuation coefficients for Chl a and organic N in the other Lakes.

There were 79 complete sets of observations in L. Wellington. As before, the regression had to be restricted to NFR and CDOM. The $R^2$ was lower (0.64) and the residual standard deviation was high, at 2.49. The coefficients were 0.075 (m$^2$ g$^{-1}$) for NFR and 2.7 m$^{-1}$ for CDOM in freshwater respectively.

There were 323 complete sets of observations in the other Lakes. In the regression with all 4 variables, Chl a and organic N were confounded. Dropping organic N as an independent variable resulted in coefficients of 0.02 m$^2$ mg Chl a$^{-1}$, 0.0157 m$^2$ g NFR$^{-1}$, and 1.73 m$^{-1}$ for CDOM in freshwater. These were all highly significant ($P < 1.E-4$). The $R^2$ was 0.54, and the residual standard deviation 0.70. The specific attenuation coefficient for Chl a is very close to theoretical values.

The CDOM attenuation for other Lakes is lower than that estimated for L. Wellington, but not too different. However, the estimated specific attenuation coefficient for NFR in L. Wellington is 4 times that estimated for the other lakes. This is consistent with the observation that L. Wellington is dominated by fine particulates, which are likely to have a greater optical cross-section per unit mass.

These regressions were repeated with NFR replaced by turbidity. There were only 179 complete sets of observations including turbidity for all lakes, with 35 in L. Wellington, and 144 in other lakes. In regressions on these data, Chl a and organic N were again confounded, and organic N was dropped from the regressions.

Turbidity provides a more consistent predictor of $K_d$ than NFR across all lakes. The NTU-specific attenuation coefficient differs by only 40% between L. Wellington (0.064) and other lakes (0.105). This is consistent with the hypothesis that fine particles in L. Wellington are
efficient scatterers, contributing even more to optical backscatter than to bulk attenuation. In
general, $R^2$ were higher, and residual standard deviations lower, using turbidity rather than
NFR. The estimated specific attenuation coefficients for Chl a were 0.020 and 0.012 m$^2$ mg
Chl a$^{-1}$ in L. Wellington and other lakes respectively.

Turbidity (NTU) may be a better predictor of $K_d$ than NFR, but our models predict NFR, not
NTU. Our best estimates of the specific attenuation coefficient for suspended solids (NFR)
are about 0.075 for L. Wellington (fine sediment) and 0.016 for other Lakes. We have used a
compromise value of 0.030 m$^2$ g$^{-1}$ in the model.

Estimates of attenuation due to CDOM in freshwater were around 2.5 m$^{-1}$ in L. Wellington,
and 1.5 m$^{-1}$ in other Lakes. We have used an average value of 2 m$^{-1}$ in the model. Differences
between the Lakes may reflect genuine differences in CDOM loads in river inputs, or
magnification of the absorption by CDOM due to greater scattering in L. Wellington. Kirk
(1983) reports values of absorption at 440 nm by dissolved fractions in L. King, L. Victoria
and L. Wellington, and in the Latrobe R flowing into L. Wellington, of 0.58, 0.65, 1.14 and
1.89 m$^{-1}$ respectively. To convert these into contributions to $K_d$, they must be divided by the
mean cosine of the downwelling light field, and appropriately averaged across the 400 to 700
nm spectrum. His observations in the Latrobe River are roughly compatible with our
regression estimates here. Because he does not report salinities, we do not know the
freshwater fractions corresponding to his measurements in the Lakes.

Estimates of specific attenuation by Chl a obtained here were generally consistent with
literature values, and we have therefore retained the CR model’s bio-physical
parameterisation of light attenuation by phytoplankton. Estimates of the specific attenuation
due to organic N are confounded in this data set, as organic N tends to co-vary with Chl a, and
we cannot distinguish the fraction of organic N due to living phytoplankton. We have
therefore retained the values used in Port Phillip Bay.

This direct calibration of the light attenuation sub-model avoids confounding the specific
attenuation coefficients with other model processes. Assuming that the specific attenuation
coefficients have been correctly estimated, the model’s ability to predict light attenuation
(Secchi depth) will depend on its ability to predict the concentrations of bio-optical variables
(TSS, CDOM/salinity, Chl a, organic N).

The full model predicts the background levels of Secchi depth in L. Wellington quite well,
although it fails to predict the full extent of the decline following high runoff (Fig. 12). The
correspondence with observations in L. Victoria (2311), L. King (2314, 2316) and the
Entrance (2322) is also very encouraging (Fig. 23, 35, 41, 47).

### 5.2.2 Suspended sediment concentrations and sediment burial

In the model, the predicted TSS concentrations in the water column represent a balance
between loads from catchments, settling and resuspension. The model also incorporates a
salinity-dependent conversion from a fine, unflocculated sediment, TSS_unfloc, to a rapidly
sinking fraction TSS_floc, so predicted TSS concentrations also depend on flocculation.
Because resuspension fluxes depend on the amount of TSS in the bed, the predicted water
column concentrations also depend on bed sediment budgets.

The suspended sediment model is rather poorly constrained a priori, as we do not have direct
estimates of sinking rates and critical shear stresses for sediment fractions, nor confirmation
of salinity thresholds and rates for the flocculation process. As well, we do not have
dynamical estimates of bottom shear stress.
In the earliest calibration runs, we attempted to predict bottom shear stress from wind speed, but this produced rapid fluctuations in TSS that did not match observations. (Unfortunately we only have TSS observations in the EPA data sets, at 2-month intervals. These could be badly aliased, but appear to suggest a fairly smooth behaviour in TSS over time in response to varying loads.) In any case, predictions of bottom stress based on surface wind stress are unlikely to be valid in L. King and L. Victoria, which are strongly stratified. We have therefore resorted, at least temporarily, to applying a constant uniform bottom shear stress. In the absence of catchment loads, this should result in the prediction of a background TSS concentration for which the sinking flux matches the resuspension flux.

The model response is complicated by flocculation. Unflocculated sediment is assumed to sink very slowly, at 0.1 m d\(^{-1}\). In L. Wellington, during and after high runoff periods, salinities are sufficiently low (Fig. 5) that sediment remains unflocculated, and about half the sediment load remains in suspension. However, during dry periods, salinities increase to the point where flocculation occurs. In L. Victoria and L. King, salinities are almost always high enough to allow flocculation (Fig. 5).

In the original parameter set, the conversion rate parameter \( r_{floc} \) was set to 1 d\(^{-1}\), and the resulting flocculated sediment sank quickly (5 m d\(^{-1}\)). This resulted in complete and rapid flocculation of sediment in L. Wellington during high salinity periods, and in L. Victoria and L. King at all times. However, this choice of parameters is not compatible with observations. For constant bottom stress and resuspension rates, the predicted ratio of TSS\(_{unfloc}\) and TSS\(_{floc}\) concentrations in the water column is the inverse of the ratio of sinking velocities. Suspended sediment concentrations decrease in L. Wellington during dry periods, but by only a factor of about 5, not 50 (Fig. 12).

The problem is worse in L. Victoria and L. King. These lakes are strongly stratified, and, in the model, resuspension occurs primarily in the bottom layer. Although there is very little vertical exchange between the two layers, observed suspended sediment concentrations in both layers are similar (Fig. 23, 24, 35,36, 41,42). These observations are not compatible with a rapidly sinking sediment fraction.

It is clear from the observations that a fine, slowly sinking sediment fraction must be present both in L. Wellington during dry saline periods, and in L. Victoria and L. King. It is possible that only part of the fine sediment load is subject to flocculation. Rather than introduce another TSS fraction, we have reduced the flocculation rate parameter \( r_{floc} \) from 1 d\(^{-1}\) to 0.01 d\(^{-1}\). This results in the retention of a fine sediment fraction even during prolonged dry periods.

Model experiments suggest that some kind of removal process for fine sediments through flocculation and settling is important. If flocculation is set to zero, suspended sediment concentrations in L. Victoria and L. King are much too high.

The normalised excess bottom shear stress, \( nxsbs \), which controls resuspension rates, has been adjusted to try to reproduce the observed TSS values at EPA sites. This parameter controls the depth of sediment which is resuspended per unit time. The flux of fine unflocculated sediment which is resuspended then depends on the TSS\(_{unfloc}\) concentration in the bed. In the model, the TSS pool in the bed is controlled by a balance between deposition, resuspension, and burial. Burial proceeds at a prescribed specific rate \( r_{bury,TSS} \), which is set to 0.001 d\(^{-1}\). This results in a TSS pool size in each basin, which is controlled by a long-term balance between TSS loads, export and burial. The bed sediment TSS pool plays an important role in P dynamics, and the burial rate has been determined partly with this in mind (see the discussion of P dynamics below).
For a given TSS pool in the bed sediment, the suspended sediment concentration during periods of low flow and load is controlled by a balance between sinking and resuspension. One can increase the suspended sediment concentration by increasing nxssbs, or decreasing sinking rates. Low sinking rates and low resuspension rates tend to match the observations in L. King and L. Victoria better, as the observations there show little contrast between surface and bottom layers. The long-term response of the model to changes in sinking rate is a little more complex however, as the bed sediment pool is also affected by a balance between sequestration and export. Thus, decreasing sinking rates will tend to increase suspended sediment concentrations and consequently the fraction of the TSS that is exported, and therefore decrease the bed sediment pool size and the resuspension flux. This negative feedback reduces the sensitivity to both nxssbs and sinking rates.

With nxssbs set to 0.01, the model over predicts background TSS concentrations in L. Wellington slightly (Fig. 12) and in the bottom layers in L. Victoria significantly (Fig. 24), but predicts surface values in L. Victoria and L. King, and bottom values in L. King, quite well (Fig. 23, 24, 35,36, 41,42). This behaviour might be improved by tuning bottom stresses separately in each basin, and potentially by taking account of resuspension in shallow areas along the Lake edges. The model also seems to over predict peak TSS concentrations associated with the June 1998 flood event in all locations. It is likely that a substantial fraction of the TSS load in the flood events consists of coarser material that is rapidly deposited.

The sediment dynamics represented in the box model are relatively crude, and intended primarily to allow the effects on light attenuation and P adsorption to be captured. If one wished to model sediment dynamics in the Lakes in a realistic manner, it would be necessary to develop a coupled hydrodynamic – sediment model, with fine spatial resolution. Data on sediment size spectra and settling rates in both the rivers and the Lakes, and data on critical shear stresses and consolidation for bed sediments, would also be required.

5.2.3 Adsorption-desorption and burial of Inorganic P

As noted above, observations of relatively constant levels of DIP in surface waters, despite large fluctuations in loads and phytoplankton biomass, suggest that DIP is strongly buffered by exchanges with adsorbed inorganic P on suspended or bed sediments. In the model, predicted DIP levels in the water column represent a balance among loads from catchments, adsorption-desorption exchanges with suspended sediments, exchange with bed sediments, and phytoplankton uptake and remineralization in the water column. We focus here primarily on the abiotic interactions. Biological exchanges are of course important, and we discuss those further in a subsection below. For now we note that N rather than P appears to be limiting to phytoplankton growth in surface waters almost all of the time. To the extent that P is present in excess to biological demand, we might expect its concentration to be influenced strongly by abiotic processes.

In early model versions, we used a linear isotherm to describe P adsorption-desorption. We noted in the interim model report that it was possible to buffer water column DIP in the observed concentration range, but only if we assumed unusually high values of the adsorption coefficient. We have now implemented the Freundlich isotherm recommended by Webster and Grace (2001), with their recommended exponent $P_{\text{ads, exp}} = 0.34$. In initial model runs, using their recommended values for the adsorption coefficients of 107 in the water column and 74 in sediments, the predicted values of water column DIP were much too high.

As for TSS, the predicted water column DIP concentrations for given loads depend not only on the short-term exchanges within the water column, but on the long-term accumulation and
dynamics of P in bed sediments. In considering these in more detail, it is helpful to treat L. Wellington and the other Lakes separately.

In L. Wellington, the water column is well-mixed and remains oxygenated, so that the surface sediment layer remains oxic, and there are no short-term releases of DIP from sediments driven by anoxia. Instead, there is a steady-state established among PIP, TSS and DIP concentrations in the sediment, according to the Freundlich isotherm:

\[
\frac{\text{PIP}}{\text{TSS}} = 74 \cdot \text{DIP}^{0.34}
\]

As PIP and TSS pools in sediments are large and change slowly, we can treat DIP concentrations in pore waters as being driven by the accumulating PIP and TSS pools:

\[
\text{DIP(sed)} = 3.2\times10^{-6} \cdot (\frac{\text{PIP}}{\text{TSS}})^{2.94}
\]

One of the striking things about this relationship is that, for the Freundlich isotherm, the DIP concentration behaves approximately like the cube of the mass concentration of PIP, and is very sensitive to small changes in this concentration. We return to this point later.

The DIP concentration in the sediments is usually much higher than that in the overlying water column, and so the flux of DIP from the sediments to the water column is given approximately by \(V_N \cdot \text{DIP(sed)}\), where \(V_N\) is the sediment-water column exchange velocity for nutrients. During times of low catchment loads, we can expect this flux to be the dominant source of P into the overlying water column.

The mass ratio of PIP/TSS in the sediments therefore plays a critical role in determining DIP concentrations in both the sediment and the overlying water column. This ratio is determined by mass balances among the sources and sinks for TP and TSS in the sediments. The mass balance for TSS was discussed in the previous section. The mass balance for TP depends on a balance between supply (through settling of particulate organic P and PIP from the water column), diffusion of DIP back to the water column, and a model term for immobilization of P in sediments.

The P immobilization term was included based on an analysis of P budgets for L. Wellington. As noted earlier, the average concentration of TP in input loads is about 130 mg m\(^{-3}\), and the typical observed values in the water column are around 70 mg m\(^{-3}\), suggesting L. Wellington sequesters just under half of the incoming P load in sediments. The TP:TSS load in loads is about 1400 mg kg\(^{-1}\), but the observed TP:TSS ratio in surface sediments is only about 300 to 600 mg kg\(^{-1}\). Observed DIP concentrations in sediments are high, around 1000 mg m\(^{-3}\). These sediment concentrations are consistent with the recommended Webster and Grace (2001) adsorption isotherm for bed sediments. If the model is run with this adsorption isotherm, it tends to predict TP:TSS ratios around 600 mg kg\(^{-1}\), and DIP concentrations around 500 to 1000 mg m\(^{-3}\).

A similar analysis of TSS suggests that Lake Wellington sequesters about 40% of the incoming TSS load. The problem is that, if L. Wellington sequesters 40% of the incoming TSS load, and the TP:TSS ratio in surface sediments is only about 40% of the TP:TSS ratio in catchment loads, then the Lake can sequester less than 20% of the input TP load. It must then export the rest, and to do this, the TP concentration in the water column must be more than 80% of the TP concentration in the loads, higher than observed. This is precisely what happens if the model is run without either TSS or P burial / immobilization. Under these conditions, it predicts high values of TP and DIP (DIP up to 95 mg m\(^{-3}\)) in surface waters.

One can reduce DIP to the observed range by increasing the P\_ads\_coeff for the water column from around 107 to 600. This converts most of the DIP in the water column to PIP.
However, this leaves a TP concentration that is too high, and results in too much export of P from L. Wellington to L. Victoria.

Another option is to assume that the observed sequestration is due to a process that immobilizes or buries P in sediments. In the preceding section, we introduced a burial rate for TSS in sediments, which maintains the TSS pool in the oxic layer, available for P adsorption, in long-term balance with loads. One could argue that we should bury P at the same rate as TSS, although this is questionable. If TSS burial is acting to maintain a fixed oxic layer, then TSS “burial” effectively represents movement of TSS to below the oxic layer, which should result in the release of adsorbed PIP, and its retention in the oxic layer. In any case, burying P at the same rate as TSS is not sufficient, as it means removing TP at a TP:TSS ratio which is much less than loads. In order to sequester TP at the observed rates, we must immobilize P in the sediment at rates that are slightly greater than burial rates of TSS.

The processes underlying P immobilization or burial in L. Wellington (or other) sediments are not well understood. Of course, a 1-layer sediment model is not in a position to represent these realistically in any case, and the introduced immobilization rate is justified solely on empirical grounds. However, there are large pools of particulate P at depths of 20 cm and greater, under anoxic conditions, in all Lakes. It is not known whether this P is associated with refractory organic matter, or is inorganic P bound to sediments in a way that is not reversible under anoxic conditions.

A significant feature of the phosphate fluxes from L. Wellington noted by Webster and Wallace (2001) is that they become negative at all five sites below about 10cm depth. A possible explanation for this behaviour is that conditions deeper in the cores are conducive to precipitation of phosphate as calcite or some other mineral. In effect, this precipitation would act as a deep sink for phosphate within the sediment column. Further field and laboratory studies are necessary if we are to understand the processes controlling long-term burial of P in sediments.

With an adjustable empirical parameter r_immob_P, we can in principle control the efficiency of P burial, and the predicted TP and DIP concentrations in the overlying water column, at least in a long-term average sense. However, in practice, there are quite strong constraints on the degree to which we can increase P sequestration and still match observations. Interestingly, these have to do with the balance between N and P limitation in the lakes.

As noted above, the N:P ratios in input loads are slightly above Redfield in L. Wellington, and substantially above Redfield in L. King. In the absence of differential sinks for N and P within the Lakes, one would expect the Lakes to operate as a P-limited rather than N-limited system. The observations show that the opposite is the case: DIN levels are drawn to very low levels in surface waters almost all of the time, while DIP levels appear to be mostly buffered at moderate, non-limiting levels. The model reproduces these observations, primarily because N is lost to denitrification. However, if P burial becomes too large, and outweighs N losses to denitrification (in a relative sense), then the system switches over into P limitation.

In model calibration, it was found that increasing r_immob_P more than slightly above the TSS burial rate (0.001 d⁻¹) led to the prediction of extended periods of P depletion, and associated elevated surface ammonia, in L. Victoria and L. King. A compromise value for r_immob_P of 0.0012 d⁻¹ was selected which avoided P depletion, at least in the calibration period. This value results in predicted water column TP and DIP concentrations in L. Wellington that are larger than those observed, and somewhat less TP sequestration than is observed (Fig. 10, 11).

The P_ads_coeff was increased from 107 to 300, to reduce predicted DIP concentrations, but these are still about twice those observed (Fig. 10). As discussed earlier under model
formulation, the analysis conducted by Webster and Grace leaves considerable uncertainty about the correct value for $P_{ads\_coeff}$ for suspended sediments: their estimates ranged from 58 to 5100, although they expressed major reservations about the higher values. Again, further studies are needed to better understand P adsorption-desorption in the water column at low concentrations of suspended sediment.

With these parameter choices, the predicted values for DIP in both surface and bottom waters in L. Victoria and L. King match observations surprisingly well. P dynamics in these lakes are modified substantially by the occurrence of periods of oxygen depletion in bottom water, which lead to anoxia and P release from sediments. In the model, this is represented by making $P_{ads\_coeff}$ in sediments depend on the oxygen concentration, with a half-saturation parameter $K_{O\_Pads}$. This parameter was set to 2000 mg O m$^{-3}$, so that desorption and DIP release would occur under conditions of bottom water hypoxia. As noted in the earlier discussion of the formulation of sediment biogeochemistry, $K_{O\_Pads}$ is essentially an empirical parameter, whose value depends on the 1-layer sediment representation, and the choice of $K_{O\_aer}$.

The model reproduces the magnitude of elevated bottom water DIP concentrations following the June 1998 runoff events fairly well, though it does not capture the timing of these episodes. This is discussed further under nitrogen and phytoplankton dynamics.

Episodes of bottom water hypoxia and sediment anoxia lead to release of virtually all the accumulated adsorbed PIP in surface sediments in L. Victoria and L. King. Thus the long-term balance for PIP accumulation there differs from that in L. Wellington. Even in the absence of any long-term P burial, the PIP pool in sediments approaches a long-term balance in which export of TP matches loads. However, this balance involves higher values of TP and DIP in the water column than are observed, so that P burial or immobilization is necessary in these Lakes as well for the model predictions to match observations.

The uncertainties about processes controlling P immobilization or burial in sediments do strongly affect the ability of the model to address transients on time scales of several years to decades. Recall that, for calibration purposes, the model was run for 12 years, with the 1995-99 forcing repeated three times. The longest time scales in the model are the sediment burial and P-immobilization time scales, of about 3 years. The model is therefore effectively calibrated to produce a “steady-state” response to forcing over the 1995-99 period. It cannot be used to address questions as to whether long-term changes in P pools and sediments would occur if current loads were maintained.

### 5.2.4 Denitrification

Given the very low flushing rates in the Gippsland Lakes, especially during periods of low flow, one would expect losses to denitrification to play a major role in the overall nitrogen budget, controlling the amount of nitrogen cycling through the water column and sediment, and consequently the amount of algal biomass and production; i.e. the degree of eutrophication. In practice, the nitrogen load is generally concentrated in winter-spring wet seasons, and much of it occurs in a small number of runoff events, especially in L. King. The DIN concentrations and algal blooms during and immediately following these runoff events are primarily controlled by physical transport and phytoplankton growth dynamics, and are discussed further in the following section. Denitrification controls the efficiency with which organic nitrogen settling out during and after these events is recycled back into the water column as DIN. Given that flushing time scales are quite long when catchment flows are low, one can think of denitrification as determining a decay time scale over which nitrogen levels and phytoplankton biomass and productivity decay during periods of low loads.
The calibration period provides a good basis for calibration of denitrification efficiencies, as it includes a prolonged period of very low loads, which follows two above average wet seasons, and is in turn followed by large isolated runoff events and a typical dry summer and autumn.

L. Wellington and Lakes Victoria and King again provide contrasting systems for calibration. In L. Wellington, water column oxygen remains high, and so we expect denitrification efficiencies to remain high, except under conditions of very high sediment organic load and respiration demand. In Lakes Victoria and King, bottom water oxygen is depleted, leading in the model to decreases in, and ultimately shutdown of, denitrification even at modest sediment respiration rates.

Because nitrification is generally thought to be the rate-limiting step, the overall denitrification efficiency was adjusted by changing the maximum oxygen-saturated nitrification rate $r_{\text{nit}}$. It was found that the predictions best matched observations with $r_{\text{nit}}$ equal to 0.4 d$^{-1}$, which is close to the value chosen initially (0.5 d$^{-1}$) based on the analysis of simple analytic models used in formulating the sediment biogeochemistry. For $r_{\text{nit}} = 0.4$ d$^{-1}$, the theoretical maximum denitrification efficiency is about 75%, close to the value adopted in Port Phillip Bay.

If $r_{\text{nit}}$ is set to substantially lower values, e.g. $r_{\text{nit}} = 0.2$ d$^{-1}$, then the Lakes switch over to P limitation, with very high modelled surface ammonia values. If $r_{\text{nit}}$ is set to substantially higher values, e.g. $r_{\text{nit}} = 0.8$ d$^{-1}$, then the Lakes become very strongly N-limited, and surface DIP builds up to higher levels than observed in all basins. Furthermore, the decline in phytoplankton biomass during the extended dry period preceding June 1998 is exaggerated in all basins.

Because surface ammonia and nitrate are depleted to very low values in surface waters in all basins except during runoff events, the key observations for calibration of denitrification are the time series of chlorophyll in all boxes, and of bottom ammonia in L. Victoria and L. King. In L. Wellington, the model predicts chlorophyll concentrations that are of the right magnitude, though slightly low, during the dry year prior to the June 1998 run-off event, and in the late summer and autumn of 1999 (Fig. 11). While peak chlorophyll concentrations associated with the June 1998 runoff event reach roughly the right magnitude, around 50 mg Chl m$^{-3}$, the model does not maintain these high concentrations over the extended period from July to December 1998, as observed. It is possible that this reflects problems in phytoplankton dynamics (discussed below), or that the model is overestimating denitrification efficiencies in L. Wellington in this period.

The model predicts denitrification efficiencies in L. Wellington ranging from 40 to 60%, with a minimum in winter due to reductions in nitrification rates at low temperatures. Predicted denitrification efficiencies in L. Victoria and L. King range from 0 to 35%. There are extended periods with denitrification efficiency close to zero, which correspond to periods of drawdown in bottom water oxygen, and elevated sediment respiration rates. Model experiments using different values of $r_{\text{nit}}$ demonstrate a positive feedback loop involving bottom water oxygen. When $r_{\text{nit}}$ is increased, denitrification efficiency increases, ammonia release from sediments is lower, organic matter production is reduced, sediment respiration rates decline, and the duration and intensity of episodes of bottom water hypoxia are also reduced, leading to still further increases in overall denitrification efficiency.

The model predicts surface chlorophyll values of approximately the right magnitude both before and after the runoff event in L. Victoria and L. King (Fig 15, 33), although the timing of blooms is not always correct. The predicted high chlorophyll in the autumn of 1999 is associated with a *Nodularia* bloom, and discussed further below.
The model achieves mixed results in its attempts to reproduce the observed elevated ammonia concentration in bottom waters in 1998/99. In northern and southern L. King, and eastern L. Victoria, the model reproduces the magnitude of bottom water ammonia concentrations quite well, but gets the timing wrong, predicting high concentrations in late 1998 and low concentrations in early 1999, whereas the observations suggest the reverse. This doesn’t appear to be due to problems in predicting bottom water hypoxia: the model predicts bottom water oxygen values reasonably well through the period from October 1998 to May 1999. It may well be due to weaknesses in the simulation of dinoflagellate dynamics, discussed below.

The model badly overestimates bottom-water ammonia values in western and central L. Victoria (Fig. 14, 20). The reasons for this aren’t clear. In western L. Victoria especially, the lake is relatively shallow, the bottom layer is quite thin, and it is possible that this reflects an inherent problem in the 2-layer structure. It does not appear that the problem is due to over-prediction of oxygen drawdown – the predicted bottom water oxygen values match observed values quite well. The situation is different in central L. Victoria, where the model badly over-predicts bottom water oxygen depletion and ammonia build-up in bottom water at EPA site 2311. (The dissolved oxygen data at this site are a little surprising, being generally much higher than the MAFRI data from site 8 to the west and site 4 to the east.)

The estimated coastal catchment loads into L. Victoria are based solely on land use classification, and not supported by in-stream monitoring. These loads could be over-estimated. Removing all coastal loads into L. Victoria reduces the predicted bottom ammonia values in July-December 1998 substantially, from over 400 to about 160 mg N m$^{-3}$ in western L. Victoria, and 260 mg N m$^{-3}$ in central L. Victoria. However, the observed bottom water ammonia values are still much lower, close to detection limits, in this period.

An alternative explanation, as for L. King, is that the model is underestimating the impact of dinoflagellate blooms in this period.

The new formulation of sediment biogeochemistry has been designed to produce plausible denitrification efficiencies and sediment fluxes of oxygen, ammonia and DIP under different conditions of bottom water oxygen, organic matter load and sediment respiration rates. While the data available for the Gippsland Lakes are not sufficient to fully test and calibrate this new formulation, they appear to be generally compatible with it. As was the case for P immobilisation, the overall model is most sensitive to decreases in denitrification efficiency, which shift the system from N to P limitation.

5.2.5 Phytoplankton dynamics

The model now contains four phytoplankton components which all interact through competition for light and nutrients, and through interactions with grazers. In some cases, these interactions are strong and involve poorly understood processes, creating particular difficulties for model formulation and calibration.

Nutrient uptake and light absorption by phytoplankton are all modelled using the CR formulation described above, with standard parameter values based on phytoplankton cell radius. These parameter values have not been modified in calibrating the model. The maximum growth rates for the small flagellate and diatom functional groups have also been left unchanged from the “standard” values developed through Port Phillip Bay and the subsequent National Land and Water Resources Audit (NLWRA) study. The key parameters that have been modified are the quadratic mortality parameter for large zooplankton, diatom sinking rates, the mortality and recruitment rates for Nodularia, and the maximum growth rates and some physiological constants for dinoflagellates and Nodularia.
The most straightforward aspect concerns small flagellates and their associated microzooplankton grazers. The parameters for these groups have not been changed. Small flagellates have the smallest cell radius (2.5 µm), and under the CR model can accordingly outcompete all larger cells when N or P is limiting. However, the microzooplankton grazing and mortality parameters are fixed so that small flagellates are subject to grazing control, even when nutrients are saturating. The result is that small flagellates tend to occur at fairly low background levels, and respond only weakly to runoff and load events. In a eutrophic system such as the Gippsland Lakes, they have relatively little impact.

The diatoms are assigned a high maximum growth rate (2 d⁻¹), and an intermediate cell radius (10 µm). Large zooplankton are assumed to have a low maximum growth rate (0.1 d⁻¹), and are not able to respond numerically to rapid increases in diatom biomass. In the absence of dinoflagellates and *Nodularia*, diatoms make the dominant contribution to phytoplankton biomass. One can distinguish two aspects to model diatom dynamics in the calibration period. The major flood event in June 1998 floods the Lakes with high nitrate surface water. Diatoms respond very rapidly to this, and a major bloom results, stripping nitrate out of surface waters. The peak bloom density is quite insensitive to parameters controlling grazing or sinking losses, and depends almost entirely on the amount of nitrate supplied. Grazing has little effect on peak bloom density because grazers are at low densities prior to the runoff event (following a year of very low loads), and have slow growth rates.

The diatom bloom gradually decays, due to a combination of grazing and sinking losses. While some organic nitrogen is recycled in surface waters, much settles into bottom waters or to the sediment. Inorganic nitrogen concentrations generally remain very low in surface waters, but continued phytoplankton growth in L. Wellington is maintained by release of ammonia from sediments. Light attenuation is high, and diatom and flagellate growth and uptake of ammonia are strongly light-limited in bottom waters in L. Victoria and L. King. There, ammonia accumulates in bottom waters, and continued diatom and flagellate growth is sustained by mixing of ammonia into surface waters. In L. Wellington, a second runoff event in November 1998 triggers a second though smaller diatom bloom.

In 1997/98, and in autumn 1999, after extended dry periods, simulated diatom biomass in surface layers approaches a steady-state, with growth (limited by the supply of ammonia via mixing and/or release from sediments) balanced by losses due to sinking and grazing. Zooplankton parameters and diatom sinking rates do affect diatom biomass at these times. High sinking rates, or high grazing pressure (low zooplankton mortality) result in lower diatom biomass. The chosen values (diatom sinking rate equal 0.2 m d⁻¹) and zooplankton mortality rate equal 0.0004 d⁻¹ mg⁻¹ m³) result in values for chlorophyll under low flow conditions which match observations reasonably well.

As discussed under model formulation, dinoflagellates were introduced as a functional group both because dinoflagellate blooms occur and are thought to be important, and because dinoflagellates, by vertically migrating, should be able to exploit situations in L. Victoria and L. King where there is elevated ammonia in bottom waters. Dinoflagellates were assigned low maximum growth rates (1 d⁻¹), and a large cell radius (20 µm). It is worth noting that if dinoflagellates are assigned a smaller cell radius than diatoms in the model, they outcompete diatoms for nutrients at low concentrations, and can displace them almost entirely from the system.

In the model, dinoflagellates were designed to exploit situations in L. Victoria and L. King where bottom nutrients are elevated by vertically migrating. However, this leads to somewhat complicated interactions with both diatoms and grazers on the one hand, and bottom nutrients on the other. Dinoflagellates are assumed to be grazed by the same large zooplankton as diatoms. Because the situations where bottom nutrients are elevated generally follow runoff events and diatom blooms, large zooplankton are most abundant precisely when nutrient and
light conditions favoured dinoflagellate growth. This leads to delay or even suppression of
dinoflagellate blooms in the model. On the other hand, if zooplankton impacts on
dinoflagellates are reduced (eg by increasing zooplankton mortalities), then large, prolonged
dinoflagellate blooms resulted, which reduce ammonia in bottom waters to very low levels for
extended periods. This is not consistent with the observations that show extended periods of
high bottom water ammonia and DIP.

The calibrated model run represents a compromise in which grazing delays dinoflagellate
blooms until early or late summer. As noted earlier, this results in patterns of elevated ammonia in bottom waters which are inconsistent with observations. The latter show low bottom water ammonia in October to December, and high bottom water ammonia in January to April, suggesting that dinoflagellate blooms occur earlier and not later.

The model representation of dinoflagellates is not adequate to simulate the timing of
dinoflagellate blooms. This could be because aspects of the autecology of the particular
species present, such as temperature or salinity preferences, conditions for cyst germination,
or micronutrient requirements, are missing. Alternatively, it could be that grazing is
misrepresented, and that dinoflagellates are not grazed by the same zooplankton assemblages
as diatoms. Detailed process studies would be required to resolve these questions.

The cyanobacterium *Nodularia* was also introduced into the model as a new functional group,
because of its significance as a source of nuisance blooms, and because of its potential role in
N fixation. *Nodularia* was also assigned a large cell radius (20 µm), to represent the effect of
heterocysts. As discussed earlier, it was assigned a low maximum growth rate of 0.4 d⁻¹, and
given a facultative capability to fix nitrogen in situations where ammonia is low. *Nodularia*
blooms are observed to occur in late summer, when surface ammonia is strongly depleted, and
tend to be concentrated in L. King.

Surface DIP levels are not especially elevated in late summer, and are sufficient to potentially
support *Nodularia* growth throughout much of the year. At least in the model, the timing of
*Nodularia* blooms is controlled primarily by the temperature requirement for growth. It was
necessary to increase the temperature threshold for growth to 19 °C in order to delay
*Nodularia* blooms until late summer.

Without intervention, the model tends to produce large *Nodularia* blooms in L. Wellington.
This is not too surprising: N:P ratios in loads into L. Wellington are low, and predicted water
column DIP concentrations are substantial. Salinities are low in L. Wellington, but the model
(following the literature) imposes an upper salinity limit to *Nodularia* growth, but not a lower
limit.

In keeping with observations, the model assumes that akinete germination occurs in L. King
but not in L. Wellington. Akinete germination is represented not as a dynamic process, but as
a fixed (small) input of *Nodularia* into L. King in January and February each year. This does
result in *Nodularia* blooms that are concentrated in L. King, but transport of seed populations
into L. Victoria and even L. Wellington is sufficient to produce small blooms there. If
germination is assumed to occur earlier, in November or December, then maximum bloom
densities again occur in L. Wellington.

The model does produce *Nodularia* blooms with approximately the right timing and location.
The bloom magnitude can respond to changes in DIP supply or in temperature or salinity, in
summer months. The model predicts an unusually small bloom in the summer of 1998. This is
due to the unusually high salinities, around 31 PSU, in L. King, following a year of low
runoff. Predicted DIP concentrations are not unusually low at the time.
In order to achieve a bloom of short duration, it was necessary to use a linear mortality rate of 0.1 d\(^{-1}\), as well as a quadratic mortality rate of 0.0002 d\(^{-1}\). The bloom duration is arguably still longer than indicated by the observed chlorophyll peak.

There are a number of aspects of *Nodularia* physiology and autecology that are not well understood, and the model represents only a first attempt to capture *Nodularia* dynamics. For many bloom species, the annual occurrence of blooms appears to be controlled by cyst or akinete germination, and it has been notoriously difficult to identify robust predictive explanations in these cases. Where germination occurs consistently, and bloom magnitude is controlled by nutrient supply, as appeared to be the case for *Nodularia* and phosphorus in Peel-Harvey Inlet (Lukatelich and McComb, 1986), robust prediction is much more feasible.

The long-term EPA time series for Gippsland Lakes appears to include some very large cyanobacterial blooms, which have led to massive increases in organic N, far outweighing catchment loads. It is not clear what circumstances have led to those. A key issue which will affect the magnitude of blooms by *Nodularia* or other N-fixing cyanobacteria in L. King and L. Victoria is their ability to access high DIP concentrations in bottom waters.

### 5.2.6 Phytobenthos

The model predicts loss of seagrass and macroalgae within the Lakes due to the high light attenuation and very low bottom light intensities. This of course partly reflects the coarse horizontal and vertical spatial resolution in the model. Each box is represented as being of uniform depth, corresponding to the average depth of the basin or sub-basin. One would expect growth of macroalgae and possibly seagrass in shallow areas along the basin margins. One would further expect the high turbidity and light attenuation, and elevated nutrient levels, to favour macroalgae over seagrass.

In the NLWRA Estuaries study, it was suggested that the light attenuation depth, or approximately the Secchi depth / 2, be taken as the lower limit for seagrass. Throughout the Lakes, this would give a depth limit for seagrass of about 0.5 m. Although the model cannot be used to address seagrass dynamics on basin margins directly, this formula can be used in model scenarios to assess the effects of turbidity and light attenuation on seagrass depth range.

In Jones Bay and Western L. Victoria, which are the shallowest boxes, the model predicts small blooms of microphytobenthos in most years, with maximum concentrations, around 5 mg Chl m\(^{-2}\), in 1997/98, when light attenuation is lowest. In the other deeper basins, the model predicts microphytobenthos blooms, around 3 mg Chl m\(^{-2}\), only in 1997-98. The model predicts much higher concentrations of chlorophyll in sediments due to settling out of planktonic diatoms, with maximum predicted concentrations in L. Wellington over 100 mg Chl m\(^{-2}\). Measurements of chlorophyll in surface sediments may not distinguish between benthic diatoms and settled planktonic diatoms.
6 Model Baseline Runs July 1995 to June 1999

While the model has been calibrated for the period July 1997 to June 1999, loads and exchanges have been assembled, as described above, to force the model for the period July 1995 to June 1999. This period includes two above average wet seasons in 1995 and 1996, and provides a more representative range of loads and flows against which to assess various management scenarios. Here, we briefly present and discuss model predictions for the extended period, both as background to the Scenarios Report, and for the additional insights they offer into the response of the Lakes to forcing.

Model predictions for the 4-year period July 1995 to June 1999 are shown in Fig. 49-54. We present time series of Chlorophyll a in surface waters, ammonia and phosphate in surface and bottom waters, and dissolved oxygen in bottom waters, for L. Wellington, central L. Victoria, and southern L. King.

Recall that loads into L. Wellington are much larger in the winter-spring wet seasons of 1995 and 1996. These loads result in high chlorophyll levels, around 30 mg Chl a m\(^{-3}\), for extended periods in both years. The intense runoff event into L. Wellington in November 1995 produces a particularly intense bloom, exceeding 100 mg Chl a m\(^{-3}\). Predicted DIP levels are generally lower in the 1995-97 period, and the model predicts a period of DIP depletion and increased surface ammonia following the November runoff event. This occurs partly because the N:P ratio in loads exceeds Redfield, and partly because the very high organic flux and sediment respiration rates during the bloom temporarily depress denitrification efficiency in L. Wellington sediments.

Loads into L. King are fairly low in 1995, and moderate in the 1996 wet season, although still considerably less than loads in the major flood event of June 1998. Predicted chlorophyll levels in L. King range from about 6 to 16 mg Chl a m\(^{-3}\) in the 1995-97 period, and do not reach the peak values predicted to follow the June 1998 flood event, or the low values predicted at the end of the long dry period preceding this flood. Periods of DIP depletion in surface waters are predicted in the late winter – spring of 1995 and 1996, and surface ammonia is slightly elevated in both periods. The model predicts high ammonia concentrations in bottom waters throughout most of the 1995-97 period, with peaks in summer, and high concentrations of DIP in bottom waters in both summers. However, bottom oxygen concentrations are higher in the summers of 1996 and 1997 than 1999.

Central L. Victoria appears to be dominated by inputs from L. Wellington rather than L. King in the 1995-97 period. Chlorophyll levels are high, around 40 mg Chl a m\(^{-3}\), throughout the spring and summer of 1995/96, with peak concentrations of 80 mg Chl a m\(^{-3}\) occurring briefly following the November 1995 event. Surface DIP is depleted and surface NH\(_4\) elevated briefly in the winter-spring of both 1995 and 1996. Predicted bottom ammonia levels are high throughout the 1995-97 period, and bottom DIP levels are high in both spring-summer periods. Again, extended periods of bottom water hypoxia are predicted in both years.

The model predicts low levels of dinoflagellates in the 1995-97 period, although conditions appear to favour dinoflagellate growth. This seems to be because the persistent diatom blooms result in high zooplankton concentrations, which suppress dinoflagellates. As noted earlier, these interactions in the model are questionable. The model also predicts smaller *Nodularia* blooms in the 1996 and 1997 summers. This may be because surface DIP is depleted in the periods leading up to blooms.
The key lessons from these extended simulations are:

- The model Lakes respond to additional loads in 1995-97 with increases in chlorophyll, increases in concentration of bottom water ammonia and phosphate, and extended periods of bottom water hypoxia;

- During periods of high loads in the western catchments, and low loads in the eastern catchments, L. Victoria is dominated by inputs from L. Wellington;

- The model as calibrated can be tipped over temporarily into P-limitation by large runoff and load events, due to the N:P ratios in inflows, and the suppression of denitrification in L. Wellington. In the long-term EPA data set, there are observations of high surface ammonia concentrations, which may correspond to episodes of surface P limitation.

- The model predicts considerable interannual variation in dinoflagellate and *Nodularia* blooms, although not necessarily for the right reasons.
7 Conclusions

While there is certainly room for further development and improvement, the data and model analysis presented here allow a number of reasonably robust conclusions about the Gippsland Lakes and their response to catchment loads of nutrients and sediment. This section summarises the key conclusions, bearing in mind their significance for management of the Lakes. We also spell out some of the key remaining uncertainties and limitations in our understanding, as captured in the model. Both aspects are discussed further in the Scenarios Report.

The Lakes are subject to very substantial nutrient loads. Over the two-year 1997-99 calibration period, the mean annual loads were about 1400 tonnes TN, and 160 tonnes TP, about equally split between L. Wellington and L. King. Interestingly, although attention has focused on catchment loads into L. Wellington, especially from the Latrobe R., loads into L. King were comparable in magnitude, and, according to the model, equally important for water quality. The calibration period 1997-99 is somewhat unusual in terms of loads as it includes both an extended dry year and an unusually large flood event in the eastern catchments. Over the 1995-99 period, which included two “normal” wet seasons, loads into L. Wellington were about 2 to 3 times loads into L. King. However, it remains true that loads into L. King have a significant impact on water quality in eastern L. Victoria and L. King.

The temporal pattern of loads differs between basins: loads into L. Wellington, dominated by the regulated Latrobe R. were more constant over time, whereas over 50% of loads into L. King in the 1997-99 period occurred in a single flood event in June 1998.

The Lakes have low flushing rates. During low flow periods, exchanges between L. Wellington and the other Lakes, and between the other Lakes and Bass Strait, are very small. The hydrodynamic modelling estimates flushing times under dry conditions of 5 to 9 months. These are long, and certainly contribute to the eutrophic state of the Lakes (but Port Phillip Bay has a longer flushing time of around one year).

Light attenuation in the Lakes is generally high, due primarily to high turbidity and CDOM, with some contribution from dense algal blooms. In L. Wellington, light attenuation is generally high enough to prevent benthic primary production, but not high enough to prevent the accumulation of high phytoplankton biomass in the water column. In L. Victoria and L. King, light attenuation is lower than in L. Wellington, and is reduced substantially during extended dry periods. However, it is still sufficient in the model to prevent growth of macrophytes in the deeper basins that constitute most of the area in these lakes, although macrophyte growth will of course occur in the shallow margins. The model predicts low biomass of microphytobenthos, at least in the deep basins, and predicts that planktonic rather than benthic diatoms may be the dominant source of chlorophyll in surface sediments.

L. Wellington is shallow and well-mixed. It behaves like a giant phytoplankton chemostat, with a persistent high standing stock of phytoplankton, and ammonia and nitrate reduced to very low levels despite high concentrations in inflows. The phytoplankton biomass in L. Wellington appears to respond more slowly to runoff events in reality than in the model: chlorophyll levels both increase and decay too rapidly in the model.

The behaviour of L. Victoria and L. King is dominated by the episodic nature of inputs, and by the strong and persistent stratification. Large runoff events flood surface waters with very high nitrate concentrations, which result in intense surface blooms. These blooms rapidly deplete nitrate and ammonia in surface waters, and DIN is low and limiting in surface waters for most of the time. Remineralization of organic matter in bottom waters and sediments leads to high rates of oxygen consumption, which in the presence of strong stratification, deplete
bottom water oxygen. Nutrient release rates from sediments are enhanced by oxygen depletion, which tends to shut down nitrification, and drive release of adsorbed P through Fe reduction. The result is the accumulation of high concentrations of ammonia and phosphate in bottom waters. High light attenuation limits utilisation of these nutrients by phytoplankton and macroalgae.

During periods of low runoff and loads, growth of phytoplankton in surface waters is sustained by mixing of nutrients across the pycnocline. Dinoflagellates that can undertake diel vertical migrations are able to take up nutrients in bottom waters at night, and fix carbon in surface waters in the day. Observations suggest dinoflagellate blooms may deplete bottom nutrients in spring and early summer, but the model does not predict the onset and duration of blooms correctly. More knowledge of the autecology of key dinoflagellate species is needed.

Observations show surface waters in all basins are N-limited rather than P-limited, although TN:TP ratios in loads are slightly above Redfield in L. Wellington, and substantially above Redfield in L. King. In the long run, N:P ratios in the water column are controlled by sediment processes, especially the burial or immobilization of P in sediments, and the loss of N to denitrification. The model indicates a relatively delicate balance between these processes: it is easy to push the model into P-limitation by increasing P burial rates or reducing denitrification efficiencies.

There are still substantial gaps in our knowledge of sediment biogeochemistry as it affects these long-term processes. We do not even know whether the large concentrations of particulate P in sediment profiles are predominantly inorganic P or refractory organic P. The observations in the calibration period suggest that both the maintenance of excess P in the long-term, and the buffering of water column DIP in the short-term, may be more robust in reality than in the model. However, the long-term EPA monitoring record includes a number of observations of high surface ammonia in L. Wellington, suggesting periods of P-limitation may have occurred in the past.

In the model, denitrification efficiencies are reduced by bottom water oxygen depletion in L. Victoria and L. King. The model displays some positive feedback in that decreases in denitrification efficiency lead to increased nutrient release, more organic matter production, and longer periods of bottom water anoxia or hypoxia. However, this feedback is constrained by both the light limitation in bottom waters, and the switch to P-limitation once denitrification efficiencies drop too far.

Nitrogen limitation in surface waters would be expected to favour the development of N-fixing cyanobacterial blooms, and substantial *Nodularia* blooms are observed. Temperatures in the Lakes may be marginal for *Nodularia*; model calibration suggests growth may be restricted to summer months of peak temperatures. Salinities are generally favourable for *Nodularia*, but may become too high in L. King in years of especially low runoff. It is not clear why *Nodularia* blooms are confined to the eastern lakes: the model tends to predict largest blooms in L. Wellington. Again, more knowledge of the autecology of *Nodularia* is needed for the model to predict bloom occurrence and magnitude with confidence.

### 7.1 Comparing Gippsland Lakes and Port Phillip Bay

It is interesting to compare Gippsland Lakes with Port Phillip Bay. The table below shows loads of N and P per unit area and per unit volume, for the 1995-99 period, for L. Wellington (western catchments only), L. Victoria + L. King (eastern catchments only), Gippsland Lakes (all loads), and Port Phillip Bay (from the PPBES period).
Normalised per unit area of the receiving water body, loads of TN into L. King and L. Victoria from the eastern catchments are comparable to those into Port Phillip Bay, and loads of TP into L. King and L. Victoria are less than half those into Port Phillip Bay. Loads of TN into L. Wellington per unit area are almost 3 times those into Port Phillip Bay, and loads of TP only slightly greater. For the Gippsland Lakes as a whole, loads of TN per unit area are about 60% higher than those into Port Phillip Bay, and loads of TP per unit area are 75% of those into Port Phillip Bay. Loads per unit area into the Gippsland Lakes are certainly not dramatically higher than those into Port Phillip Bay, with the exception of N loads into L. Wellington.

On the other hand, when normalised per unit volume of the receiving water body, N loads into Gippsland Lakes are substantially higher than those into Port Phillip Bay. This is particularly true for L. Wellington, which is much shallower than Port Phillip Bay, but also true for the Lakes as a whole. Even for L. Victoria and L. King, the TN load per unit volume is more than twice that into Port Phillip Bay. A similar conclusion holds for P loads per unit volume, except that the load into L. King and L. Victoria is comparable to that into Port Phillip Bay.

It is questionable whether it is appropriate to compute loads into L. King and L. Victoria based only on the eastern catchments. Given that about 60% of the N and P load into L. Wellington is exported to L. Victoria, one should arguably add this to the eastern catchment loads. This serves to approximately double the figures given for L. King and L. Victoria in the Table.

There is some inconsistency in the literature as to whether loads per unit area or loads per unit volume provide the best measure of eutrophication stress. In well-mixed systems such as L. Wellington and Port Phillip Bay, loads per unit volume determine the immediate impact in terms of nutrient levels and associated chlorophyll. However, where sediment-related processes such as denitrification control the system response, one might expect loads per unit area to be more important. L. Wellington is subjected to N loads per unit volume which are more than an order of magnitude greater than those into Port Phillip Bay. These result in chlorophyll levels that are also roughly an order of magnitude higher than those observed in Port Phillip Bay. The model does not predict substantial or prolonged shut-down of denitrification in L. Wellington.

The persistent stratification in L. Victoria and L. King makes these basins much more vulnerable to eutrophication. Stratification combined with high light attenuation allows oxygen depletion and nutrient build-up in bottom waters. This means that denitrification is shut down at much lower levels of primary production than would be required in a well-mixed system. Oxygen depletion also leads to P release from sediments, N loads per unit volume into L. King and L. Victoria are 2 to 4 times those into Port Phillip Bay, depending on whether we include export from L. Wellington, but chlorophyll levels are again close to an order of magnitude higher.

<table>
<thead>
<tr>
<th></th>
<th>mg N m⁻² d⁻¹</th>
<th>Mg N m⁻³ d⁻¹</th>
<th>mg P m⁻² d⁻¹</th>
<th>mg P m⁻³ d⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>L. King + L. Victoria</td>
<td>9.77</td>
<td>2.00</td>
<td>0.86</td>
<td>0.18</td>
</tr>
<tr>
<td>L. Wellington</td>
<td>30.67</td>
<td>11.66</td>
<td>3.53</td>
<td>1.34</td>
</tr>
<tr>
<td>Gippsland Lakes</td>
<td>18.18</td>
<td>4.58</td>
<td>1.93</td>
<td>0.49</td>
</tr>
<tr>
<td>Port Phillip Bay</td>
<td>11.36</td>
<td>0.84</td>
<td>2.56</td>
<td>0.19</td>
</tr>
</tbody>
</table>
7.2 Key Limitations and Uncertainties

Perhaps the most important uncertainties for management are those affecting the long-term storage and cycling of phosphorus and nitrogen. As indicated above, and by Webster and Wallace (2001), we do not understand well those processes affecting the long-term accumulation of P or N in sediments, the composition of the large pools of particulate N and P present in sediments, and the conditions, if any, under which these pools could be remobilized. The model has been calibrated to match the observed cycling of N and P in the water column, and between water column and sediments, as closely as possible. However, the representation of sediment biogeochemistry must still be described as semi-empirical. This means that there is particular uncertainty in using the model to address questions about long-term transients such as:

1. Will water and sediment quality in the Lakes deteriorate if loads are maintained at current levels?

2. How long would it take for water and sediment quality to improve following a large reduction in loads?

The model is currently calibrated so as to maintain sediment pools more or less in balance with loads on time scales of a few years, so that it is bound to answer the first question in the negative. The model answer to the second question is quite short, of order a year or less. This is primarily because denitrification and flushing combine to deplete available nitrogen pools on that time scale. There is some support for that conclusion in the observations: water quality improved markedly in the Lakes during the 1997/98 dry year, and the model is calibrated to reproduce that transient.

Comparison with observations suggests that the representation of phosphorus dynamics in the model does not fully capture the processes that buffer P concentrations in the water column. In the model, the Lakes are predominantly N-limited, but can be easily tipped over into P-limitation. There appears to be little evidence for P-limitation in the observations.

Although the model includes explicit dinoflagellate and Nodularia functional groups, our knowledge of their physiology and ecology (especially cyst / akinete formation and germination, and grazing or other losses) is quite limited, and the model is only able to reproduce their dynamics with limited success. For management applications, the ability to predict phytoplankton bloom composition is not necessarily critical if the model is able to reproduce other indicators such as chlorophyll, nutrients and oxygen. However, Nodularia may be particularly important, given its ability to fix nitrogen. The long-term EPA data suggest that very large Nodularia blooms have in the past resulted in extremely high organic N concentrations in the Lakes. Nodularia (or other N-fixing cyanobacteria) could strongly affect outcomes if management strategies led to differential reductions in N and P loads, or if it turned out that efflux of P from sediments continued long-term after reductions in both N and P loads.

While there is some uncertainty in the physical exchanges represented in the box model, and derived from the hydrodynamic model, it appears that these have only a marginal effect on the model conclusions. The model predicts very long flushing times for the Lakes under low flow conditions. These would have to be much shorter for flushing to play an important role in the short-term cycling of N and P, and this would be incompatible with salinity observations. There is more uncertainty about the hydrodynamic and box model representation of vertical mixing. Comparison with salinity observations suggests the model may over-estimate vertical mixing rates. However, if anything the model appears to over-estimate the build-up of bottom ammonia and depletion of bottom oxygen in L. Victoria at least, suggesting that the effects of any increased vertical mixing are small compared with the effects of biogeochemical fluxes.
8 References


Fig. 10. L. Wellington: comparison of predicted (solid line) and measured ammonia, nitrate (NOx) and phosphate (DIP) at EPA site 2306, in surface (asterisks) and bottom (squares) waters.
Fig. 11. L. Wellington: comparison of predicted (solid line) and observed chlorophyll a, organic N (TN - DIN), and non-PO₄ P (TP - DIP) at EPA site 2306, in surface (asterisks) and bottom (squares) waters.
Fig. 12. L. Wellington: comparison of predicted (solid line) and observed dissolved oxygen, TSS (NFR), and Secchi depth at EPA site 2306, in surface (asterisks) and bottom (squares) waters.
Fig. 13. Western L. Victoria: comparison of predicted (solid line) and measured ammonia, nitrate (NOx) and phosphate (DIP) at MAFRI site 8 (asterisks), for surface waters.
Fig. 14. Western L. Victoria: comparison of predicted (solid line) and measured ammonia, nitrate (NOx) and phosphate (DIP) at MAFRI site 8 (asterisks), for **bottom** waters.
Fig. 15. Western L. Victoria: comparison of predicted (solid line) and measured chlorophyll a, organic N (TN - DIN), and non-PO₄ P (TP - DIP) at MAFRI site 8 (asterisks), for surface waters.
Fig. 16. Western L. Victoria: comparison of predicted (solid line) and measured chlorophyll a, organic N (TN - DIN), and non-PO$_4$ P (TP - DIP) at MAFRI site 8 (asterisks), for **bottom** waters.
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Fig. 20. Central L. Victoria: comparison of predicted (solid line) and measured ammonia, nitrate (NOx) and phosphate (DIP) at EPA site 2311 (asterisks), for bottom waters.
Fig. 21. Central L. Victoria: comparison of predicted (solid line) and measured chlorophyll a, organic N (TN - DIN), and non-PO₄ P (TP - DIP) at EPA site 2311 (asterisks), for surface waters.
Fig. 22. Central L. Victoria: comparison of predicted (solid line) and measured chlorophyll a, organic N (TN - DIN), and non-PO₄ P (TP - DIP) at EPA site 2311 (asterisks), for bottom waters.
Fig. 23. Central L. Victoria: comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at EPA site 2311 (asterisks), for surface waters.
Fig. 24. Central L. Victoria: comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at EPA site 2311 (asterisks), for bottom waters.
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Fig. 29. Eastern L. Victoria: comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at MAFRI site 4 (asterisks), for surface waters.
Fig. 30. Eastern L. Victoria: comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at MAFRI site 4 (asterisks), for bottom waters.
Fig. 31. Southern L. King: comparison of predicted (solid line) and measured ammonia, nitrate (NOx) and phosphate (DIP) at MAFRI site 19 (asterisks), and EPA site 2314 (squares), for surface waters.
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Fig. 34. Southern L. King: comparison of predicted (solid line) and measured chlorophyll a, organic N (TN - DIN), and non-PO₄ P (TP - DIP) at MAFRI site 19 (asterisks), and EPA site 2314 (squares), for **bottom** waters.
Fig. 35. Southern L. King: comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at MAFRI site 19 (asterisks), and EPA site 2314 (squares), for surface waters.
Fig. 36. Southern L. King: comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at MAFRI site 19 (asterisks), and EPA site 2314 (squares), for **bottom** waters.
Fig. 37. Northern L. King: comparison of predicted (solid line) and measured ammonia, nitrate (NOx) and phosphate (DIP) at MAFRI site 18 (asterisks), and EPA site 2316 (squares), for surface waters.
Fig. 38. Northern L. King: comparison of predicted (solid line) and measured ammonia, nitrate (NOx) and phosphate (DIP) at MAFRI site 18 (asterisks), and EPA site 2316 (squares), for bottom waters.
Fig. 39. Northern L. King: comparison of predicted (solid line) and measured chlorophyll a, organic N (TN - DIN), and non-PO₄ P (TP - DIP) at MAFRI site 18 (asterisks), and EPA site 2316 (squares), for surface waters.
Fig. 40. Northern L. King: comparison of predicted (solid line) and measured chlorophyll a, organic N (TN - DIN), and non-PO₄ P (TP - DIP) at MAFRI site 18 (asterisks), and EPA site 2316 (squares), for **bottom** waters.
Fig. 41. Northern L. King: comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at MAFRI site 18 (asterisks), and EPA site 2316 (squares), for \textit{surface} waters.
Fig. 42. Northern L. King: comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at MAFRI site 18 (asterisks), and EPA site 2316 (squares), for **bottom** waters.
Fig. 43. “Entrance” (Shaving Point): comparison of predicted (solid line) and measured ammonia, nitrate (NOx) and phosphate (DIP) at EPA site 2322 (asterisks), for surface waters.
Fig. 44. “Entrance” (Shaving Point): comparison of predicted (solid line) and measured ammonia, nitrate (NOx) and phosphate (DIP) at EPA site 2322 (asterisks), for **bottom** waters.
Fig. 45. “Entrance” (Shaving Point): comparison of predicted (solid line) and measured chlorophyll a, organic N (TN - DIN), and non-PO₄ P (TP - DIP) at EPA site 2322 (asterisks), for surface waters.
Fig. 46. “Entrance” (Shaving Point): comparison of predicted (solid line) and measured chlorophyll a, organic N (TN - DIN), and non-PO₄ P (TP - DIP) at EPA site 2322 (asterisks), for bottom waters.
Fig. 47. “Entrance” (Shaving Point): comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at EPA site 2322 (asterisks), for surface waters.
Fig. 48. “Entrance” (Shaving Point): comparison of predicted (solid line) and measured dissolved oxygen, TSS (NFR), and Secchi depth at EPA site 2322 (asterisks), for bottom waters.
Fig. 49. Predicted time series of chlorophyll a in surface waters in L. Wellington, L. Victoria and L. King, for the full July1995 to June1999 baseline period.
Fig. 50. Predicted time series of ammonia in surface waters in L. Wellington, L. Victoria and L. King, for the full July1995 to June1999 baseline period.
Fig. 51. Predicted time series of phosphate (DIP) in surface waters in L. Wellington, L. Victoria and L. King, for the full July1995 to June1999 baseline period.
Fig. 52. Predicted time series of ammonia in bottom waters in L. Victoria and L. King, for the full July1995 to June1999 baseline period.

Fig. 53. Predicted time series of phosphate (DIP) in bottom waters in L. Victoria and L. King, for the full July1995 to June1999 baseline period.
Fig. 54. Predicted time series of dissolved oxygen in bottom waters in L. Victoria and L. King, for the full July 1995 to June 1999 baseline period.