Gippsland Lakes Environmental Study

Assessing Options for Improving Water Quality and Ecological Function

**Final Report
November 2001**

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Gippsland Coastal Board

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Prepared for the **Gippsland Coastal Board**

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Limitations Statement

Acknowledgements

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Executive Summary

The Gippsland Lakes represent a unique aquatic ecosystem and their beneficial use ranges from recreation and tourism to commercial fishing. Catchment modifications, together with the creation of a permanently open Entrance to Bass Strait in 1889, have resulted in a Lakes' environment that is substantially altered from its condition two centuries ago. Major water quality concerns include recurring blooms of the blue-green cyanobacterium *Nodularia* and extended periods of bottom water hypoxia (the reduction in oxygen concentration due to bacterial consumption of algal detritus). In response to the need to investigate strategies for ameliorating the environmental condition of the Lakes and following recommendations from the Gippsland Lakes Environmental Audit (Harris et al. 1998), the Gippsland Coastal Board (GCB) commissioned CSIRO to undertake the Gippsland Lakes Environmental Study (GLES).

The GLES is a partnership between the Department of Natural Resources and Environment (NRE) and the CSIRO for the GCB and stakeholders including Government agencies, regional industry, catchment and water authorities. The project was overseen by a Steering Committee comprising representatives from the GCB, NRE, West and East Gippsland Catchment Management Authorities, Gippsland Water, East Gippsland Water, Southern Rural Water, Loy Yang Power, Environment Protection Authority Victoria (EPA) and Parks Victoria. The project aims to help managers to understand the biogeochemical function of the Gippsland Lakes to the level of primary production including the factors controlling water quality and algal blooms. It also aims to provide managers with the capacity to assess options to address these problems. This is the Final Report for the project.

A hydrodynamic model is used to investigate the alterations to the flushing characteristics and salinity regime associated with hypthothetical engineering options and hypothetical changes to river discharge. A biogeochemical model simulates the fate of nutrients and primary production within the Lakes. It is used to examine the effects of changes in flushing, nutrient loads and sediment loads on water and sediment quality, and plant growth in the Lakes' ecosystem. A set of scenarios involving testing the effects on system behaviour of changes to river discharge, nutrient loads and exchange between the Lakes and the ocean were agreed with stakeholders via the Steering Committee for the project. The models were used to simulate the response of the Lakes to these scenarios and as an aid to understanding system behaviour. They were not used to predict the probability of different outcomes or events arising from the scenarios.

The GLES does not address possible impacts on invertebrates, fish, and birds. Nevertheless, the Lakes' higher level ecological function is based upon its biogeochemical function. Consequently, the impact of management actions eventually applied to the Lakes on their higher level ecological function will be determined (and understood) largely through alterations to nutrient cycling and primary production.

Flow and exchange in the Lakes

The Gippsland Lakes are a series of large, shallow, coastal lagoons connected to the ocean by a narrow, permanent man-made channel (Lakes Entrance) at their eastern end. Lakes Wellington, Victoria, and King are the three main water bodies. The annual river

discharges are highly variable from year to year, and a large proportion of the annual flow is typically delivered in 'events' of relatively short duration. This variability in the flows and the associated delivery of suspended sediments and nutrients has a profound influence on the biogeochemical function of the Lakes and their environmental health.

The salinity within Lake Wellington is largely determined by the balance between the freshwater inflow from the western rivers (principally the La Trobe, Thomson, and Avon Rivers) and by mixing of saline water from western Lake Victoria through McLennans Strait. Lake Wellington tends to be relatively fresh and well-mixed compared to the other Lakes.

Lakes Victoria and King are relatively deep compared to Lake Wellington and both demonstrate a significant degree of salinity stratification most of the time. This stratification limits vertical mixing and allows the bottom waters to become hypoxic. During periods of high freshwater inflow caused by rainfall events in the catchments and typically occurring in winter and spring, surface salinities are reduced as the surface layer is 'flushed' out of the system and stratification is at its strongest.

The hydrodynamic model shows that, during periods of low run-off, the exchange between the Lakes and Bass Strait is mainly due to flow driven through the Entrance by low-frequency oceanic water level variations associated with weather systems and low-frequency oceanic signals. Flushing times of the main bodies of the Lakes are approximately 6 months during low-flow periods, however this is dramatically reduced during flood events.

Nutrient cycling and water quality in the Lakes

The biogeochemical model simulates the response of the Lakes to catchment flows and loads of nutrients, sediments and organic matter by explicitly considering the (coupled) cycles of inorganic sediments, nitrogen, phosphorus and carbon within the Lakes system. The model includes four functional groups of phytoplankton, namely flagellates, diatoms, the cyanobacterium *Nodularia*, and dinoflagellates, as well as other primary producers including seagrass, macroalgae, and microphytobenthos. It considers processes within the water column, within the sediment, and at the sediment-water interface. Benthic processes and sediment – water column exchanges play a critical role in shallow coastal systems such as the Gippsland Lakes.

The model has been calibrated against MAFRI and the EPA time series data collected in the Lakes from July 1997 to June 1999. The analysis of these measurements and the model results leads to the following key conclusions about the biogeochemical function of the Gippsland Lakes, and their response to nutrient and sediment loads.

Lake Wellington tends to have a high phytoplankton biomass, but low concentrations of dissolved inorganic nitrogen due to phytoplankton uptake. Between major run-off events, growth is supported by base loads and by recycling and release of ammonia and phosphate from sediments.

High discharge events in Lakes King and Victoria flood surface waters with high nutrient concentrations, which result in large transient blooms. Stratification and the lack of vertical mixing in these Lakes create a positive feedback loop, in which organic

matter settling into bottom waters drives oxygen consumption and bottom-water hypoxia. This leads to shut down of denitrification, and to high rates of ammonia and phosphate release that can sustain further phytoplankton growth between run-off events, particularly of dinoflagellates that can migrate vertically. Periods of nitrogen-limitation, with elevated surface phosphate, favour *Nodularia* blooms.

There is high light attenuation in all the Lakes due to Coloured Dissolved Organic Matter (CDOM) input by rivers, to suspended sediments, and at times to dense phytoplankton blooms. Bottom plant growth is inhibited and this inhibits system recovery. Phytoplankton dominates primary production, a characteristic of a eutrophic system. All of these effects (high loads per unit volume of receiving waters concentrated in run-off events, stratification, and high light attenuation) combine to make the Gippsland Lakes particularly susceptible to eutrophication.

The construction of the Entrance caused the flora and fauna to switch from freshwater to marine species. Up until the late 1960s, Lake Wellington was clear and dominated by aquatic macrophytes, but the rise in salinity associated with the drought of 1968 killed the macrophytes and the Lake has been turbid and dominated by phytoplankton ever since. In Lake Victoria and Lake King, climate (and rainfall) variability appear to be a major determinant of water quality. Chlorophyll measurements from the Lakes over the last 15 years do not show a clear trend, but the analysis is confounded by the very large interannual variability evident in the data.

Inevitably, there is imperfect agreement between model and observations. Such disagreement can arise from a number of sources including limitations in the structure and formulation of the model, errors in model parameters, and errors due to spatial and temporal aliasing in observations. In general, the model captures the key processes controlling the biogeochemical cycles of nutrients and organic matter in Gippsland Lakes, and their response to changes in catchment loads, on time scales of days to years. The calibrated model reproduces the spatial contrasts among basins, and the event, seasonal and interannual contrasts over time, in key biogeochemical indicators (nutrients, chlorophyll, dissolved oxygen, light attenuation).

Analysis of model scenarios

The simulated response to each scenario was obtained by applying the hydrodynamic and biogeochemical models using specified inputs of nutrient and suspended sediment loads and freshwater inflows. The measured loads and flows for the 4-year period July 1995 to June 1999, used in model calibration and analysis of the current condition, defined the *baseline scenario*. The other scenarios utilised modifications to the flows and loads in the baseline scenario. Except for the scenarios involving the Macalister Irrigation District, the loads of nitrogen, phosphorus, and suspended sediment were all altered by the same proportion for each scenario. In formulating the scenarios, there was no attempt to predict the effects of specific actions in the catchments, such as changes in land use or water allocation.

Phytoplankton biomass in all three basins responds approximately proportionately to changes in total load with some departures associated with bloom dynamics for individual functional groups. Reducing loads showed a more than proportional improvement in bottom water condition (reduced hypoxia and nutrient release)

indicating that even small reductions in nutrient loads are worthwhile. However, the model predicts that very substantial reductions in loads, of order 50 to 70%, are required to eliminate episodes of bottom-water hypoxia. Conversely, increasing loads by 1% per annum produced an increase in phytoplankton biomass approximately proportional to loads, but a more significant increase in the duration and (potentially) the impact of periods of bottom water hypoxia.

Predicted phytoplankton species composition changes with loads in ways that reflect a complex set of biochemical and ecological interactions. Because *Nodularia* is nitrogen-fixing, it appears to receive a competitive advantage under small reductions in nitrogen loads. However, as loads are reduced further, the reduction in phosphorus load results in lower biomass of this cyanobacterium as well.

Loads from western catchments (to Lake Wellington) are two to three times those from eastern catchments input to Lake King via the Mitchell, Nicholson and Tambo Rivers in the 1995-99 period. Reducing loads from the eastern catchments has little impact on Lake Wellington and only a modest impact on the other two Lakes, whereas reducing western loads has an almost equal impact in Lakes Wellington and Victoria to reducing total loads by the same percentage.

A 40% reduction in phosphorus load from the Macalister Irrigation District (MID) reduces the western catchment phosphorus load by approximately 10%. A 40% reduction in MID nitrogen load reduces the western catchment nitrogen load by only 3%. These relatively small load reductions decrease phytoplankton concentrations in the Lakes by about 10% or less, but *Nodularia* concentrations in Lakes Victoria and King are predicted to decrease by about 20%.

The model does not allow for the long-term release of nutrients from 'deep' stores within the sediments. From an analysis of sediment core data, the time scale for continued release from this store is estimated to range between 10 – 100 years depending on assumptions made. But 'deep' fluxes calculated from core data are less than half the modelled sediment fluxes, suggesting that nutrient fluxes would decline substantially when loads to the Lakes are decreased. Observations in the drought year of 1997/98 support a short recovery time scale. There was a considerable improvement in water quality, especially in Lakes Victoria and King, in response to low loads during this period, but this was also a time of reduced stratification.

We consider it unlikely that errors in calibration arising from errors in load estimates would change the qualitative response of model indicators to relative changes in loads, but the uncertainty in load estimates does mean that managers need to think carefully about how load reduction targets are set particularly in the light of high interannual variability. Other potentially significant uncertainties or limitations that will affect the model predictions results include the assumption of constant riverine CDOM concentrations, the role of benthic filter feeders (neglected in the model), and the assumed autecology of *Nodularia* and dinoflagellates. The assumption of differing autecology for these phytoplankton will alter bloom composition, but predictions of total phytoplankton biomass are subject to mass balance constraints and so are considered to be robust

With loads fixed, the dominant effect of altering river flows appears to be the effect on flushing: reduced flows result in reduced flushing and increased eutrophication, while increased flows have the opposite effect. Increased flows are also likely to increase entrainment and ventilation of the bottom layer and improve oxygen concentrations there, but the proportional benefits are less than those for reducing loads. The model reproduces salinity and stratification over river flows which vary by orders of magnitude, which lends reasonable confidence to model predictions concerning the relative effects of flushing and stratification for a 20% change in flows.

When the modelled depth of the Entrance channel is increased/decreased to cause an approximately 50% increase/decrease in tidal range, quite small changes in salinity and flushing in the main basins result. The effect on predicted chlorophyll concentrations are little affected in Lake Wellington and are modest (<10%) in the other two Lakes.

The hydrodynamic model was used to investigate the impact of constructing a second entrance to the Lakes near Ocean Grange, similar in cross-section to the present Entrance. The second entrance approximately doubles the rate at which the main body of the Lakes exchanges with Bass Strait, but would have a smaller effect on Lake Wellington. The model simulations show the median and 95%ile phytoplankton concentrations to reduce by a factor of two in Lakes Victoria and King, although peak levels following flood events are not affected. The *Nodularia* blooms in the autumn are predicted to be virtually eliminated probably due to increased salinity. It should be noted that this prediction depends on an assumed salinity tolerance for *Nodularia* in the Gippsland Lakes, and is therefore uncertain and needs to be tested through process studies. The model suggests that episodes of stratification, hypoxia, and nutrient accumulation in bottom waters in L. Victoria and L. King would still occur, but for significantly shorter periods. These results are based on the model prediction that vertical mixing would increase with the construction of a second entrance. The second entrance is not predicted to have significant benefits for nutrient cycling or phytoplankton blooms in L. Wellington.

It has been suggested that the environmental condition of the Gippsland Lakes might improve if they were returned to freshwater systems, by constructing locks across McLennans Strait and Lakes Entrance. A water budget analysis using discharges, evaporation rates and precipitation rates measured over the last 23 years was used to hindcast the water levels within the Lakes that would have resulted if such locks were in place. Even considering the uncertainty in the analysis (primarily associated with the estimation of evaporation rates), the results indicate that locks are unlikely to cause problematic water level changes in the Lakes except perhaps in periods of drought. Preventing saltwater ingress should reduce the frequency of occurrence of stratification and hypoxic conditions in bottom waters, but would have major implications for the ecology by shifting the Lakes from estuarine to freshwater.

Implications for management

A preliminary estimate suggests that pre-European loads of total suspended solids, total nitrogen, and total phosphorus were respectively 55%, 45%, and 75% less than current loads in western catchments and 10%, 10% and 25% less in eastern catchments. Overall load reductions of order 50 to 70%, required to return the Lakes to a mesotrophic condition, will require a concerted effort. In their analysis, Grayson et al. (2001b)

considered reductions in total nitrogen and phosphorus loads only. It is likely that the composition, and in particular the bio-available and labile fractions of the loads, would have been different in pre-European times especially in the western catchments which have a higher level of agricultural activity than the eastern catchments.

Small decreases in loads lead to more or less proportional decreases in chlorophyll, and disproportionate increases in bottom oxygen, due to non-linear feedbacks in the modelled nutrient and carbon cycles. Thus, there is predicted environmental benefit in achieving small reductions in catchment loads, or in avoiding incremental increases in catchment loads of the kind simulated in the long-term growth scenario.

Model uncertainties are unlikely to change significantly the predictions about the relative effects of small (<20%) changes in loads, but there is increased uncertainty about larger changes so it is not possible to assert confidently that 70% or 50% reductions in loads will eliminate bottom water hypoxia. The uncertainty about load estimates is not expected to adversely affect the conclusions about the relative response of indicators. However, managers should be very cautious about setting absolute load reduction targets due to uncertainty in load estimates and to the very high interannual variability in loads.

Decreases in river discharge (keeping loads fixed) have proportionally smaller impacts on chlorophyll and bottom water hypoxia as increases in load. The analysis of the impacts of changes in flow was restricted and could be extended to consider changes to base and peak flows separately and realistic changes to the flow-load relationship that are likely to result from catchment activities.

Although the second entrance appears to be superficially attractive in terms of water quality indicators in Lakes Victoria and King, it would have little or no beneficial effect on L. Wellington, would involve a very substantial capital outlay, and probably considerable ongoing expenditure to maintain dredged channels. The construction of locks to maintain the Lakes as freshwater systems faces similar considerations as a second entrance. Such works would also require substantial capital outlay and ongoing maintenance expenses.

Where possible management actions involve major capital outlays or major changes to the ecology of the system, as with the construction of a second entrance or locks, it would be prudent and desirable to reduce uncertainties up front. If a second entrance were to be seriously contemplated, further studies should include more detailed engineering models of flows around the entrances, and further process studies leading to refinement of the biogeochemical model such as enhancing its ability to predict *Nodularia* blooms. Due consideration would need to be applied to all the other major environmental consequences of a likely increased salinity regime in the Lakes and conjunctive waterways including the potential impacts on the flora, invertebrates, fish and birds. Similarly, turning the Lakes fresh would cause major ecological changes which are not addressed in this report, but would need to be evaluated by the affected communities and environmental managers very carefully if this strategy were to be considered. For incremental management actions, as for reductions in catchment loads, uncertainty is not so critical and an adaptive management approach might be appropriate, supported by an appropriate monitoring program.

Recommendations for continuing work

We suggest that a monitoring program be implemented with the prime purpose of assessing progress towards management objectives and to inform decisions on modification of management actions over time in response to system change. The large temporal variability of the Lakes at time scales of weeks to years will require decades of measurements before the analysis yields statistically significant results.
We propose that a more effective strategy is to use the hydrodynamic and biogeochemical models in diagnostic mode to continuously assess the cause and effect relationship between management actions and system biogeochemical response as evidenced in an ongoing monitoring program.

The suggestions for a monitoring program presented here are designed specifically to support the ongoing assessment of the effects of management actions on key indicators of the biogeochemical function of the Lakes, taking into account the diagnostic use of the hydrodynamic and biogeochemical models. Accurate evaluation of riverine loads of nutrients, CDOM, and suspended particulate matter has the highest priority in our proposed monitoring strategy. Other parameters that need to be monitored are in-lake salinity, dissolved oxygen, chlorophyll a, light attenuation, suspended solids, and nutrients.

While the hydrodynamic and biogeochemical models are generally able to reproduce most aspects of the observed system behaviour, at least qualitatively, the calibration process revealed a number of limitations that could be addressed with further process studies. We do not understand processes controlling long-term storage and release of phosphorus and nitrogen in sediments and need to know how much of the large measured nutrient stores are labile and what are the mechanisms and rates of material transport within the sediment column. A better understanding of phosphorus dynamics including adsorption/desorption is needed for assessing whether nitrogen or phosphorus is liable to limit phytoplankton growth. We require more understanding of dinoflagellate and *Nodularia* autecology and population dynamics to improve model representation of phytoplankton composition. The model representation of vertical mixing in Lakes Victoria and King is not as certain as it should be and needs to be verified or improved if the uncertainty surrounding the benefits of a second entrance is to be reduced.

1. Introduction

The Gippsland Lakes are a system of coastal lagoons situated about 200 km east of Melbourne. They represent a unique aquatic ecosystem and their beneficial use ranges from recreation and tourism to commercial fishing. European settlement began in the region in the 1840's and has resulted in considerable modification to the Lakes' catchments including the clearing of lowland and foothill forests, draining of wetlands, and diversion of water from the rivers for urban, agricultural, and industrial use. These changes have increased the delivery of nutrients and sediments to the Lakes and have altered the regime of freshwater inflow. Catchment modifications, together with the creation of a permanently open Entrance to Bass Strait in 1889, have resulted in a Lakes' environment that is substantially altered from its condition two centuries ago. Major water quality concerns include recurring blooms of the blue-green cyanobacterium *Nodularia* and extended periods of bottom water hypoxia (the reduction in oxygen concentration in the water column due to bacterial consumption of phytoplankton detritus). The Gippsland Lakes algal bloom (*Nodularia*) of the summer of 1987/88 caused significant economic loss to the Gippsland region through its deleterious impact upon the tourism industry and has been described as a disaster by Gordon (1988). Norman (1988) reported on the possible deleterious effects of the same bloom on aquatic life including birds, shellfish, crustaceans and fish.

In 1998 the Gippsland Coastal Board (GCB) commissioned an audit to collate the scattered existing information on the Lakes into a form that could provide a basis for more concerted management planning and action in the future. The Gippsland Lakes Environmental Audit (Harris et al. 1998) suggested that based upon existing measurements of nutrients and chlorophyll concentrations, the condition of the Lakes has not improved over the past 15 years. Further algal blooms have occurred during this time and are expected to recur.

The Gippsland Lakes Environmental Audit suggested that future management of the Lakes should be based on ecosystem modelling. It made nine recommendations in its conclusions including the following two:

> *"A three-dimensional model of the Lakes needs to be constructed to examine in more detail the precise interactions of freshwater and marine inflows, tidal mixing and Entrance dynamics and saltwater/freshwater balances. The hydrodynamic model would be integrated with an ecological model to provide a synthesis of scientific understanding of the ecosystem of the Gippsland Lakes..."*

and

> *"The integrated model should be used to predict the impacts of management decisions (scenarios) which are designed to alter the nutrient loads to the Lakes ..."*

To address these two recommendations, the GCB commissioned the Gippsland Lakes Environmental Study (GLES).

The GLES is a partnership between the Department of Natural Resources and Environment (NRE) and the CSIRO for the GCB and stakeholders including Government agencies, regional industry, catchment and water authorities. The project was overseen by a Steering Committee comprising representatives from the GCB, NRE, West and East Gippsland Catchment Management Authorities, Gippsland Water, East

Gippsland Water, Southern Rural Water, Loy Yang Power, Environment Protection Authority (EPA) Victoria, and Parks Victoria.

The project 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. These options include:

- Reduction of nutrients and sediment loads in the rivers flowing into the Lakes.
- Alterations to the river discharges to the Lakes due to further impoundment, increased irrigation extraction, or implementation of environmental flows.
- Alteration to the flushing of the Lakes due to engineering works such as the construction of a second entrance or the deepening of the present Entrance.

The design of the study was based on the premise that the modelling should rely on available data and that there was only a limited capacity to collect new data. A hydrodynamic model is used to describe the currents and mixing within the Lakes and how nutrients and other particulate and dissolved materials are transported throughout the Lakes. Besides simulating the hydrodynamics of the Lakes as they are, the model is also used to investigate alterations to the flushing characteristics and salinity regime associated with hypothetical engineering works and with changes to river discharge. A biogeochemical model simulates the fate of nutrients and primary production within the Lakes. It is used to examine the effects of changes in flushing, nutrient loads and sediment loads on a range of environmental indicators, including algal blooms and bottom water oxygen. A set of scenarios involving testing the effects on system behaviour of changes to river discharge, nutrient loads and exchange between the Lakes and the ocean were agreed with stakeholders via the Steering Committee for the project. The models were used to simulate the response of the Lakes to these scenarios and as an aid to understanding system behaviour. They were not used to predict the probability of different outcomes or events arising from the scenarios.

The GLES does not address possible impacts on invertebrates, fish, and birds. Nevertheless, the Lakes' higher level ecological function is based upon its biogeochemical function. Consequently, the impact of management actions eventually applied to the Lakes on their higher level ecological function will be determined (and understood) largely through alterations to nutrient cycling and primary production.

This is the Final Report of the GLES and its purpose is to provide an overview of the project and to present its key findings. The second and third sections of the report focus on conveying an understanding of the physical, chemical and biological processes that govern water quality and the ecological response to meteorological conditions and the riverine inputs of freshwater, nutrients, and suspended sediments. Section 2 describes the hydrodynamics of the system and Section 3 concentrates on nutrient cycling and water quality. Section 4 presents the results of the scenario analyses and includes discussion of the implication of model uncertainty on the results. The outputs are the simulated impacts on key environmental indicators (dissolved oxygen, frequency and intensity of algal blooms, etc.) of a specified set of alterations to loads, freshwater input, and exchanges between the Lakes and the ocean. In Section 5, we present a discussion of the implications of the results of the study for management of the Lakes. Finally, Section 6 uses the results and understanding gained from this study to recommend

monitoring strategies for assessing the ongoing condition of the Lakes and the effectiveness of implemented management actions. This section also presents recommendations for further process studies that would address key remaining knowledge gaps.

The report is supported by a series of more detailed technical reports. A list of these reports is provided in the Reference section. The reports are available at the Gippsland Coastal Board web site: http://www.vcc.vic.gov.au/gcboard/. It is also expected that a brief summary report will be produced for wider circulation beyond the Steering Committee.

2. Flow and exchange in the Lakes

The GLES required us to describe the currents and mixing within the Lakes in order to determine how solutes and particulate material are transported throughout the system. The horizontal and vertical exchanges within the Lakes are necessary inputs to the biogeochemical model, which is used to simulate the biogeochemical and primary production response. Numerical models were used to determine the exchanges and flows within the Lakes. In combination with the measurements, the models were also used to develop an understanding of the hydrodynamics of the Lakes and how these respond to changes in river flows and engineering modifications.

Geography and bathymetry

The Gippsland Lakes are a series of large, shallow, coastal lagoons approximately 69 km in length and 10 km wide at the widest point, connected to the ocean by a narrow, permanent man-made channel (Lakes Entrance) at their eastern end (Figure 2.1). Lakes Wellington, Victoria, and King are the three main water bodies and constitute most of the 364 km^2 total area of the Gippsland Lakes (Table 2.1). Lake Reeve, lying along the coastal dune barrier, has an area of 50 km^2, but it is usually dry except during times of high rainfall and has not been included in the Lakes' total area.
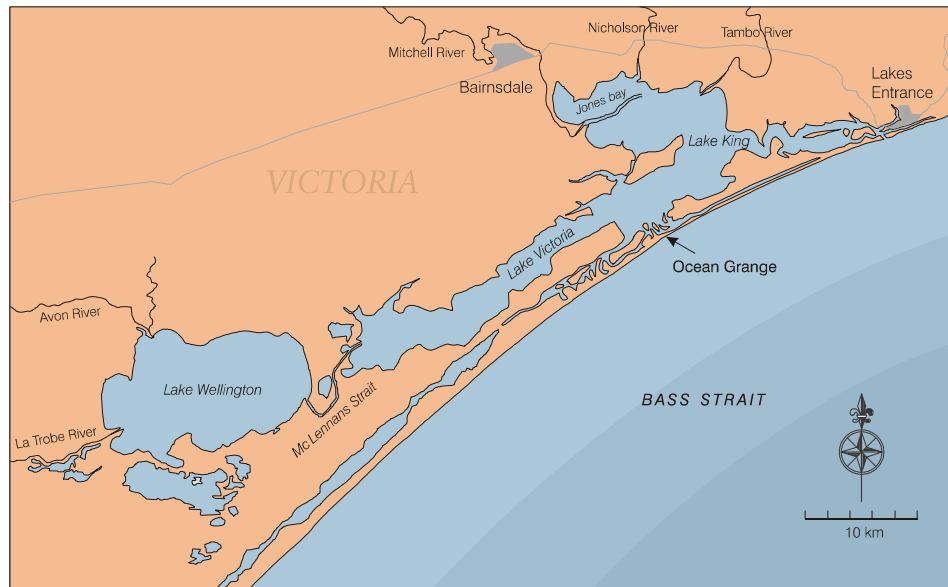
Figure 2.1 The Gippsland Lakes.

	L. Wellington	L. Victoria	L. King
Area (km^2)	148	75	98
Mean Depth (m)	2.6	4.8	5.4
Max. Depth (m)	6	9	10
Salinity	0.5-10	4-17 (Surface) 7-25 (Bottom)	8-26 (Surface) 15-36 (Bottom)

Table 2.1 Characteristics of the Gippsland Lakes.

Five major rivers, the La Trobe, Avon, Mitchell, Nicholson, and Tambo, flow into the Lakes and drain an area of 20,600 km^2. Prior to the construction of the navigation channel to the sea at Lakes Entrance in 1889, the Lakes were probably fresh much of the time, but would have had some salinity while the barrier was breached (Collett 1987). The Lakes are now considered to be estuarine. Being more closely connected to the sea than is L. Wellington, L. Victoria and L. King show a stronger marine influence. These latter two Lakes are deep enough (Table 2.1, Figure 2.2) that they stratify in salinity, which will be shown to have significant ecological ramifications. The connection between L. Wellington and L. Victoria occurs via McLennans Strait, a channel of 9.7 km length, 200 m width, and 5 m mean depth. The restricted nature of the connection between L. Wellington and the other Lakes (and hence to the sea) causes the Lake to be relatively fresh. Lake Wellington is characterised as being a flat-bottomed basin of fairly uniform, shallow depth (Figure 2.2).
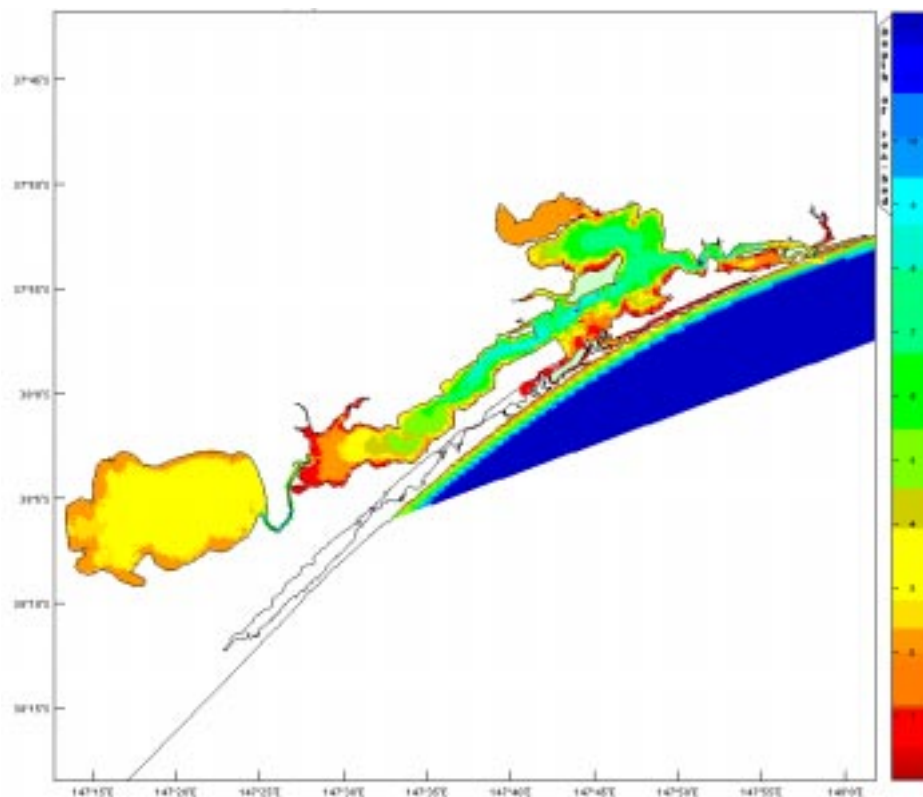
Figure 2.2 Bathymetry of the Gippsland Lakes as used in the hydrodynamic model application. Note that depths in the coastal region are restricted to a maximum of 12 m for computational efficiency.

Modelling flow and exchange

An inverse and a hydrodynamic model were used to obtain the required quantitative knowledge of currents and mixing within the Lakes. The *inverse model* uses available measurements of salinity throughout the Lakes combined with known freshwater inputs to infer the exchanges of water and salt between various sections of the Lakes. The *hydrodynamic model* applied to the Lakes solves the continuity, momentum, and

advection/diffusion equations, to simulate the three-dimensional distributions of velocity, temperature and salinity. The hydrodynamic model is used to estimate the currents and mixing within the Lakes in the study period, and also to predict how the currents and mixing will change under model scenarios involving alterations to the river discharges, modifications to the existing Entrance and construction of a second entrance. The hydrodynamic model and its application to the Gippsland Lakes are described by Walker and Andrewartha (2000) and what follows is largely based upon their report. A summary description of the model is provided in Appendix A.

The data required for the application of the inverse model are time series of the net freshwater input into specified major basins within the Lakes and the measured salinities within these basins. Water column salinity was measured at 1-m increments at 18 sites over the 2-year period between 1 July 1997 to 30 June 1999 at fortnightly intervals by MAFRI (Longmore 2000b). Sampling sites were located throughout L. Victoria and L. King. No measurements were obtained in L. Wellington as part of this measurement program, so data collected by the EPA were used instead. The EPA collects salinity data for L. Wellington (site number 2306) every second month (monthly prior to 1995) on an ongoing basis at both the surface and bottom of the water column. These measurements were also used for the validation and calibration of the hydrodynamic model. In addition to river discharge, the hydrodynamic model also requires that other 'forcing' time series be specified including the wind stress on the water surface, atmospheric pressure gradients, sea level (including tides) and surface heat fluxes. We were able to compare water levels predicted by the hydrodynamic model to continuous water level measurements obtained at 14 sites throughout the Lakes by the University of Melbourne since November 1998 (Grayson et al., 1999).

River discharges, precipitation, evaporation

River discharge is the main source of freshwater input into the Lakes and is a key determinant of the hydrodynamics of the system. The net freshwater discharge forces water to flow through the Lakes and so provides a direct mechanism for flushing the Lakes. Further, the input of freshwater at one end of the Lakes and exchange with seawater at the other allows salinity stratification to occur within the deeper basins. Thirdly, because the rivers carry most of the nutrients and sediment input into the Lakes, nutrient and sediment loads exhibit similar temporal behaviour to the river hydrographs.

For this project, Grayson et al. (2001a) have tabulated discharges over a 24-year period for the main rivers flowing into L. Wellington (the western rivers – the La Trobe, Thomson, and Avon) and the main rivers flowing into L. King (the eastern rivers – the Mitchell, Nicholson and Tambo). Although river discharges into the Lakes show a seasonal pattern of low flows during summer and autumn and higher flows in winter and spring, the annual averages tend to be highly variable and a large proportion of the total flow is delivered in 'events' of relatively short duration (Figure 2.3 and Figure 2.4).

In most years, the western rivers have a higher annual discharge than the eastern rivers. The average flow in the western rivers between 1975-99 was 4,380 ML d^{-1} versus 3,080 ML d^{-1} in the eastern rivers. The last two years of the discharge record, which are also the period for model calibration, represent two of the three years in the 24-year record

for which the eastern discharge exceeds the western discharge. The eastern discharges were not unusually high during these two years; rather the western discharges were lower than normal. The scenarios were tested using flows and loads from the years 1995-99. The year 1995-96 shows moderately high discharge from the western rivers, whereas discharges in the following year are close to the median discharges for both the western and eastern rivers. Because of the importance of individual flow events (Figure 2.4), annual statistics can depend strongly on the month selected as start of year. The annual discharges for the calendar year 1997 for the western and eastern rivers are only 1,540 and 910 ML d^{-1} respectively. These discharges are only about a third of the 24-year averages.
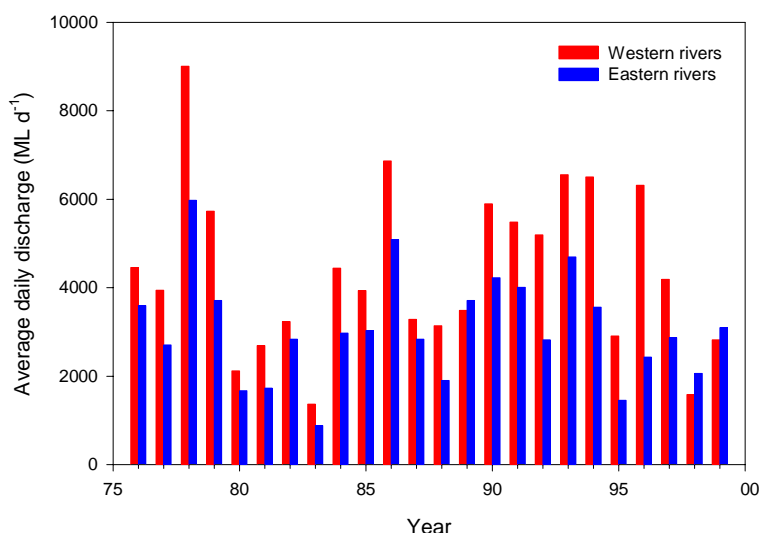
Figure 2.3 Daily discharges averaged annually for the western and eastern rivers. The averaging time is July of a particular year to June of the following year. For example, the bars centered on year '76 are the average discharges between July 1 1975 and June 30 1976.

The episodic nature of the flows in the Gippsland rivers is illustrated in Figure 2.4 for the period used in the scenarios. The discharge from the western rivers in 1995-99 is dominated by two flow events occurring in late October and early November 1995. Between them, these two events accounted for 33% of the discharge for the period between July 1995 and June 1996, but they represented only 6% of the time (flow exceeded 10,000 ML d^{-1}). In the last week of June 1998, floods in the eastern rivers delivered 440,000 ML of water to L. King. This volume represents 58% of the total water discharged to the Lake between July 1997 and June 1998 and is a third larger than the total volume of water delivered to the Lake in calendar year 1997 (330,000 ML). The flow in the western rivers was also elevated during the June 1998 flood mainly due to contributions from the Avon River, which drains the most eastern catchment of the rivers flowing into L. Wellington.
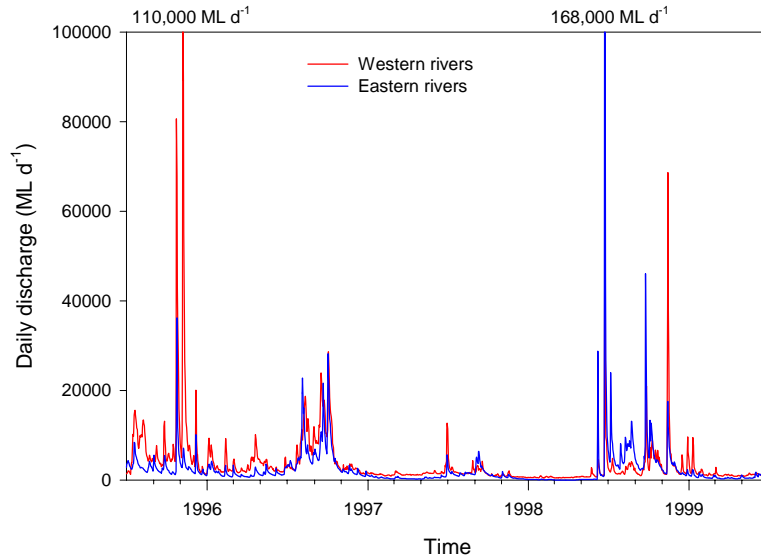
Figure 2.4 Daily discharges for the western and eastern rivers.

Other important contributors to the freshwater balance in the Lakes are precipitation falling on the Lakes surface and evaporation. Daily precipitation and pan evaporation rates have been measured at the East Sale RAAF base (via the Bureau of Meteorology). The average annual precipitation and evaporation rates between 1975 and 1999 were 1.6 mm d^{-1} and 3.7 mm d^{-1}, respectively. Over the surface areas of the Lakes (364 km^2) these rates represent daily volumetric gains and losses of 600 ML d^{-1} and 1,330 ML d^{-1}, which are 8% and 18% respectively of the combined average river inflow to the Lakes. Precipitation does not show a strong seasonal variation but evaporation ranges between a low of 1.4 mm d^{-1} in June to a maximum of 6.4 mm d^{-1} in January. In February, the month of lowest river discharge, the rate of water loss through evaporation exceeds the river input on average although the contribution of precipitation is large enough for the freshwater water balance to be slightly positive.

Other hydrodynamic forcing

Besides the freshwater inflow, other important drivers of the hydrodynamic response of the Lakes include the water levels in Bass Strait at the Entrance and a series of meteorological parameters including the wind, air temperature, and solar radiation. Winds and other meteorological parameters required to run the Lakes' hydrodynamic model were measured at East Sale RAAF base. Sea level variations and the long-shore and cross-shore sea-surface slopes in the coastal zone offshore from the Entrance are modelled using a 5-km resolution model of Bass Strait, forced by tides, low-frequency sea-level variations observed at locations on either side of the Strait, atmospheric pressure, and winds.

Water level measurements at a number of sites inside the Lakes were obtained from the University of Melbourne (Grayson et al. 1999) and used to calibrate and validate the model simulations. Figure 2.5 compares measured and modelled water levels at a site in L. King. The comparison demonstrates that the model does a reasonable job of predicting water levels, but it also illustrates important features of the water level response within the Lakes. The tidal range within the Lakes is small (of order 5 cm) despite the tidal range on the adjacent coast being about 1 m. This is due to massive attenuation through the artificial channel at Lakes Entrance and due to some further attenuation in the channels leading to Metung. In the main bodies of L. Victoria and L. King, water level follows the water level in Bass Strait reasonably closely on time scales of 1 week or more. These low-frequency variations, driven by weather systems and low-frequency oceanic signals propagating through Bass Strait, can alter sea level by tens of centimetres or more, on time scales of five to ten days.
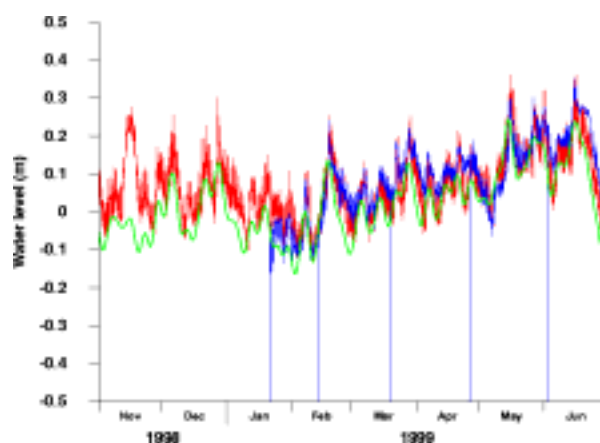
Figure 2.5 Water levels measured (blue) and modelled (red) in L. King near the mouth of the Nicholson River. The low-frequency water levels in Bass Strait at Lakes Entrance are also shown (green). The vertical lines are due to servicing of the water-level recorder.

Due to the restriction imposed by McLennans Strait the tides in L. Wellington are even smaller (~2 cm range) than they are in the main body of the Lakes. Water levels in L. Wellington follow Bass Strait water levels on long time scales (months), but are also substantially influenced by river inflows and the wind. Measured and modelled water levels for a 2-month period in this Lake are shown in Figure 2.6. The elevated water levels in L. Wellington in mid-November 1998 are due to high river discharges into the Lake (see Figure 2.4). The discrepancy between measured and modelled water levels at this time is likely to be due to the inability of the model to represent the spreading of higher water levels into the wetlands adjacent to L. Wellington and possible release of water into Lake Coleman. Water levels within the Lakes can rise by more than 2 m as a result of high river discharges (Lester 1983). The sharp drop in water level of ~0.6 m in the last week of December 1998 is the response to strong westerly winds which caused an upward tilt in the water surface towards the eastern side of L. Wellington.

Both measured and modelled water levels show fluctuations of ~15 cm in amplitude having periods of about a week, and these tend to show the opposite phase to fluctuations in Bass Strait. Westerly winds cause water levels to depress at the L.

Wellington measurement site. Conversely, a westerly wind would cause water to pile-up against the Victorian shore of Bass Strait due to the action of Coriolis force. The predominant wind direction over the Lakes is westerly, with the infrequent strong winds being westerly (Lester, 1983) and occurring in late winter and spring. Easterly and westerly winds each occur for about 20% of the time in summer.
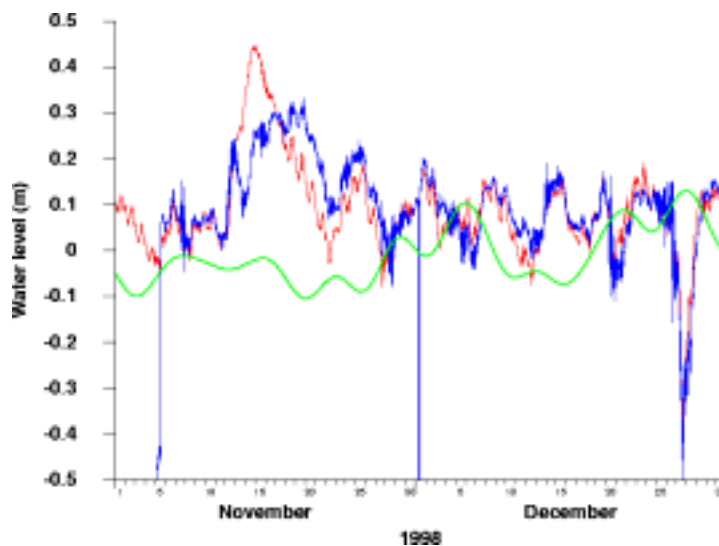
Figure 2.6 Measured (blue) and modelled (red) water levels near the mouth of the La Trobe River in western L. Wellington. Bass Strait sea level is green.

Winds are an important agent for causing water levels to vary throughout the Lakes and also for driving currents and mixing within the Lakes. An elevated water level along the eastern side of L. Wellington induced by westerly winds would cause the water level at the western end of McLennans Strait to be higher than at its eastern end. An outflow from L. Wellington into L. Victoria would result. Within the bodies of the Lakes, currents are induced through the direct action of the wind stress on the water surface. This is certain to be an important mechanism for horizontal mixing within the Lakes. Turbulence generated by the wind-induced shear flow at the water surface acts to mix the water column vertically as well. In L. Wellington, the water is shallow enough and the winds are strong enough that the water is generally in a well-mixed state both horizontally and vertically. In L. Victoria and L. King which are deeper than L. Wellington, wind-induced turbulence acts to continuously erode the stratification.

The presence of stratification (or vertical variation) of a water property such as salinity or temperature is an indicator of a lack of complete vertical mixing, but also density stratification acts to inhibit vertical mixing. The degree of stratification within these Lakes will be shown to have important biogeochemical consequences.

Because water density is determined by its temperature as well as by its salinity, non-uniform temperature in the water column affects the degree of density stratification and can thereby impact the mixing dynamics. Figure 2.7 shows the measured temperatures at seven depths in L. King during the 2-year calibration period for the hydrodynamic model, 1997-95 (Longmore, 2000b). The difference in temperature between the temperatures at 1 m and 7 m depths is mostly less than 3^0C. A temperature variation of this size has the same effect on density as a salinity variation of 0.8. Since the salinity

difference between 1 m and 7 m depths was usually at least several times larger than this in L. Victoria and L. King, we conclude that temperature stratification is of less significance than is salinity stratification in the dynamics of these Lakes.
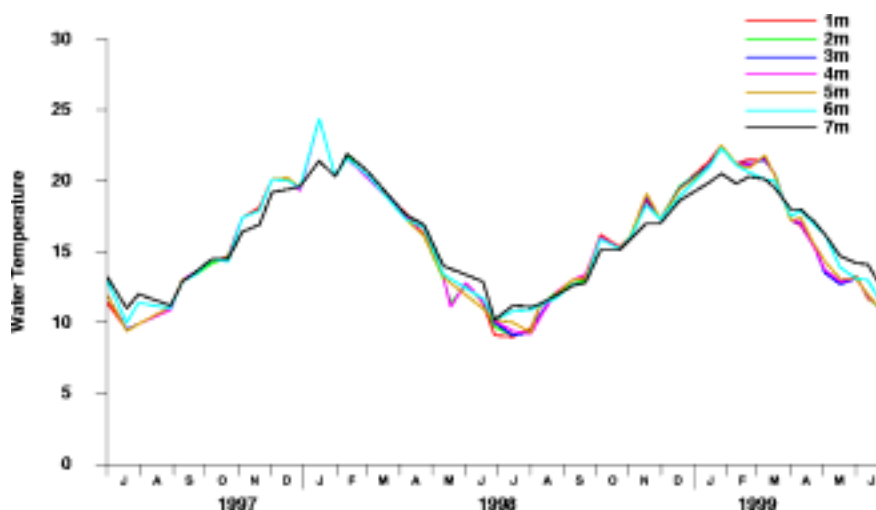
Figure 2.7 Temperatures at 7 depths (1–7 m) at MAFRI site 2 (eastern L. King).

The seasonal variation in temperature is pronounced and consistent from year to year. All Lakes show a similar pattern to that measured in L. King (Figure 2.7). Water temperatures reach a maximum of ~22^{0}C in January and a minimum of ~10^{0}C in July. Temperature profoundly affects the rates of reaction of many biochemical and ecological processes and so it was necessary to account for this dependence in the biogeochemical model. Water temperatures were modelled successfully by the hydrodynamic model. Required input for the thermodynamic component of the model included solar radiation, air temperature, evaporative heat loss, cloud cover, humidity, river temperature, and the water temperature in Bass Strait. Some of the meteorological inputs were estimated from measurements made at the East Sale RAAF base. River temperatures were measured by MAFRI in the Tambo and Nicholson Rivers. Bass Strait water temperatures followed temperatures measured during the Port Phillip Bay Environmental Study (Harris et al. 1996).

Stratification and circulation

Figure 2.8 shows the conceptual model of circulation and salinity distribution for a time following a period of elevated discharge in the western and eastern rivers. This picture is largely built upon the work of Bek and Bruton (1979) and confirmed by the results of Lester (1983), on the results of the inverse and hydrodynamic models developed as part of the GLES, and on the interpretation of intensive physical measurements made throughout L. Victoria and L. King during 1997-99 (Longmore 2000b).
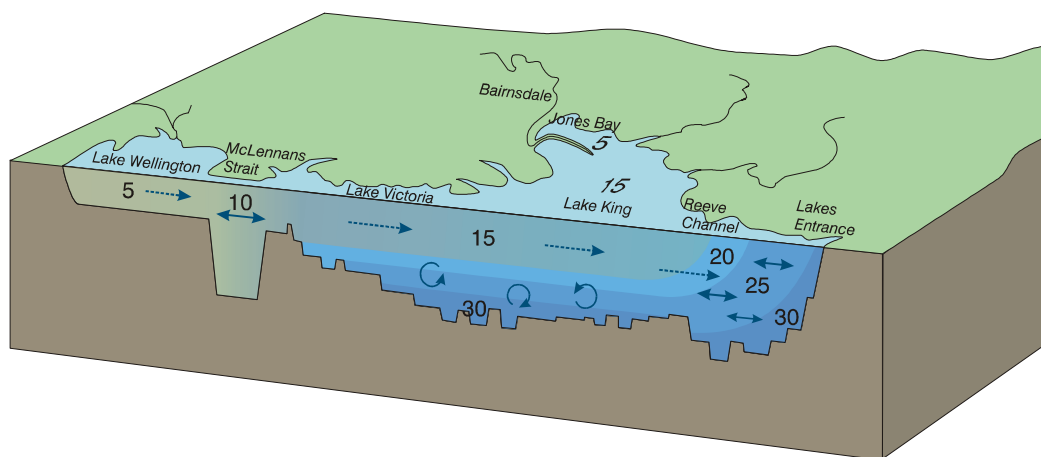
Figure 2.8 The conceptual model of stratification and circulation within the Gippsland Lakes. The numbers denote representative salinities and the colour scheme is indicative of salinity variation within the water column.

Lake Wellington

Lake Wellington can be treated as well-mixed both vertically and horizontally. The salinity within the Lake is largely determined by the balance between the freshwater inflow from the western rivers and by mixing of saline water from western L. Victoria through McLennans Strait. Loss of water through evaporation from the water surface is an important factor affecting the Lakes' salinity during times of low river discharge particularly during the summer months.

The normal pattern of riverine inflows to the Lakes is for minimum flows to occur in January and February and to reach a peak in September (Robinson, 1995). The normal pattern of salinity that results in L. Wellington is for maximum salinities of ~10 to occur in late spring-early winter and for minimum salinities of less than 5 to occur in late spring. Sufficiently large freshwater inflows can displace most of the water in L. Wellington and render it virtually fresh. Bek and Bruton (1979) report that the flood of June 1978 caused salinities in L. Wellington to reduce to ~1. Prior to the flood salinities were ~8. Droughts can result in high salinities especially if winter flushing of salt from the Lake does not occur. The drought of 1967/68 resulted in salinities within L. Wellington reaching 22.2 (Arnott 1968), which are the highest recorded.

Figure 2.9 shows the measured salinities for the calibration period 1997-99. Due to the small inflows through 1997 the salinity did not reduce to normal values in late spring (Oct.-Nov.). The salinity falls by about 5 due to the flow event in June 1998. Note that this flow event was primarily an elevated discharge into L. King so that its effect on L. Wellington is less pronounced than a major flow into the Lake might have been. Thereafter, the seasonal salinity variation reverts to its normal cycle. The figure also shows that the model represents the salinity variation reasonably well. It appears to under-predict salinities in November-February 1999, but this may be due to the lack of measurements in the middle of this period.
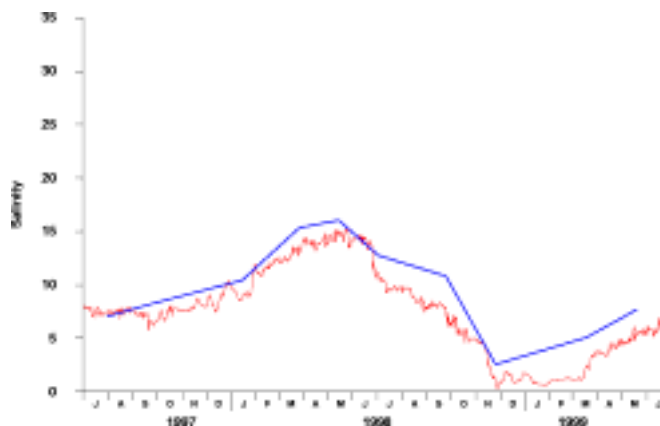
Figure 2.9 Observed (blue) and modelled (red) surface salinity at EPA site 2306 (L. Wellington).

Lakes Victoria and King

Lakes Victoria and King are relatively deep compared to L. Wellington (Table 2.1) and both demonstrate a significant degree of stratification in salinity most of the time. As in L. Wellington, salinities within L. Victoria and L. King respond to freshwater inflows from the rivers and to losses through evaporation. Brackish water enters L. Victoria from exchange with L. Wellington through McLennans Strait. The eastern rivers discharge into the northern end of L. King both directly and via Jones Bay, a shallow embayment (~2 m deep) formed by the silt jetties of the Mitchell River (Figure 2.1).

The presence of salinity stratification is not only an indicator that vertical mixing is suppressed, but it is also a cause of it. During periods of high freshwater inflow, surface salinities are reduced as the surface layer is 'flushed' out of the system and vertical mixing may cause salinities near the bottom to decrease also. This is a time of maximum density stratification in L. Victoria and L. King. Figure 2.10 shows how salinities in eastern L. King responded to the elevated flows in the eastern rivers in June 1998. After a year of low freshwater input into the Lakes, surface salinities had climbed to about 30. The June inflows caused the salinity in the top 4 m of the water column to drop to about 15, and there is an indication of a modest (~5) reduction in salinity near the bottom as a result of this flow event. Surface salinities within L. Victoria and L. King were reduced to 1-2 due to the unusually large floods of June 1978; bottom salinities reduced to 4-6 in these two Lakes (Bek and Bruton, 1979).

Once the freshwater inflows reduce, the salinity in the water column gradually increases. Salt is mixed upwards across the halocline and causes the salinity in the upper parts of the water column to increase. The more saline bottom waters are replenished by water of relatively high salinity from Reeve Channel, which being denser than the water in the Lakes' basins, flows in along the bottom. Thus the general pattern of flow within L. Victoria and L. King is a deep inflow of relatively saline water, upward mixing of saline water across the halocline, and an outflow of relatively fresh water nearer the water surface. Figure 2.10 shows how salinities increased during two periods of reduced flow before and after the flow events of June 1998. As a consequence of the low flows through 1997, the stratification in L. King almost disappeared at the end of the year. In L. Victoria (not shown) the stratification is mostly absent between June 1997 and the

June 1998 flow event. Surface and bottom salinity measurements obtained by the Victoria EPA between 1986-99, show that the absence of significant stratification in eastern L. Victoria and northern L. King through the first half of 1998 was an unusual occurrence.
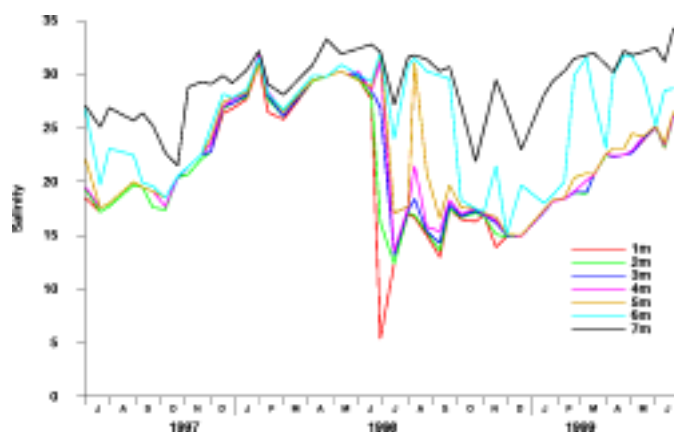
Figure 2.10 Salinities at 7 depths (1–7 m) at MAFRI site 2 (eastern L. King).
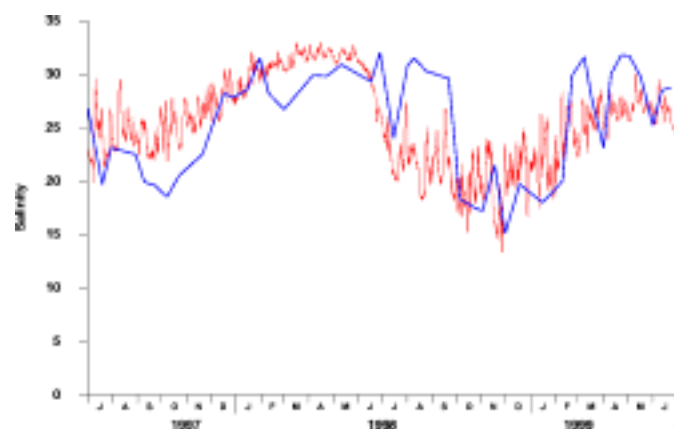
Figure 2.11 Observed (blue) and modelled (red) salinity at 6m depth, MAFRI site 2 (L. King).

The hydrodynamic model performs well in simulating the surface salinities throughout L. Victoria and L. King, but appears to do less well in simulating salinities near the bottom. Figure 2.11 illustrates the problem by comparing model simulations and measurements at a site in L. King. It appears that the model under-estimates bottom salinities, and therefore the degree of stratification, in the months following the June 1998 flood event. This behaviour could be due to the model overestimating vertical mixing or misrepresenting the intrusion of ocean water through Reeve Channel. However, as Figure 2.10 shows, the halocline through this time occurs between depths of 5 m and 7 m. A small error in the prediction of the depth of the halocline could result in what seems to be a large error in the salinity prediction. Further, internal waves or internal seiching at the site could cause the halocline to oscillate up and down so the exact time that a particular salinity measurement was made would affect the apparent

salinity at the measurement depth. Continuous recordings of salinity by Longmore (2000b) in L. Victoria at 7 m depth show short-term oscillations in salinity of 5-10 so that water samples collected at two-weekly intervals would be subject to significant aliasing.

Flushing of the Lakes

By flushing of the Lakes, we mean the ability of the Lakes to export to the ocean an introduced contaminant or some other tracer. This is potentially an important property since it helps to determine the concentrations and ecological impacts of nutrients and sediments within the Lakes. One mechanism for flushing the Lakes is the continuous through-flow of water required to balance the river inputs of fresh water, precipitation, and evaporation. The flushing time due to through-flow, which is the volume of the Lake(s) divided by the net freshwater input, is highly variable. The maximum flow in the eastern rivers during the event of June 1998 (168,000 ML d^{-1}) would have been capable of flushing the combined volumes of L. Victoria and L. King (890,000 ML) in 5 days. The total amount of water discharged during the event (440,000 ML) was about half that required to replace the volume of these two Lakes.

The two flow events in the western rivers in late October and early November 1995 had volumes of 320,000 ML and 440,000 ML, comparable to the volume of L. Wellington (380,000 ML). During 1997, the total inflow from the western and eastern rivers was 890,000 ML, the input from precipitation over all the Lakes' surface was 120,000 ML and the evaporative loss was 450,000 ML. The net freshwater input during the year of 560,000 ML was able to replace less than half the 1,300,000 ML volume of the combined Lakes Wellington, Victoria, and King. For the period between 1975 and 1999, the average flushing times due to freshwater through-flow are calculated to be 98 days for L. Wellington and 206 days for the whole of the Gippsland Lakes. However, flushing due to through-flow occurs only sporadically, and between such events the flushing times might be very long.

Water level changes in Bass Strait at Lakes Entrance cause water to flow in and out of the Lakes and constitute a second potential mechanism for flushing the Lakes. Analyses using the hydrodynamic model suggest that the tides are not an important mechanism for flushing the Lakes although they appear to play a role in maintaining a well-mixed water column in the narrow channels leading to the Entrance. For a tidal range of 5 cm the volume of water exchanged with L. Victoria and L. King is 9,000 ML. This is much less than the 45,000 ML volume of Reeve Channel (Bek and Bruton, 1979) which connects the Entrance to the main body of L. King. Thus, water flowing through the Entrance penetrates only a fraction of the length of Reeve Channel on the flooding tide before the tide changes.

Exchange associated with low-frequency sea level fluctuations appears to be much more effective than tidal exchange for flushing the Lakes. Because the periods of these fluctuations are much longer than the tides (>7 days versus 12 hours), the water level fluctuations within L. Victoria and L. King are much better able to follow those in Bass Strait. The exchange volume associated with a typical water level change within the Lakes of 30 cm is 52,000 ML, which is similar to the volume of Reeve Channel. Consequently, when sea level rises, salty oceanic water is able to penetrate the length of

Reeve Channel. Being generally more dense than the water inside the Lakes, some of the Bass Strait water is able to sink to the deeper parts of Reeve Channel and propagate as a density flow into the deeper parts of L. Victoria and L. King.

To examine the flushing behaviour of the Lakes, the hydrodynamic model simulated the dilution of a passive dissolved tracer. This analysis is not intended to be realistic, but rather it illustrates the way in which dissolved material within the Lakes is exchanged with the ocean. This was done by setting the initial concentration of a hypothetical tracer to 1.0 everywhere inside the Lakes, and zero in Bass Strait and in river inflows (and precipitation). The concentration of tracer then generally decreases with time over the course of the simulation, due to exchange with Bass Strait and river inflow. Figure 2.12 shows tracer concentrations at a MAFRI site 2 in southeastern L. King.
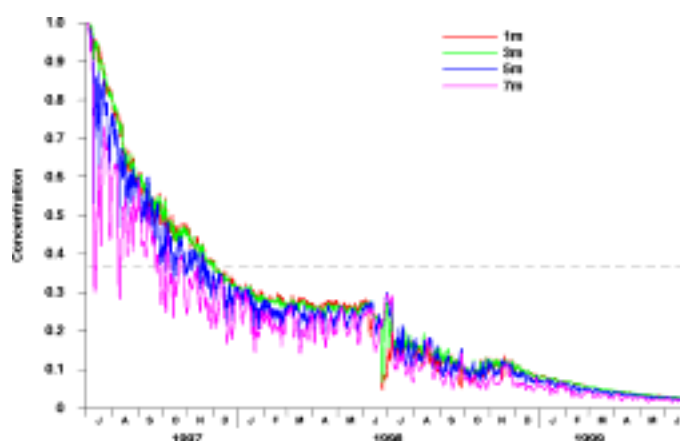
Figure 2.12 Simulated concentration of a passive dissolved tracer which was initially set to a value of 1.0 everywhere in the Lakes, at 1, 3, 5 and 7 m depth in L. King.

In the first year of the simulation shown, the tracer concentration decreases in an approximately exponential way, with a time constant of about 5 months. The calculated flushing time due to freshwater through-flow is 760 days for the last 6 months of 1997 so exchange with the ocean induced by low-frequency water-level fluctuations must be the main agent for flushing during this time. At 5 and 7 m depths, the time series of tracer concentration show short-lived spikes down to lower concentration values due to the incursion of salty Bass Strait water along the bottom of the Lake. Figure 42 of Walker and Andrewartha (2000) shows that these incursions are usually correlated with high water levels at the Entrance. After February 1998, the tracer concentrations level off, probably due to import of tracer from other parts of the Lakes' system and to evaporation, which would tend to concentrate the tracer. The flow event of June 1998 causes a sharp reduction in tracer concentrations, illustrating the impact of flow events on flushing.

3. Nutrient cycling and water quality in the Lakes

The GLES is intended to provide a quantitative understanding of the Lakes ecosystem function, and specifically the processes underlying environmental issues such as water quality and algal blooms. This objective is achieved through the development and application of an integrated process-based simulation model (the *biogeochemical model*) that provides insight into the current system state and function, and also a capability to predict the likely response of the Lakes to management actions.

From a management viewpoint, the Lakes as a system are subject to river flows and catchment loads of nutrients, sediments and organic matter, and to exchanges with Bass Strait. The environmental problems in the Lakes, as reflected in nutrient concentrations, algal (phytoplankton) blooms, light attenuation and oxygen depletion, are symptomatic of eutrophication: i.e., of loads of nutrients, organic matter and possibly sediments that exceed the natural system's capacity to assimilate them. The biogeochemical model provides a tool for understanding and predicting the response of the Lakes to flows and catchment loads, by representing the key processes controlling the fate and impacts of nutrients, sediments and organic matter within the Lakes.

The specific simulation model used here is based on the successful model developed for the Port Phillip Bay Environmental Study (Harris et al., 1996; Murray and Parslow, 1997). However, that model has evolved considerably in the course of this study (and other recent coastal studies). Detailed technical descriptions can be found in Murray and Parslow (1997), Parslow et al. (2001a) and Baird et al. (2001). Note that the biogeochemical model can be thought of as the mathematical expression of an underlying *conceptual model* of biogeochemical cycling in coastal and estuarine systems. We start by describing the structural components and processes that make up this underlying conceptual model.

The biogeochemical model of coastal system structure and function

The biogeochemical model explicitly considers the (coupled) cycles of inorganic sediments, nitrogen, phosphorus and carbon within the Lakes system. It considers processes within the water column, within the sediment, and at the sediment-water interface. As discussed below, benthic processes and sediment – water column exchanges play a critical role in shallow coastal systems such as the Gippsland Lakes.

The model represents fine (cohesive) inorganic sediments, and divides these into a slowly-sinking unflocculated fraction, and a rapidly sinking flocculated fraction. Unflocculated sediments are converted into flocculated sediments at a rate that increases with salinity over the salinity range 6 to 10. Suspended (unflocculated) sediments are delivered by rivers, and suspended sediment is subject to settling and sedimentation within the basins, and resuspension from bottom sediments. The proportion of fines within surface sediments is determined by sedimentation and resuspension fluxes, and a burial rate that removes fines from the surface sediment layer. Changes in the total amount of fine sediment (suspended plus bed sediment) occur as a balance between catchment loads, burial and export to Bass Strait. Fine sediments play two important roles in the biogeochemical model: suspended sediment concentrations strongly affect

light attenuation and therefore plant growth, and exchanges of inorganic phosphorus between dissolved and adsorbed particulate phases strongly affect phosphorus dynamics.

The cycles of nitrogen, phosphorus and carbon are strongly interrelated through the formation and breakdown of organic matter. Nitrogen and phosphorus are considered as the macronutrients most commonly limiting to primary production and algal blooms in estuarine systems (see later discussion on nitrogen vs phosphorus limitation in Gippsland Lakes). Biogenic silica can be limiting to diatom growth in coastal systems (e.g. at times in Port Phillip Bay, Harris et al., 1996), but concentrations are high and non-limiting in the Gippsland Lakes.

We deal first with the nitrogen cycle. The model describes the cycling of nitrogen through a number of dissolved inorganic, dissolved and particulate (non-living) organic, and living organic pools (Figure 3.1). In the water column, dissolved inorganic nitrogen (DIN = ammonia + nitrate) is taken up by phytoplankton, which can be consumed by zooplankton in turn. Ammonia is produced either by zooplankton excretion, or through microbial breakdown of the detrital organic matter produced by zooplankton feeding and mortality.
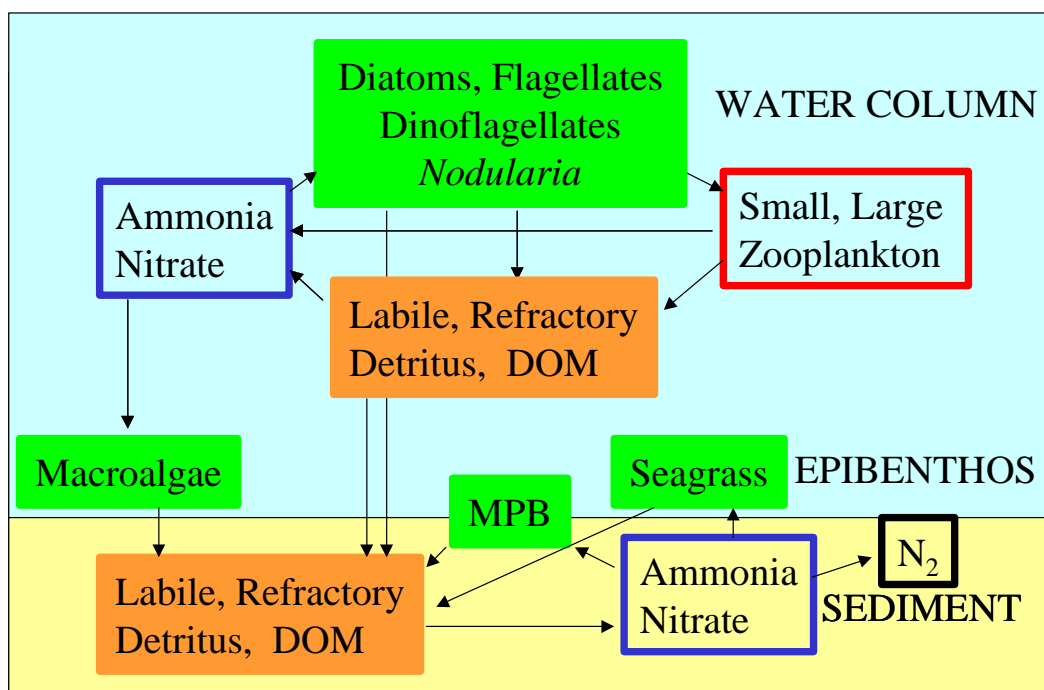
Figure 3.1 The conceptual view of nitrogen cycling in coastal ecosystems incorporated in the biogeochemical model.

Particulate organic matter settling to the sediment is also subject to microbial breakdown which releases ammonia into pore waters. Nitrifying and denitrifying bacteria in sediments can convert ammonia to nitrate, and nitrate to N_2 gas, which is then lost to the system. Nitrate and ammonia in pore waters can also diffuse back to the overlying water column.

The model represents three functional groups of benthic plants. Microphytobenthos (benthic diatoms) live within surficial sediments, taking up dissolved nutrients from pore water, and forming organic matter. Macroalgae are assumed to be attached to surface sediments, but take up nutrients from the water column, while seagrass take up nutrients from the sediment pore waters, but release oxygen into the overlying water column. Diatoms can also settle to the sediment, where they are subject to mortality, if not first resuspended.

Non-living organic nitrogen is divided into four functional components: labile detritus of planktonic and microphytobenthic origin, labile detritus originating from seagrass and macroalgae, refractory particulate organic nitrogen, and refractory dissolved organic nitrogen. The distinction between labile and refractory organic matter is important, as it is the "readily available" pool of nitrogen (dissolved inorganic plus labile organic) which most directly affects water quality, while the refractory dissolved pool is most likely to be exported from the Lakes.

With the exception of denitrification, the above processes all convert nitrogen from one form or functional component to another, but conserve total available nitrogen. Thus the total nitrogen pool within the system is determined by the catchment loads, export to Bass Strait, and denitrification. The balance between export and denitrification can play a major role in determining the sensitivity of coastal systems to nitrogen loads. In weakly flushed systems such as Port Phillip Bay, denitrification is the principal sink for nitrogen, and effectively controls system function and health (Harris et al. 1996).

The phosphorus cycle is almost identical to the nitrogen cycle shown in Figure 3.1, and described above, with a few key differences. Inorganic phosphorus is assumed to occur in two forms: dissolved inorganic phosphorus (DIP) and adsorbed inorganic phosphorus (PIP). The model assumes exchange between these two pools on a time scale of 1 day. This is comparatively rapid compared with the turnover times of DIP and PIP pools, so use of a shorter exchange time scale would have little impact on model predictions: i.e., the DIP and PIP pools tend to come to near-equilibrium. Adsorption of inorganic phosphorus to particles is complex, and in practice there are likely to be fractions with very different exchange time scales (Webster and Grace, 2001). However, these are poorly understood, and representing them is beyond the scope of this model. Fractions with very long exchange times could play a role in P storage and release from bed sediments (see later discussion). The balance between the dissolved and particulate phases is expressed as an adsorption isotherm which relates DIP to the mass ratio of PIP to the total suspended sediment (TSS). Inorganic phosphorus can be exchanged between water column and (benthic) sediment, either by diffusion and adsorption/desorption of DIP, or by sedimentation and resuspension of PIP. Within the sediment, the model allows a burial rate for PIP, which effectively removes or immobilises phosphorus.

Therefore, the total phosphorus pool in the system is controlled by a balance between catchment loads, export and burial. The adsorption-desorption exchanges, combined with sediment – water column exchanges, act to buffer water column DIP concentrations against fluctuating loads or phytoplankton uptake.

The carbon cycle is again almost identical to the nitrogen and phosphorus cycles. Organic carbon enters the Lakes, along with organic nitrogen and phosphorous, as dissolved and particulate organic matter. Dissolved inorganic carbon (DIC) is fixed by

plants, organic carbon is consumed and respired by zooplankton, and DIC is released by microbial breakdown of detrital organic matter. DIC is assumed not to be limiting in the system, as it is highly buffered in salt water, and potentially replenished by exchange with the atmosphere. The carbon cycle is represented primarily so that the associated oxygen cycle can be modelled. Oxygen is released through photosynthesis, and consumed through respiration, and of course subject to exchange across the air-water interface.

Dissolved oxygen is a key water quality indicator in its own right, but it also plays an important role in sediment biogeochemistry. Dissolved oxygen concentrations in the sediment pore water are determined by a balance between net respiration and diffusion from the overlying water column. Where the respiration demand exceeds supply, sediments become anoxic, and part of the respiration demand is met anaerobically. (Anaerobic respiration occurs predominantly through reduction of sulphate, rather than consumption of oxygen: it is important because it allows oxygen consumption and release of DIC and inorganic nutrients to be decoupled.) Depletion of sediment oxygen has two critical effects on sediment biogeochemistry. It inhibits or prevents nitrification, and therefore denitrification, leading to high release rates of ammonia into bottom waters. It also results in reduction of iron from the ferric to ferrous oxidation state in surface sediments, which greatly reduces the ability of sediments to adsorb phosphate, leading to release of DIP from sediments.

As explained above, denitrification and PIP burial represent key internal sinks for nitrogen and phosphorus respectively, and potentially control the total N and total P pools in the system. Thus, oxygen depletion in sediments has the potential to change the overall N and P budgets, as well as leading directly to large fluxes of ammonia and DIP into the water column.

The nitrogen, phosphorus, carbon and oxygen cycles are tightly coupled through the formation and breakdown of organic matter. The model assumes a fixed O:C:N:P stoichiometry of 138:106:16:1 (Redfield) for phytoplankton, zooplankton, microphytobenthos and labile planktonic detritus, and a stoichiometry of 716:550:30:1 (Atkinson and Smith, 1983) for macroalgae, seagrass, and labile benthic detritus. Refractory dissolved and particulate organic carbon, nitrogen and phosphorus are produced indirectly in the breakdown of labile detritus of either planktonic or benthic origin, and can therefore vary independently. The model assumes catchment loads of particulate and dissolved organic matter have macrophyte stoichiometry.

The formation of organic matter through primary production is a key process in the model. The growth rates of all primary producers depend on the availability of DIN, DIP and light, in a manner proposed by Baird and Emsley (1999). For microalgae, potential nutrient uptake rates are calculated assuming that diffusion to the cell surface is a rate-limiting step, while light absorption is computed based on the cell's absorption cross-section. These are then combined using a relationship between growth rate and internal cell quotas of N, P and C, which results in growth rates that are close to a "law of the minimum". A similar approach is used for macroalgae, except that nutrient supply is calculated using a physical model of solute transfer across a benthic boundary layer.

Note that plants can be either N or P limited, depending on concentrations of these nutrients. The balance between N and P limitation depends not only on N:P ratios in catchment loads, but also on processes controlling N and P cycling and distribution within the system, and especially on the key internal sinks: denitrification and PIP burial.

Light availability can play a critical role in controlling plant growth and uptake of nutrients and it is therefore important to capture the factors controlling light attenuation. The model represents background light attenuation by the water itself, plus contributions from phytoplankton pigments, particulate and dissolved organic matter, and inorganic suspended sediments. The model includes contributions from coloured dissolved organic matter (CDOM), consisting primarily of humics of terrestrial origin.

The Port Phillip Bay model represented two phytoplankton functional groups, small flagellates and diatoms, grazed on respectively by small and large zooplankton functional groups. Small flagellates have a higher affinity for nutrients, and out-compete diatoms under nutrient limitation, but flagellates are grazer controlled, so that phytoplankton blooms formed when nutrient concentrations are high tend to be dominated by diatoms.

In the Gippsland Lakes model, two further functional groups of phytoplankton, dinoflagellates and *Nodularia*, are added. Dinoflagellates are represented as slow-growing, with low affinity for nutrients, so that they would normally be out-competed by both flagellates and diatoms. However, dinoflagellates are allowed to vertically migrate, allowing them to access nutrients in dark bottom waters which are effectively inaccessible to the other phytoplankton groups.

Nodularia blooms are a particular issue in the Gippsland Lakes. *Nodularia* is a cyanobacterium, and has the ability to fix nitrogen. Otherwise, it is slow-growing and a poor competitor for nutrients, so is most likely to bloom when DIP concentrations are high, and the system is strongly N-limited. Studies in other systems, especially the Peel-Harvey in Western Australia, suggest that *Nodularia* has strong temperature and salinity requirements for growth (Lukatelich and McComb, 1986). In the model, *Nodularia* growth rates decline sharply at temperatures below 19^0C, and salinities above 25. These criteria are based on studies in Peel-Harvey Inlets, with limited tuning to reproduce the timing of *Nodularia* blooms in Gippsland Lakes. There are a number of other potential factors affecting *Nodularia* blooms which are not captured in the model, some of which are discussed later under "Nitrogen cycling and phytoplankton blooms". We emphasize that the understanding of *Nodularia* autecology, and its representation in the model, is quite limited, and later identify this as a key gap in knowledge and a subject for further study.

Physical forcing and catchment loads

The conceptual basis for the biogeochemical model described above is generic, in the sense that the processes included in the model could be expected to occur in any shallow coastal estuary or embayment. Indeed, the model described here (or its immediate antecedent) has been used within the National Land and Water Resources Audit as the basis for the Simple Estuarine Response Model (SERM), a web-based generic model of diverse Australian estuaries (http://www.marine.csiro.au/serm/). The

particular biogeochemical state and function of the Gippsland Lakes depends on the way in which these processes respond to the physical forcing and catchment loads applied to the Lakes.

The physical forcing comprises the river flows, precipitation and evaporation, winds, tides and low frequency oscillations in Bass Strait, and their interaction with the bathymetry to control circulation, mixing and stratification within the Lakes. These were dealt with at length in Section 2. After consideration of the physical exchanges, and observed distributions of water properties, it was decided that the key processes affecting nutrient cycling, phytoplankton blooms and hypoxia in the Lakes could be represented in a box model which distinguishes the major basins, and coarsely resolves horizontal gradients within L. Victoria and L. King. The biogeochemical box model (Figure 3.2), has 8 horizontal compartments, comprising L. Wellington, western, central and eastern L. Victoria, northern and southern L. King, Jones Bay, and the Entrance / Reeves Channel.
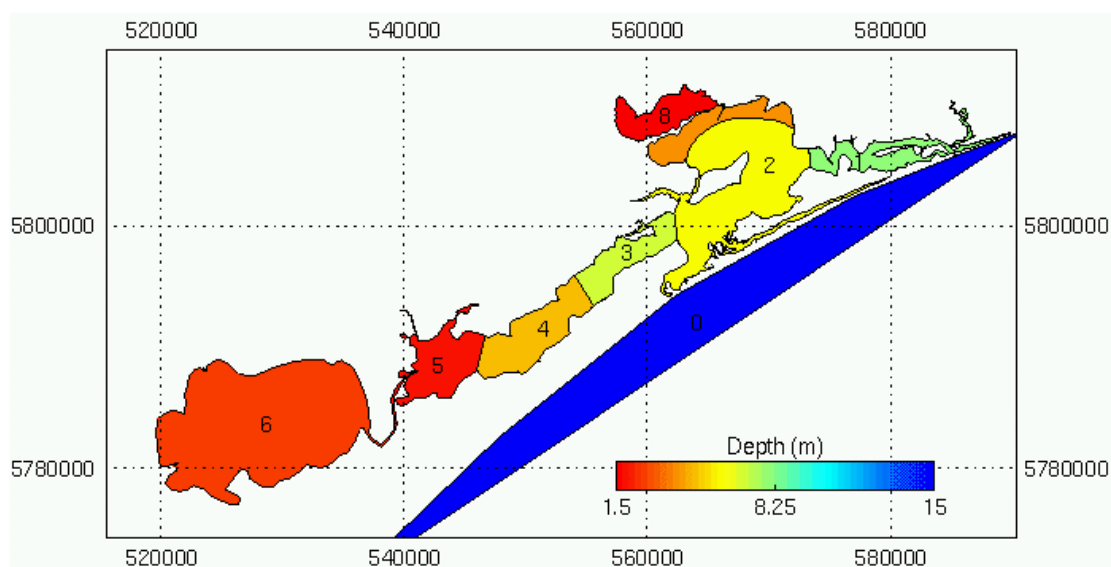
Figure 3.2 The box model structure used in the biogeochemical model of the Gippsland Lakes.

While L. Wellington and Jones Bay are shallow and generally well-mixed, L. Victoria and L. King are generally stratified. Stratification strongly affects nutrient cycling in the latter two basins, and the box model divides the water column in those basins into a surface and bottom layer.

In the results presented here, vertical and horizontal exchanges between the box model cells were calculated so as to best reproduce the temporal and spatial pattern in salinity and other conservative tracers predicted by the hydrodynamic model. The box model therefore closely approximates the basin-scale flushing characteristics described for the hydrodynamic model in the previous section.

Just as the flushing and physical structure of the Lakes are strongly influenced by the pattern of river discharge from western and eastern catchments, so the biogeochemical state and function are controlled by flows and loads of sediments and nutrients. Time

series of loads of total suspended solids (TSS), total nitrogen (TN) and total phosphorus (TP) were estimated using gauged river discharge, and concentration data acquired through water quality monitoring at downstream sites in the principal rivers. Statistical relationships between concentration and flow were developed for each catchment, and these were used to derive estimates of daily loads of TSS, TN and TP (Grayson et al., 2001a). These loads were augmented by estimates of loads from the Macalister Irrigation District (MID), and from non-gauged coastal catchments around L. Victoria and southern L. King. Grayson et al. (2001a) detail the methods used to estimate loads and a summary is provided in Appendix B.

In this study, attention is focused on the period July 1995 to June 1999, and in baseline scenarios and model calibration, the biogeochemical model has been forced by flows and loads for this period. The combined TN, TP and TSS loads from western catchments (La Trobe and Avon rivers and irrigation drains) into L. Wellington, and from eastern catchments (Mitchell, Nicholson and Tambo rivers) into Jones Bay / L. King, are shown in Figure 3.3.

The most striking feature of these loads is the large temporal variability, at event, seasonal and interannual scales. Similar variability was noted in flows, but this is exaggerated in the loads, because nutrient and sediment concentrations generally increase with flow. Because of the tremendous variation in daily loads (over 5 orders of magnitude in eastern catchments), we have plotted cumulative loads from July 1995 to June 1999, so that the event loads and background or base loads can be more easily compared.

Several patterns are apparent in these plots (and in the longer-term time series produced by Grayson et al. (2001a). Catchment flows and loads are generally high in winter and spring, and very low in summer and autumn. However, there is interannual variability imposed on this seasonal pattern. Within the study period, the winter-spring rains failed in 1997, resulting in an 18-month period of unusually low flows and loads.

Within wet seasons, a large fraction of the loads occurs in brief run-off or flood events, lasting only a few days. In the western catchments, the run-off events in October and November 1995 contributed TSS, TN and TP loads of 130,000, 1,400 and 160 tonnes respectively, almost equal to the average annual loads over the July 1995 to June 1999 period (Figure 3.3, Table 3.1). This event was less significant in the eastern catchments, where the June 1998 flood event dominated inputs, contributing TN and TP loads in excess of the annual average, and over twice the average annual load of TSS.
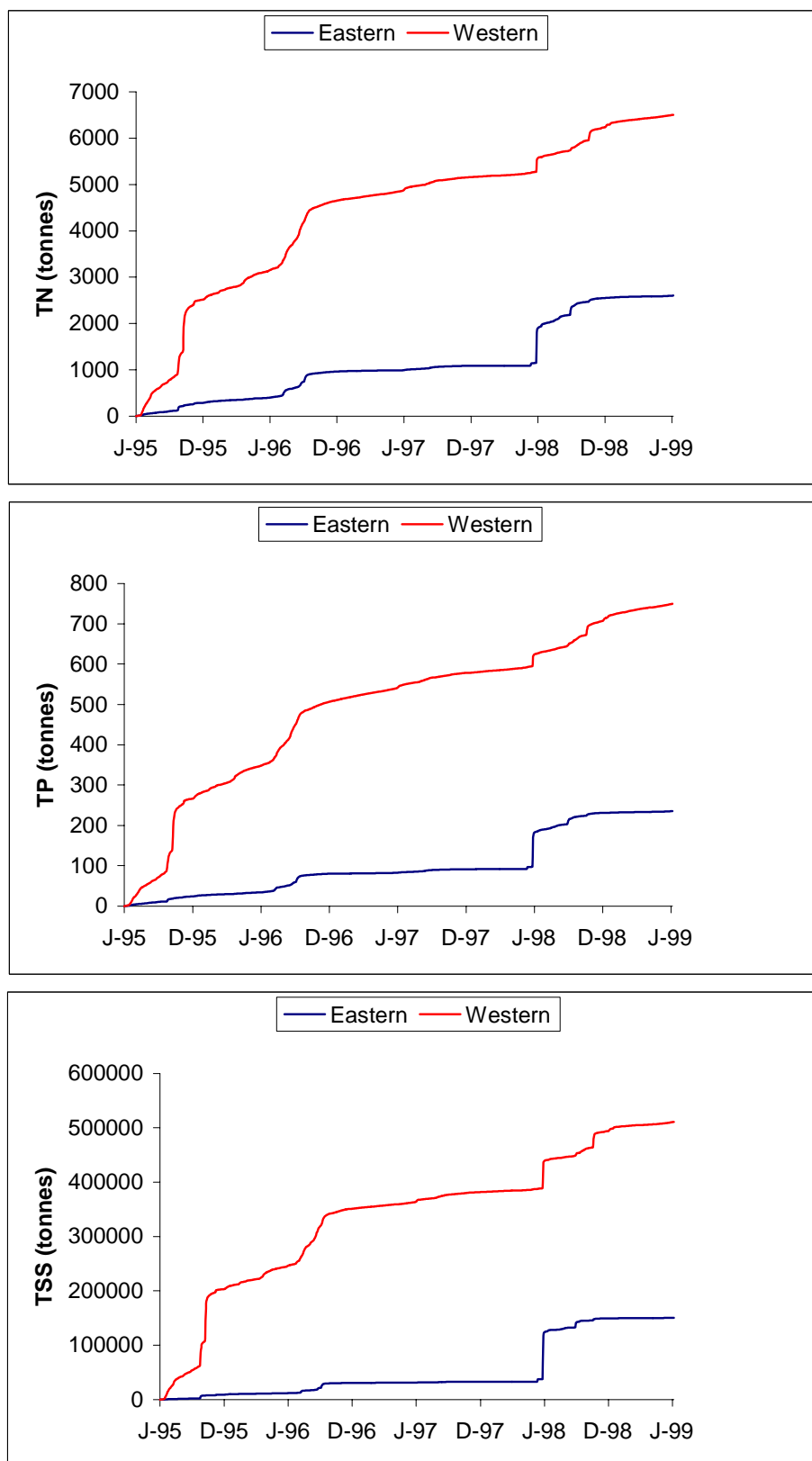
Figure 3.3 Cumulative TN, TP and TSS loads into L. Wellington (western) and L. King (eastern) from July 1995 to June 1999.

There are substantial uncertainties in estimates of daily loads, of order 20% under conditions of background flow, and up to 100% during flood events (Appendix B). These arise primarily from, and are based on, residual or unexplained variability in the statistical relationships between concentration and flow. Errors in estimates of average annual loads will be much lower, although given the large contributions from flood events, uncertainties in these loads will carry over into annual loads to some extent. In the calibration process, errors in loads will lead to errors in calibrated model parameters. Because the model is calibrated against time series of loads, and across all basins, one would again expect errors in daily loads in individual catchments to partly cancel out, but again errors in loads during large flood events could have a disproportionate effect. The implications of load uncertainty for management are discussed later.

	L. Wellington			L. King		
	TSS	TN	TP	TSS	TN	TP
1995-99 tonnes/year	128,000	1,600	190	38,000	650	60
Oct-Nov 1995 (tonnes)	130,000	1,400	160	5400	120	9
June 1998 (tonnes)	51,000	300	30	87,300	770	90

Table 3.1 Comparison of average annual and event TSS, TN and TP loads into L. Wellington (western catchments) and L. King (eastern catchments).

In order to understand and model the impact of these TN and TP loads on the Lakes, we need to determine their composition; i.e. to allocate the TN and TP across the various inorganic and organic fractions represented in the model. For the dissolved inorganic fractions, this was done by using log-log regressions of measured TN, TP, NO_x (nitrate + nitrite), NH_3 (ammonia) and DIP on discharge to determine the ratios NO_x/TN, NH_3/TN and DIP/TP as a function of discharge. However, we did not have direct information on the composition of the remaining fractions. The remaining TN was assumed to be organic N and split 50:50 between labile detrital N and dissolved organic N (DON). Adsorption isotherms were used along with DIP and TSS concentrations to estimate the amount of PIP present, and the remaining TP, not accounted for by labile detrital P, was assigned to DOP.

Substantial fractions of the TN and TP loads are readily available as nutrients for phytoplankton growth as DIN, DIP, desorbable PIP, and as labile forms that can be broken down in the Lakes. In the western catchments, DIN accounts for 676 tonnes/year out of the average annual TN load of 1600 tonnes/year, and labile detrital N accounts for a further 475 tonnes/year Similarly, DIP accounts for 41 tonnes/year and PIP for a further 92 tonnes/year out of an annual load of 190 tonnes/year. In the eastern catchments, DIN accounts for 130 tonnes/year, and labile detrital N a further 260 tonnes/year, out of 650 tonnes/year, while DIP accounts for 7 tonnes/year and PIP 23 tonnes/year out of 60 tonnes/year.

The allocation of the organic N and P load between labile and refractory, and between dissolved and particulate forms, is not based on data. This creates an uncertainty of about 30% in the load of readily available N and P. While labile particulate organic matter is retained in the Lakes and remineralized within weeks, contributing to phytoplankton biomass, more than half of the refractory dissolved organic matter load is likely to be exported. An increase in the refractory fraction would increase export

efficiencies and reduce nutrient cycling, primary production and oxygen depletion in the Lakes.

Nutrient and sediment concentrations in river discharge are high, and increase at times of flood events. At times of low flow, when the discharge is rapidly diluted by mixing into the receiving basins, it is the load rather than the concentration which determines impacts in the receiving waters at basin scale. However, during flood events, when the river discharge represents a substantial fraction of the receiving waters (about 50% in the June 1998 flood event), the concentrations of available nutrients in the discharge are important. For example, estimated NO_x and DIP concentrations in discharge into L. King during the June 1998 flood event were around 600 mg N m^{-3} and 40 mg P m^{-3} respectively.

The N:P ratios and the TP:TSS ratios in loads have important implications for biogeochemical function, as discussed below. The TN:TP ratio over the July 1995 to June 1999 period was 19:1 by moles in L. Wellington, and 24:1 by moles in L. King, compared with a Redfield ratio of 16:1 by moles for phytoplankton. The TP:TSS ratio was about 1.6 mg P / g TSS in L. King, and 1.5 mg P / g TSS in L. Wellington.

From the point of view of biogeochemical function and environmental health, there are several important characteristics of the loads into the Gippsland Lakes. First, the average loads of TSS, TN and TP are all large. Second, loads are concentrated into flood events, especially in the eastern catchments, and these events can dramatically change water quality in the Lakes. Third, a large fraction of the TN and TP loads consists of dissolved inorganic nutrient, or readily available nutrient.

Table 3.2 shows daily loads into L. Wellington from western catchments, into L. Victoria and L. King combined from eastern catchments, and into the combined Lakes from all catchments, expressed both per unit area and per unit volume of receiving waters. For interest, we compare these with loads into Port Phillip Bay during the 1991 to 1995 period.

	Load N/area	Load N/vol.	Load P/area	Load P/vol.
L. King + L. Victoria	9.77	2.00	0.86	0.18
L. Wellington	30.67	11.66	3.53	1.34
Gippsland Lakes	18.18	4.58	1.93	0.49
Port Phillip Bay	11.36	0.84	2.56	0.19

Table 3.2 Average daily loads of TN and TP into the Gippsland Lakes over the 1995-99 period, for eastern and western catchments and all catchments, expressed per unit area (mg m^{-2} d^{-1}) and per unit volume (mg m^{-3} d^{-1}) of receiving waters. These are compared with loads into Port Phillip Bay in the 1991-95 period of the Port Phillip Bay Environmental Study.

On a per area basis, 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. However, TN loads per unit volume into the 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.

Loads of nutrients from atmospheric deposition and precipitation were not estimated in this study. If atmospheric nitrogen loads per unit area into the Gippsland Lakes were similar to those in Port Phillip Bay, then the annual load would be about 180 tonnes/year, or less than 10% of the catchment load. Given the relative degree of urbanization in the catchments, and allowing for the industrial activity in the La Trobe Valley, this should probably be treated as an upper bound.

Aside from these loads, and the physical exchanges discussed in the previous section, the Lakes are also subject to seasonal cycles in temperature and solar irradiance. These cycles moderate biological rate processes, including photosynthesis, and may for example influence the timing of phytoplankton blooms.

Observed and modelled response of the Gippsland Lakes to catchment loads

We now consider the response of the Lakes to the temporal pattern of flows, loads and physical exchanges over the July 1995 to June 1999 period. This analysis is based on the function and output of the calibrated biogeochemical model, and directly on observations from the Lakes. We concentrate on the period from July 1997 to June 1999, because this was a period of intensive monitoring and measurement undertaken by MAFRI (Longmore et al., 2000b), and these data formed the primary basis for model calibration. Given the strong interannual and event-based variation in loads during this period, it provides excellent contrasts for calibration of model processes.

As we have just seen, the Lakes are subject to large perturbations and changes in inputs, on time scales from flood events of a few days to prolonged droughts over 18 months. It is generally not appropriate to treat the Lakes as being in steady-state: we must consider their dynamical response to this forcing.

Suspended solids and light attenuation

We consider first the behaviour of inorganic fine sediments in the Lakes, and the related phenomenon of light attenuation. Here, as for nutrients, L. Wellington behaves quite differently from the other Lakes. Lake Wellington receives very large annual loads of TSS, and although these are concentrated in flood events, there is substantial background or base load, especially from the La Trobe River. Lake Wellington is also shallow and well-mixed, so that there is continuing resuspension of fine sediment from the bottom. Finally, salinities in L. Wellington tend to be low, which reduces flocculation rates. These effects all combine to produce sustained high levels of suspended sediment ($\sim$20 g m^{-3}) in the water column in L. Wellington (Figure 3.4).

Unfortunately, observations of TSS are only available from EPA monitoring, roughly at 2-monthly intervals. One might expect TSS concentrations to vary on short time scales (days or less) as a result of wind fluctuations as well as run-off events, and the data cannot resolve these scales, although the scatter of the measurements does suggest a high variability. The model appears to over-predict water column TSS in L. Wellington by about 50% on average. The observations suggest a decline by about 50% in the drought period leading up to the June 1998 flood event and the model predicts a weaker decrease. The model predicts a short-lived peak in TSS at the time of the June 1998 flood event, which is not matched in the observations.

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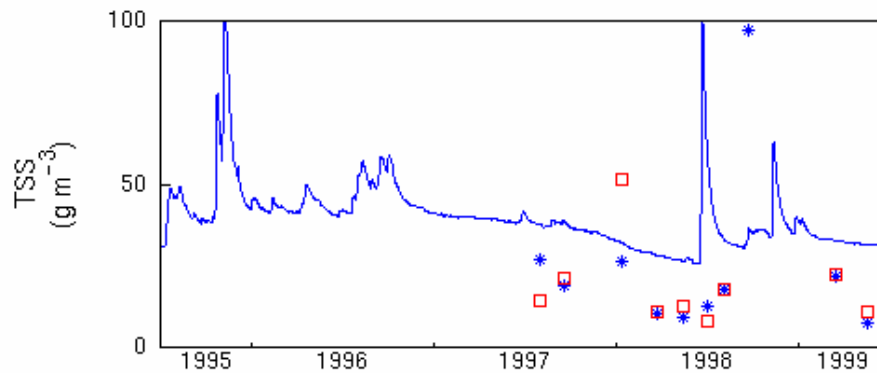
Figure 3.4 Predicted and observed TSS in L. Wellington for the period July 1995 to June 1999. EPA data from surface (asterisks) and bottom (squares).

Lakes Victoria and King receive lower direct loads of TSS, although the observations and model suggest about 60% of the TSS load into L. Wellington is exported into L. Victoria. However, these Lakes are deeper and usually stratified, so resuspension is likely to be less important, and to have reduced impact in surface waters. Finally, they generally have salinities above 10 even in surface waters, leading to maximum flocculation rates in the model, except immediately following flood events.

The model predicts much lower concentrations of TSS in L. Victoria and L. King than in L. Wellington. For example, in southern L. King, predicted surface concentrations are very low, around 1 g m^{-3} or less, and predicted bottom concentrations around 7 g m^{-3}, at the end of the drought (Figure 3.5). The model predicts a short-lived transient increase to around 100 g m^{-3} at the time of the flood event, but this settles rapidly out of surface waters. The concentration declines more slowly in bottom waters, because sinking is partly offset by resuspension.

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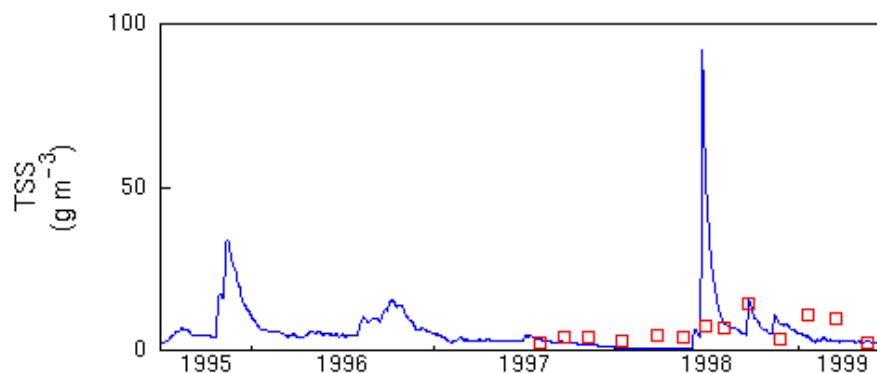
Figure 3.5 Predicted (line) and observed (squares) TSS in L. King (south) surface waters for the period July 1995 to June 1999.

The model captures the large differences in suspended sediment among the Lakes, although it over-predicts concentrations in L. Wellington, and tends to over-predict concentrations in surface waters, and under-predict concentrations in bottom waters, in L. Victoria and L. King. The GLES did not attempt to develop a fully-fledged model of sediment dynamics in the Gippsland Lakes: this would be better done at the spatial resolution of the hydrodynamic model, and would require much more knowledge of sediment size composition in river loads, critical shear stresses for resuspension, etc. These processes were only crudely parameterised in the biogeochemical model. The intent here was to reproduce the temporal and spatial variation in suspended sediments in the Lakes sufficiently to model light attenuation and P dynamics. The reproduction of light attenuation, as measured by comparison with Secchi depths, is discussed below. The errors in predicted TSS will affect the representation of P adsorption-desorption exchanges in the water column, but these errors are still small compared with uncertainty in the adsorption-desorption parameters, and likely to be effectively compensated by calibration of the adsorption coefficient (see later discussion of P dynamics).

Light attenuation is high in the Gippsland Lakes, with Secchi depths ranging from about 0.2 m to a maximum of around 1 m in L. Wellington (Figure 3.6) and from a minimum of around 1 m to a maximum of around 5 m in L. King south (Figure 3.7). High suspended sediment concentrations certainly contribute to light attenuation, especially in L. Wellington, but a statistical analysis of Secchi depths, TSS, chlorophyll and salinity in the long-term EPA data set suggested that coloured dissolved organic matter (CDOM) in catchment run-off also makes an important contribution, with attenuation coefficients due to CDOM in fresh water being ~ 2 m^{-1}. This conclusion is consistent with historical observations by Kirk (1983).

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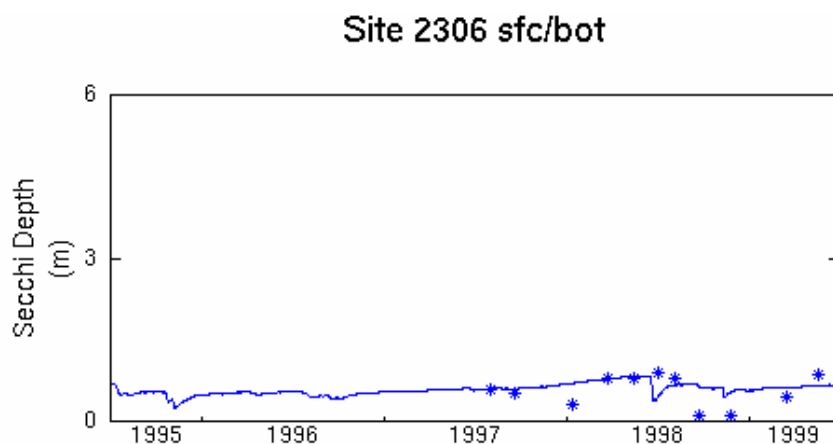
Figure 3.6 Predicted (line) and observed (asterisks) Secchi depth in L. Wellington for the period July 1995 to June 1999.
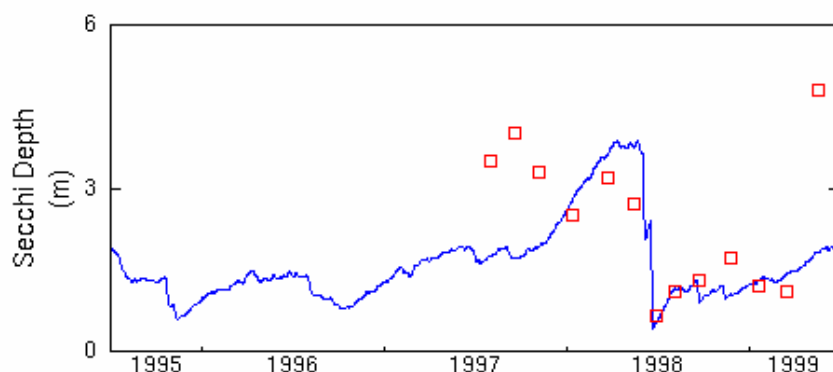
Figure 3.7 Predicted (line) and observed (squares) Secchi depth in L. King (south) for the period July 1995 to June 1999.

In L. Wellington, where salinities generally remain low and dilution of CDOM by marine water is limited, CDOM makes an important contribution to light attenuation year round, with suspended solids typically contributing around 1 m^{-1} to the attenuation coefficient. In southern L. King, there is substantial dilution of CDOM by mixing with marine water, and by the end of the drought, Secchi depths have increased to over 4 m. In the model, the increase in TSS associated with the June 1998 flood is very short-lived, and it is primarily CDOM, supported by increased chlorophyll from phytoplankton blooms, which drives Secchi depths in L. King to low values, around 1 m, in the months following the flood. Model predictions compare well with observations in the months preceding and following the June 1998 flood event, but the model appears to under-estimate Secchi depths in winter 1997 and May 1999 in L. King (Figure 3.7). This is not due to over-estimation of chlorophyll at these times (cf Figure 3.10), nor to over-estimation of TSS (cf Figure 3.5).

For the light attenuation coefficients typically observed and predicted in L. Wellington, the mean water column irradiance is 2 to 10% of surface irradiance, which could reduce but not prevent phytoplankton growth. However, bottom light levels at 3 m are less than 0.01% of surface irradiance, effectively preventing growth by benthic plants, except in shallow margins. In L. King, for the observed and predicted light attenuation in the months following the flood event, the 1% light penetration depth is around 2 m, and phytoplankton growth will be severely light limited in bottom layers (below 4 m), and growth by benthic plants impossible except in shallow margins. However, at the end of the drought period, the 1% light depth is around 7 to 8 m, and one might expect phytoplankton growth in bottom waters, and some microphytobenthic or macroalgal growth even in the main basins.

Nitrogen cycling and phytoplankton blooms

We now consider the response of the Lakes to nutrient loads, treating nitrogen first. Again, there are marked differences in the responses between L. Wellington and the other Lakes, due primarily to differences in depth and stratification.

Lake Wellington

The EPA observations show very low values of nitrate and ammonia, typically around 1 mg N m^{-3}, in L. Wellington waters throughout the July 1997 to June 1999 period. The model also predicts very low background ammonia and nitrate concentrations, with a temporary increase associated with the June 1998 flood event. In the model, DIN is kept at very low levels by phytoplankton uptake: phytoplankton biomass is always moderate (around 5 to 10 mg Chl a m^{-3}), and demand for DIN is high. The model predicts a phytoplankton bloom of diatoms to around 50 mg Chl a m^{-3} following the June 1998 flood event, but this bloom decays rapidly (Figure 3.8). With the exception of a smaller bloom associated with a later (November) run-off event, chlorophyll levels remain around 10 mg Chl a m^{-3} or less, decreasing slowly in the dry periods preceding June 1998, and in the summer and autumn of 1999.

The decay of the modelled June 1998 phytoplankton bloom, on time scales of a few weeks following exhaustion of DIN (Figure 3.8), is due to zooplankton grazing, and sinking out of diatoms. The predicted biomass of large zooplankton is relatively low at the end of the preceding drought, and increases fairly slowly in response to the phytoplankton bloom, so that zooplankton have little impact on the peak chlorophyll levels.
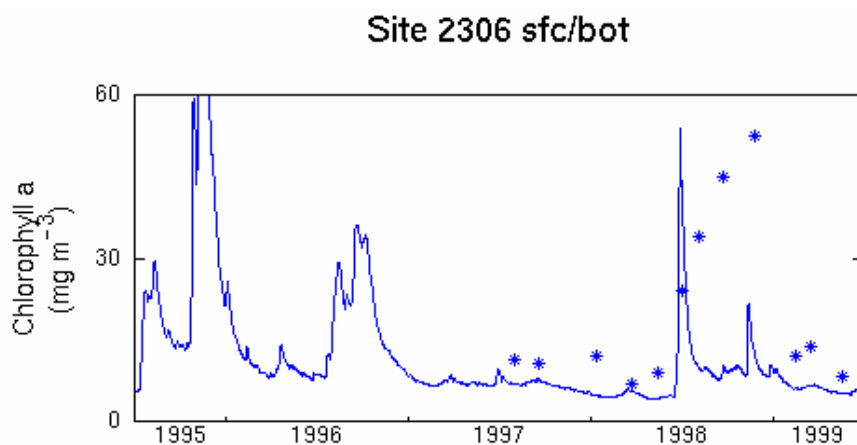
Figure 3.8 Predicted (line) and observed (asterisks) chlorophyll a in L. Wellington for the period July 1995 to June 1999.

The model predicts extended periods of high phytoplankton biomass (20 to 30 mg Chl a m^{-3}) in the winter/spring of 1995 and 1996, due to the large and prolonged run-off and loads in those years (Figure 3.8). The model also predicts a period of high ammonia concentrations in L. Wellington following the large flood event in November 1995: this is due to a shift in N:P balance within the system which is discussed later.

Associated with the decay of the bloom, there is a net transfer of organic nitrogen out of the water column to the sediment via sedimentation of diatoms and detritus. Once organic matter reaches the sediment, it breaks down, releasing ammonia. As this ammonia builds up in sediment pore waters, it is released back to the overlying water column via physical pore water exchange. Sediment efflux plus water column recycling contribute about 5 times the catchment DIN load (Table 3.3). Given that the catchment

load is concentrated in run-off events, recycling in water column and sediments is by far the dominant source of DIN during low flow periods. While water column recycling makes the larger contribution, and is important in supporting high levels of chlorophyll and primary production, the planktonic system would run down fairly quickly, on time scales of weeks, if not supported by the flux from sediments.

	L. Wellington	L. King + L. Victoria	All Lakes
Catchment DIN load	13	3	8
Recycling in water col.	41	59	49
Sediment efflux	19	21	20
Phytoplankton uptake	72	78	75

Table 3.3 DIN balance in water column in calibrated model of the Gippsland Lakes. All figures are mg N m^{-2} d^{-1}.

A substantial fraction of the pore water ammonia is nitrified and then denitrified, removing it from the system. In L. Wellington, the model predicts denitrification efficiencies of around 40 to 60%; i.e. on each pass through the sediment, about 40 to 60% of the organic nitrogen is lost. This leads to a fairly rapid depletion of the available nitrogen in the system.

Nitrogen is also lost through export to L. Victoria. However, except during periods of high run-off, exchange between L. Wellington and L. Victoria is very low. While a substantial fraction (about 50%) of the TN load into L. Wellington is exported (Table 3.4), much of this occurs during flood events, or in the form of refractory organic nitrogen.

	L. Wellington	L. King + L. Victoria	All Lakes
Catchment load	1600	650	2250
Denitrification	806	276	1082
Export	794	1168	1168
% total load denitrified	50%	19%	48%

Table 3.4 Fate of TN loads in calibrated model of the Gippsland Lakes. All figures are tonnes/year. Note that the total load into L. King and L. Victoria includes the 794 tonnes/year exported from L. Wellington.

The observations show an extended period of high chlorophyll biomass in L. Wellington through the winter and spring following the June 1998 flood, whereas predicted chlorophyll levels decay quickly after the initial bloom (Figure 3.8). It appears that the model either over-estimates phytoplankton loss rates, or over-estimates the removal of available nitrogen through denitrification, in this period. It is possible that continual resuspension of organic detritus and/or settled phytoplankton reduces the effective transfer of organic nitrogen from the water column to the sediment, and increases water column recycling efficiencies. Alternatively, the model may over-estimate denitrification efficiencies in the sediment in this period.

To sum up, L. Wellington behaves somewhat like a laboratory chemostat, in the sense that inputs of DIN are rapidly assimilated by phytoplankton, and DIN concentrations in the water column are generally very low. However, the catchment loads of DIN vary

strongly over time, and during summer and drought periods with low base loads, phytoplankton growth is primarily supported by release of DIN from sediments. Lake Wellington is poorly flushed during periods of low flow, and denitrification is predicted to be an important internal sink for nitrogen, controlling the time scales of removal of bio-available N from the system following large run-off events. High nitrification and denitrification efficiencies are maintained in the (model) sediments, despite high organic matter loads, because the well-mixed water column remains well-oxygenated. A schematic view of these dominant processes in L. Wellington is shown in Figure 3.9.
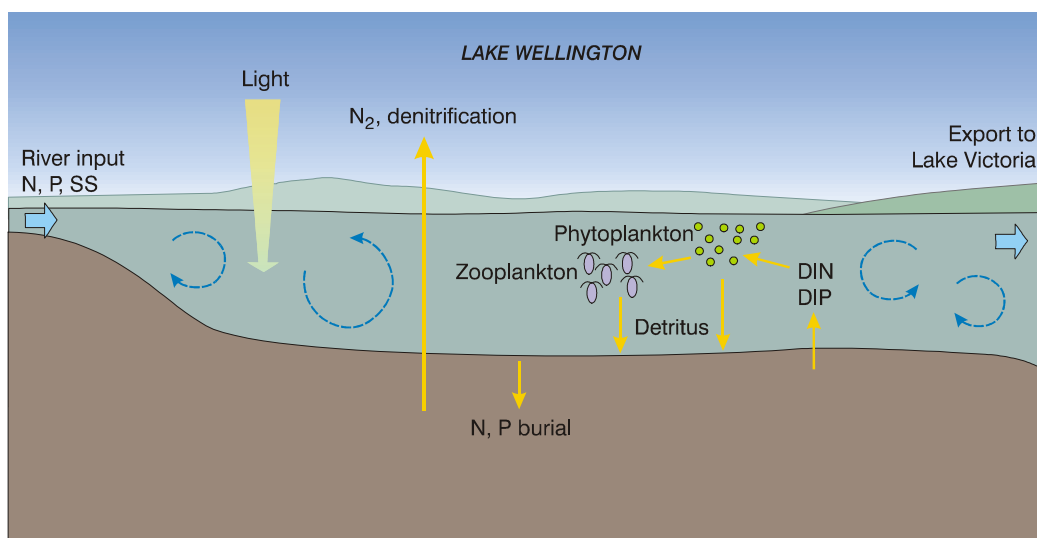
Figure 3.9 Schematic view of the current state and function of L. Wellington.

Lake Victoria and Lake King

Predicted and observed ammonia and nitrate concentrations are also very low (around 1 mg N m^{-3}) in surface waters in L. Victoria and L. King throughout the July 1997 to June 1999 period, except for a brief period around the June 1998 flood. In southern L. King, phytoplankton biomass declines to quite low levels, around 1 mg Chl a m^{-3}, by the end of the drought period preceding the June flood (Figure 3.10). The flood replaces much of the surface water throughout L. Victoria and L. King, increasing nitrate concentrations to high levels, and stimulating a phytoplankton bloom which reaches over 30 mg Chl a m^{-3}. This initial bloom response is triggered directly by catchment loads of DIN.

In the months after the flood event, L. Victoria and L. King remain vertically stratified, and this strongly affects the cycling of nitrogen. The sedimenting phytoplankton and organic detritus sink out of the surface layer to the bottom layer and sediment surface. Breakdown of this detritus results in very high oxygen consumption in bottom water and surficial sediments, which exceeds oxygen resupply via mixing across the halocline, leading to rapid oxygen depletion in bottom waters (Figure 3.11). Bottom waters are predicted and observed to become hypoxic or even anoxic in the months after the bloom. In the model, this leads to anoxia in surface sediments, and the shutdown of nitrification, resulting in high rates of ammonia release into bottom waters. Bottom

water ammonia is predicted and observed to build up to high concentrations, around 200 to 400 mg N m^{-3}, in L. Victoria and southern L. King (Figure 3.12).
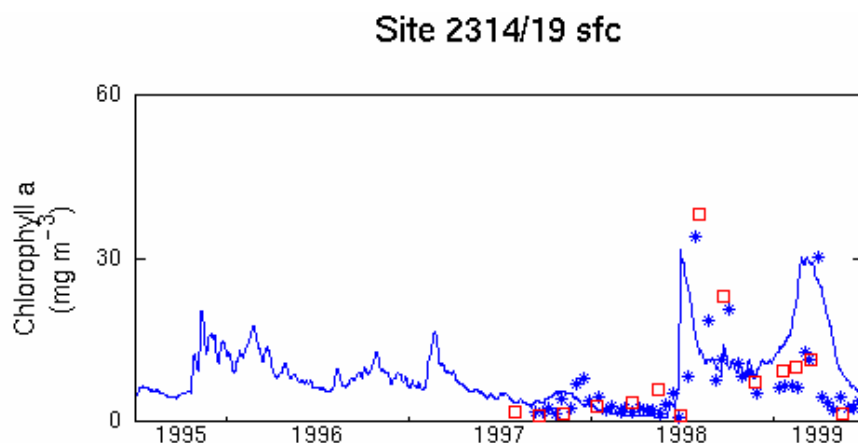
Figure 3.10 Predicted (line) and observed (asterisks, squares) chlorophyll a in L. King (south) surface waters for the period July 1995 to June 1999.
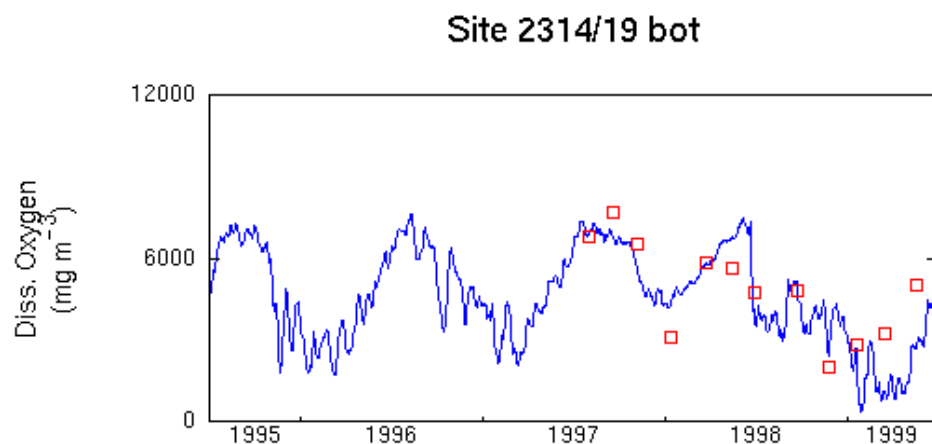
Figure 3.11 Predicted (line) and observed (asterisks, squares) dissolved oxygen in L. King (south) bottom waters for the period July 1995 to June 1999.

The model predicts that recycling of nitrogen in the water column and sediment greatly exceeds the direct catchment load of DIN into L. Victoria and L. King. The predicted mean sediment efflux of 21 mg N m^{-2} d^{-1} is only about half the mean of the observed sediment DIN fluxes measured at 3 sites in L. Victoria and L. King at roughly 3 monthly intervals during the 1997-99 MAFRI program (Longmore, 2000b). The predicted mean denitrification efficiency is less than 2/3 that observed.

Site 2314/19 bot
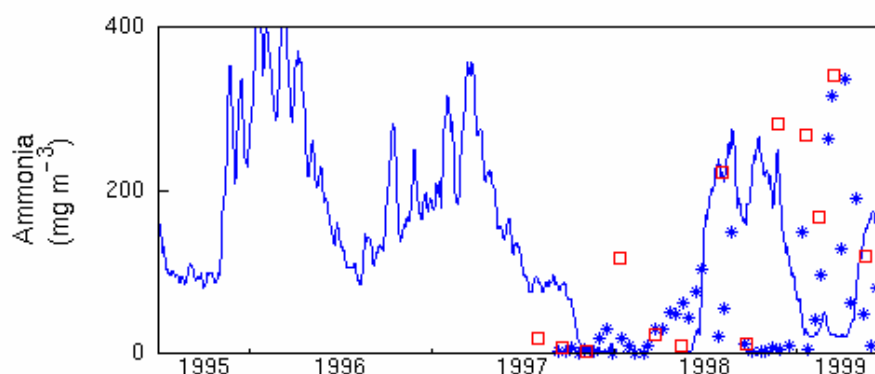
Figure 3.12 Predicted (line) and observed (asterisks, squares) ammonia in L. King (south) bottom waters for the period July 1995 to June 1999.

It's not clear whether these discrepancies are due to temporal or spatial aliasing in the observations, or to a mismatch in basin-wide fluxes between model and reality. It can be difficult to scale up fluxes at a small number of benthic sites to basin-wide fluxes, and quarterly sampling may not capture transients in fluxes associated with key events. The low predicted denitrification efficiencies in L. Victoria and L. King are due to bottom water hypoxia, and are based on the entire 1995-99 period, which includes extended periods of bottom water hypoxia in 1995-97. Mean denitrification efficiencies in the 1997-99 period are higher.

If the differences between predicted and measured fluxes are not due to problems of scaling up, they imply that the model substantially underestimates both sediment respiration and denitrification efficiencies. This would in turn imply that the model badly underestimates losses of nitrogen to denitrification in L. Victoria and L. King. Increasing denitrification rates to this level would result in a rapid run-down of the available nitrogen in the Lakes after runoff events. It is difficult to see how this could be reconciled with the observed time series of chlorophyll and water column TN in L. Victoria and L. King.

The observed patterns of surface chlorophyll and bottom water nutrients between July 1998 and May 1999 appear to reflect interactions among several phytoplankton functional groups. After the decay of the initial diatom bloom, phytoplankton in surface waters are dependent for supply of DIN primarily on mixing across the halocline, but this flux is quite small. Nonetheless, the observations show moderate levels of phytoplankton biomass (chlorophyll) in surface waters, and depletion of bottom water ammonia in September-October, despite the fact that light intensities in bottom waters should be insufficient to sustain significant phytoplankton growth at this time (Figure 3.10 and Figure 3.12). It seems likely that the chlorophyll is due to a bloom of dinoflagellates, which have been observed to undertake diel vertical migrations, taking up nutrients at night in bottom waters, and photosynthesizing in surface waters, under similar conditions in the Huon Estuary in Tasmania (CSIRO Huon Study Team, 2000).

The model is unable to reproduce the timing of this dinoflagellate bloom, and predicts that it should occur later, in January and February of 1999, probably because the grazing interactions among large zooplankton, diatoms and dinoflagellates are misrepresented. As a result, the model also incorrectly predicts the timing for draw-down of bottom water ammonia (Figure 3.12). Given the model's failure to reproduce the timing of dinoflagellate blooms, low confidence must be attached to the model's ability to predict the response of dinoflagellates, as opposed to other phytoplankton functional groups, in later scenarios. That said, if dinoflagellates are favoured by strong vertical gradients in nutrients, as postulated, then scenarios which result in little or no accumulation of DIN in bottom waters would still be expected to reduce the frequency and intensity of dinoflagellate blooms.

A short-lived chlorophyll peak observed in March /April of 1999 in southern L. King (Figure 3.10) has been attributed to *Nodularia*. Chlorophyll may be a poor indicator of *Nodularia* biomass, as N:Chl a ratios in this species may depart widely from other phytoplankton groups. This cyanobacterium can fix nitrogen, and so is capable of blooming even when the availability of ammonia in surface waters is low. The model is able to approximately reproduce the observed timing and magnitude of the *Nodularia* bloom in 1999 by imposing temperature and salinity growth requirements which are similar to those observed for *Nodularia* in the Peel-Harvey system in Western Australia (Lukatelich and McComb, 1986). However, the model also tends to predict *Nodularia* blooms in L. Wellington, unless the resting stages (akinetes) are assumed not to survive or germinate there.

The factors controlling akinete germination and *Nodularia* growth and mortality in the Gippsland Lakes are still poorly understood. Environmental controls on akinete survival and germination are not represented explicitly in the model: instead, *Nodularia* is assumed to recruit in L. King in early summer as a result of akinete germination. Consequently, the timing and distribution of *Nodularia* blooms in the model is controlled solely by growth and loss processes. The fact that the model tends to predict *Nodularia* blooms in L. Wellington as a result of exchange with L. Victoria suggests that there are factors controlling growth and survival other than temperature, salinity, light and nutrients, which are not captured in the model. Given all these limitations, model scenario predictions concerning *Nodularia* blooms should be treated cautiously. The scenarios predict reductions in bloom intensity and duration either due to large reductions in P loads or to increases in salinity. These predictions are plausible, but additional process studies are needed before *Nodularia* dynamics can be modelled with confidence.

The model predicts prolonged periods of high surface chlorophyll, bottom water hypoxia or anoxia, and elevated bottom water ammonia, in L. Victoria and L. King in the spring / summer of 1995/96, 1996/97, and 1998/99, following the wet season loads of 1995, 1996 and 1998. In contrast, the drought year of 1997 results in low surface chlorophyll, low bottom ammonia, and high bottom oxygen (at least in L. King) (Figure 3.10, Figure 3.11, and Figure 3.12). This stems from several inter-related effects. The low freshwater input results in the weakening and transient disappearance of vertical stratification. The low nitrogen loads reduce organic matter production and deposition. With increased vertical mixing and reduced bottom respiration, oxygen in bottom water is not depleted, and consequently nitrification (and denitrification) are maintained in surface sediments, increasing the system capacity to remove available nitrogen. This

positive feedback, in which reductions in loads lead to increased bottom oxygen concentrations, increased denitrification efficiency, and further reductions in organic matter production, plays a critical role in the response of the Lakes to load scenarios, discussed later.

Reduced loads of CDOM in the drought year also reduce light attenuation and allow phytoplankton growth in bottom waters. Thus, phytoplankton growth contributes to the predicted reduction in ammonia and increase in oxygen in bottom waters in the spring/summer of 1997.

In summary then, we can distinguish several characteristic stages in the response of L. Victoria and L. King to nitrogen loads. These stages are shown schematically in Figure 3.13, Figure 3.14, and Figure 3.15. Run-off events during "normal" wet seasons inject high concentrations of nitrate into surface waters, triggering phytoplankton blooms (Figure 3.13). In the stratified conditions which prevail through the following spring and summer, sedimentation of organic matter is followed by high oxygen demand and hypoxia or anoxia in bottom waters. Sediment anoxia leads to the shut down of nitrification and the release of ammonia into bottom waters, where it accumulates to high concentrations (Figure 3.14). Strong vertical gradients in nutrient and light can foster dinoflagellate blooms, and/or *Nodularia* blooms in late summer, once surface DIN becomes strongly limiting.

These stages are typical of eutrophic systems. However, after the drought year of 1997, stratification is weak or absent, light attenuation is low, bottom oxygen is high, chlorophyll and nutrient levels are low in both surface and bottom waters, and L. Victoria and L. King could be described as mesotrophic (Figure 3.15).
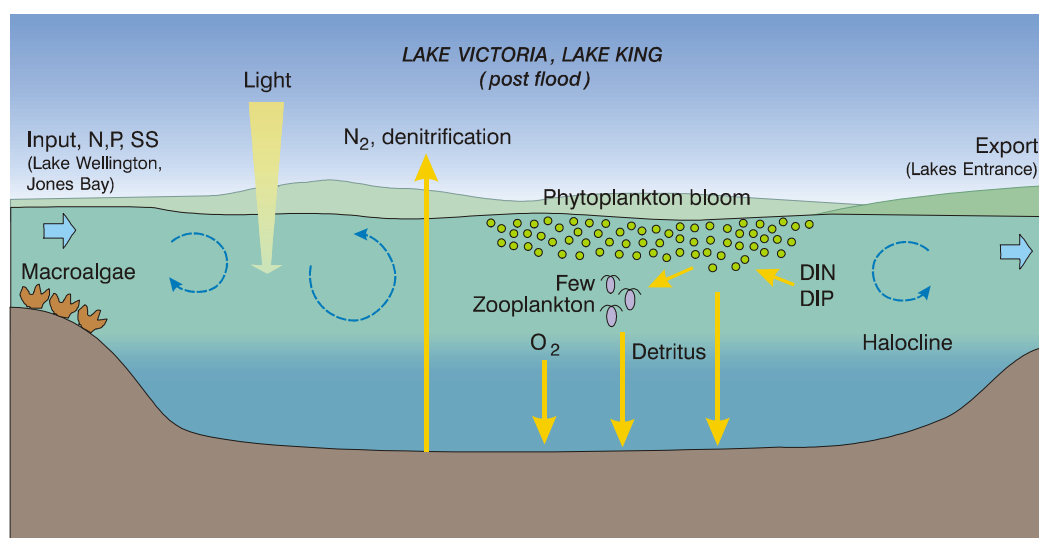
Figure 3.13 State and function of L. Victoria and L. King immediately post-flood.
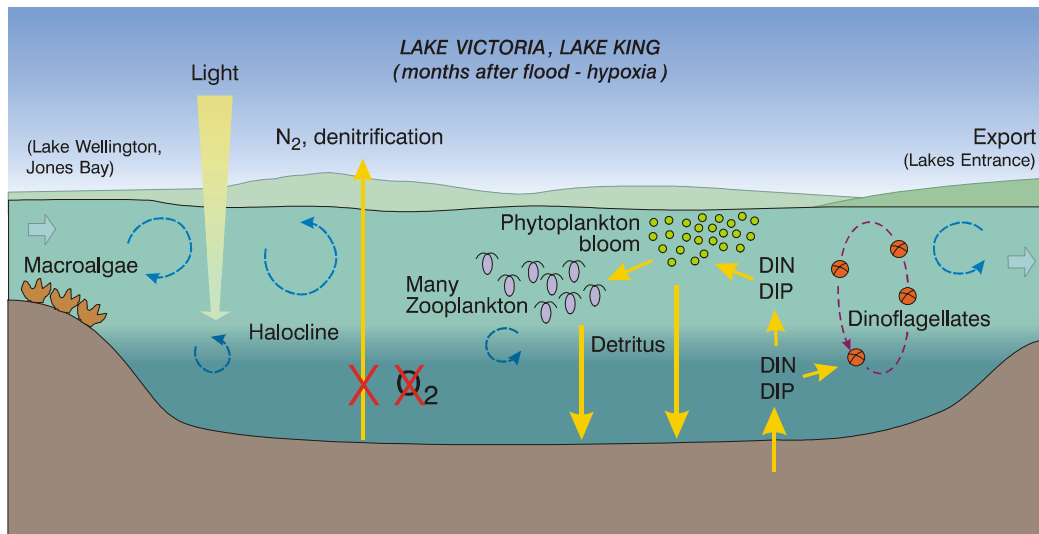
Figure 3.14 State and function of L. Victoria and L. King in the spring and summer following high run-off.
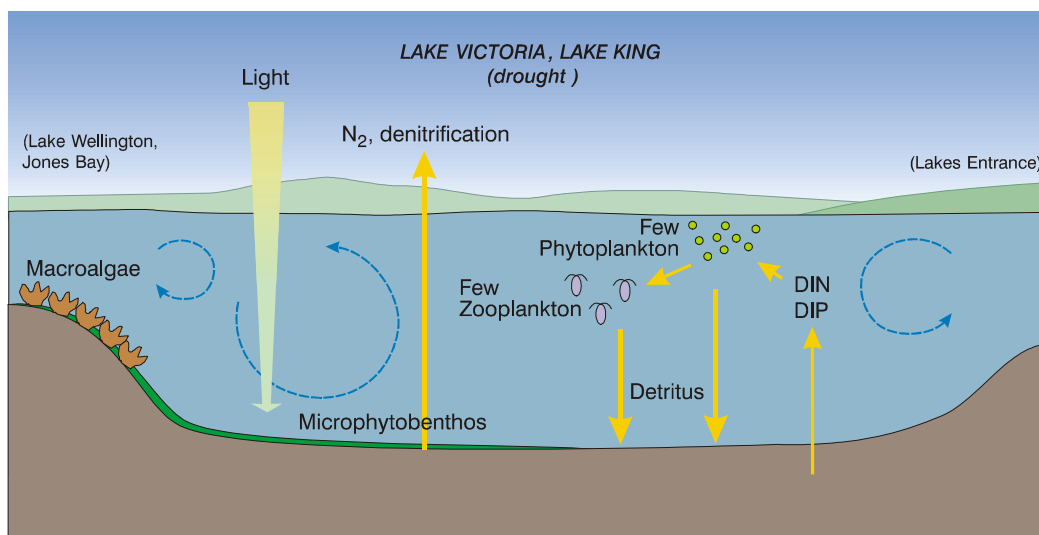
Figure 3.15 State and function of L. Victoria and L. King after a drought year.

The phosphorus cycle

The phosphorus cycle has been ignored in this discussion so far. This might seem inappropriate, as the TN:TP ratios in catchment loads exceed the Redfield ratio for phytoplankton especially in eastern catchments, so one might expect phosphorus rather than nitrogen to limit phytoplankton growth. However, the observations show surface DIN concentrations to be very low and limiting to phytoplankton growth almost all the time, while measured surface DIP concentrations are typically around 10 mg P m^{-3}, which would not be expected to limit phytoplankton growth.

The primary reason why N rather than P is limiting is that denitrification acts as a major internal sink for nitrogen, and this outweighs the internal sinks for phosphorus, discussed below. A secondary reason is that the adsorption-desorption exchanges of phosphate between adsorbed and particulate phases act to buffer DIP concentrations in the water column, in the face of fluctuations in supply and uptake.

While DIP is generally non-limiting, the calibration process for the model revealed some interesting features of the P cycle. It was necessary to introduce a small but non-zero term for P burial in sediments, especially in L. Wellington. Without this term, TP and DIP concentrations build up to levels where the TP load is all exported to L. Victoria, and this is inconsistent with observations. With this term, the model buries about 30% of the TP load to L. Wellington, and exports the rest to L. Victoria (Table 3.5).

Sediment cores collected in L. Wellington by Longmore (2000a), show downward fluxes of dissolved P below 10 cm depth. Webster and Wallace (2001) suggest that 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.

	L. Wellington	L. King + L. Vic	All Lakes
Catchment DIP load	0.8	0.2	0.5
Recycling in water col	5.7	8.2	6.8
Sediment efflux	3.4	4.4	3.8
Phytoplankton uptake	9.9	10.9	10.4

Table 3.5 DIP fluxes in water column in calibrated model of the Gippsland Lakes. All figures are mg P m^{-2} d^{-1}.

Following experiments using sediments from the Gippsland Lakes, it has been argued (Webster and Grace, 2001) that a Freundlich P-adsorption isotherm, in which the PIP mass concentration is related to a fractional power of the DIP concentration, should be used. However, when this isotherm is used, the predicted DIP concentration behaves approximately like the cube of the PIP mass concentration, and becomes very sensitive to changes in PIP in either the water column or sediments.

With the current model representation of P burial and the Freundlich adsorption isotherm, the balance between N and P limitation is sensitive to fluctuations in loads. In fact, during large run-off events in 1995 and 1998, the model switches briefly over into P-limitation, resulting in high surface ammonia. This is attributable to both the high N:P ratio in catchment loads, and possibly the transient shutdown of denitrification. It is not clear whether the N:P balance is similarly poised in the real system, but the long-term EPA data sets do include some observations of high surface ammonia concentrations in the Lakes.

In the model under current conditions, these brief periods of P-limitation appear to have little effect on the overall pattern of nutrient cycling, phytoplankton blooms and hypoxia in the Lakes. P-limitation would be expected to select against N-fixing cyanobacteria, but the predicted periods of P-limitation tend to occur in winter and spring, and don't

affect *Nodularia* blooms in late summer / autumn. The balance between N and P limitation is controlled primarily by the internal sinks of denitrification and P burial. Given the N:P ratio in catchment loads, if all internal sinks were shut down, the Lakes would be expected to be marginally P-limited. Any factors which act to decrease denitrification efficiencies and/or increase P-burial efficiencies will push the Lakes towards P-limitation. In the model, oxygen depletion is the principal factor, and it tends to decrease denitrification and P-burial simultaneously.

The development of anoxic sediments in L. Victoria and L. King in the months following run-off events leads to reduction of ferric iron in surface sediments and the release of adsorbed PIP into the overlying water column. This results in large increases in bottom water DIP (Figure 3.12 to Figure 3.16). High rates of water column recycling along with sediment release of DIP maintain excess DIP in the water column, and greatly outweigh direct catchment loads of DIP (Table 3.5). Due to release of DIP from anoxic sediments, only a small fraction (13%) of the TP load into L. Victoria and L. King is buried (Table 3.6), and this helps to maintain an N:P ratio biased towards N limitation in surface waters. By maintaining excess water column DIP, sediment release in the model also contributes to *Nodularia* blooms in late summer / autumn. The model predicts high bottom water DIP concentrations, and *Nodularia* blooms, in the summers of 1995/96, 1996/97, and 1998/99, but not in the drought summer of 1997/98 (Figure 3.16).

	L. Wellington	L. King + L. Vic	All Lakes
Catchment load	190	60	250
P burial	62	24	86
Export	128	164	164
% total load buried	33%	13%	34%

Table 3.6 Fate of TP loads in calibrated model of the Gippsland Lakes. All figures are tonnes P/year. Note that the total load into L. King and L. Victoria includes the 128 tonnes/year exported from L. Wellington.
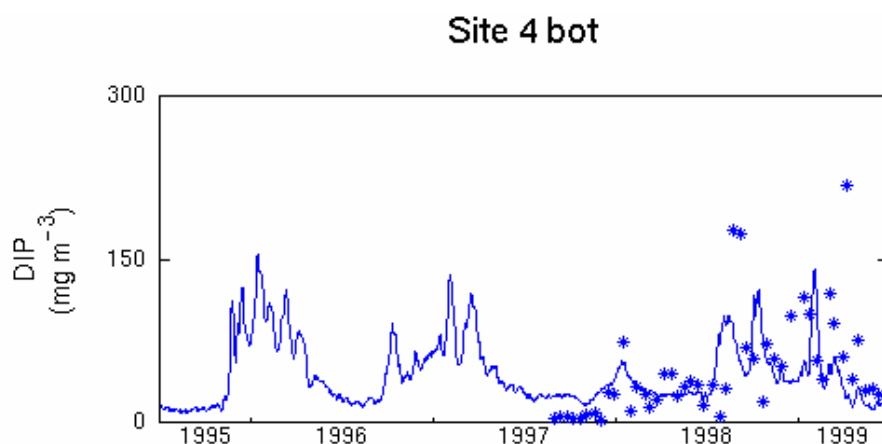
Figure 3.16 Predicted (line) and observed (asterisks) DIP in L. Victoria (east) bottom waters for the period July 1995 to June 1999.

Why the Gippsland Lakes are particularly susceptible to eutrophication

With this improved understanding of the N and P cycles in the Gippsland Lakes, we can now provide more insight into the environmental problems (poor water quality, algal blooms and bottom water hypoxia), which motivated this study. To put this explanation into perspective, recall that average daily loads of TN and TP into the Gippsland Lakes, expressed per area of receiving waters, are only respectively 1.6 and 0.75 times loads into Port Phillip Bay (Table 3.2). Yet water quality in Port Phillip Bay overall is reasonably good, and the system can be described as mesotrophic (Harris et al. 1996), whereas water quality in the Gippsland Lakes is poorer, and the system is characteristically eutrophic, with chlorophyll concentrations typically an order of magnitude higher. Why are the Gippsland Lakes more vulnerable to nutrient loads than Port Phillip Bay?

The Gippsland Lakes are poorly flushed, with typical flushing times (outside flood events) of 5 to 9 months. Long flushing times are important, and certainly contribute to the environmental problems in the Gippsland Lakes, but flushing times for Port Phillip Bay are even longer, about 12 months, so that flushing time alone certainly doesn't explain the greater susceptibility of the Gippsland Lakes.

Four factors contribute to the increased susceptibility of the Gippsland Lakes. First, the Lakes are shallow, so that comparable loads per unit area translate into much larger loads per unit volume. This is especially significant in the case of L. Wellington, where the TN load per unit volume is about 13 times higher than in Port Phillip Bay. Higher loads per unit volume translate almost directly into a chlorophyll concentration which is also on average about an order of magnitude higher.

Second, the loads into the Gippsland Lakes include a number of run-off events which are sufficiently large to flood the surface waters in the Lakes with high nutrient concentrations. These concentrations are sufficient to stimulate large algal blooms, with chlorophyll levels around 30 mg Chl a m^{-3} or higher. By comparison, the fresh water input into Port Phillip Bay is much smaller as a fraction of the Bay volume, and even during large flood events, the Yarra River plume tends to be confined to the eastern and northern edges of the Bay.

Third, vertical salinity stratification has a profound effect on the response of L. Victoria and L. King to catchment loads. These Lakes are deeper than L. Wellington, and TN loads per unit volume from eastern catchments are only about twice those into Port Phillip Bay (perhaps 3 times allowing for export from L. Wellington). However, nutrient loads in winter-spring wet seasons result in sufficient organic matter sedimentation to deplete oxygen in bottom waters. Bottom water hypoxia or anoxia may have undesirable direct consequences, such as mortality of benthic fauna, but it also results in shutdown of nitrification, and high rates of ammonia and phosphate release from sediments. By partially or wholly blocking the internal sinks of denitrification and P burial, bottom water hypoxia / anoxia increases and extends the impact of wet season loads throughout the following spring, summer and autumn, and increases the frequency and intensity of dinoflagellate and *Nodularia* blooms.

Finally, the light attenuation in the Lakes is high, due to both high levels of turbidity in L. Wellington, and high CDOM concentrations in river discharge into all Lakes. High light attenuation limits light penetration into the water column. Macrophyte production

is restricted to shallow margins and phytoplankton growth is inhibited in the bottom layers of L. Victoria and L. King. The lack of primary production in bottom waters is significant, as plant growth would help to supply oxygen and reduce nutrient accumulation there. Given the inferred importance of CDOM to light attenuation and biogeochemical function in the Lakes, consideration should be given to direct monitoring of CDOM concentrations (optical measures are simple and inexpensive), and to analysis of the factors controlling CDOM inputs from catchments.

Water quality trends in the Lakes

Harris et al. (1998) undertook an environmental audit of the Lakes in which they reviewed the water quality an ecological status of the Gippsland Lakes. It is clear that major changes have been wrought to the Lakes as a consequence of human activity. Prior to the permanent opening of the Entrance in 1889, the Lakes were a series of lagoons that were only periodically open to the sea. Marine influences at this time were small and the Lakes were populated by freshwater species of species of flora and fauna. After the Entrance was permanently opened, marine species began to invade. Harris et al. contend that the system has still not fully adjusted to the transition to an estuarine system.

Over the last 150 years, forests have been cleared and towns and sewers built which have altered the delivery of nutrients, sediments and freshwater to the Lakes and are certain to have caused changes in the Lakes' condition. Up until the late 1960s, L. Wellington was clear and dominated by aquatic macrophytes, but the drought of 1967/68 caused salinity levels to rise sufficiently that the macrophytes were killed. Since then, the Lake has been turbid and dominated by phytoplankton, a condition which is maintained by the introduction of carp and generally decreased freshwater inflows due to more frequent droughts since that time and to river regulation.

In L. Victoria and L. King, historical aerial photographs show the seagrass beds to have maximal extents in the late 1960s and late 1990s (Roob and Ball 1997). Seagrass require clear, low nutrient water so the extent of their beds could be considered to be an indicator of the condition of these two Lakes. Harris et al. (1998) suggest that the drought conditions of the late 1990s are responsible for the expansion of the beds at this time. Drought periods tend to deliver smaller loads of nutrients and suspended sediments to the Lakes than wet periods. Harris et al. (1998) contend that long-term changes in precipitation and run-off associated with long-term climate variation are likely to be major drivers for determining the condition of the Lakes.

As part of the GLES, we have undertaken a trend analysis on chlorophyll measurements collected by the EPA since 1986 at one site in L. Wellington and three sites in the other two Lakes (sites 2311, 2314, and 2316). The linear regressions show chlorophyll concentration increases in all three Lakes; for L. Wellington the rate is $0.29 \mu g \, L^{-1}$ per year and $0.08 \, \mu g \, L^{-1}$ per year in the other two Lakes (Figure 3.17). However, these trends are not statistically significant as the r^2 for the two fits are only 0.04 and 0.01, respectively. The analysis is confounded by the very large interannual variability evident in the data. Harris et al (1998) suggest that there is some upward trend in eutrophication in Lake Wellington based on the changes in phosphate levels measured over two years in the 1970s and in the measurements made since 1986. Harris et al. also say that for Lakes Victoria and King that "Analysis of EPA water quality data from the

Lakes shows no clear trend or marked deterioration in the lower Lakes since the mid 1980s", a conclusion consistent with our analysis. The natural variability in the data is such that the effects of any changes to catchment water quality in the late 1990s are not yet apparent.
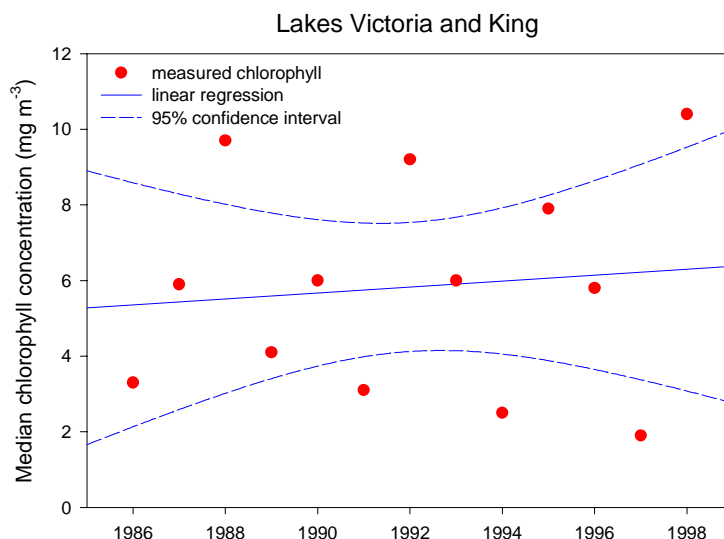
Figure 3.17 Yearly median chlorophyll concentrations averaged from three sites in L. Victoria and L. King. Also shown are the linear regression and its 95% confidence intervals.

Model limitations and uncertainty

How should we consider the model-predicted responses to changes in loads, freshwater discharges, and oceanic exchange given the imperfect agreement between model and observations? The model has been calibrated to match the observed cycling of N, P and C in the water column, and sediments, as closely as possible. Disagreement between simulations and observations can arise from a number of sources including limitations in the structure and formulation of the model, errors in model parameters and input data, and errors due to spatial and temporal aliasing in observations.

Structural limitations are inevitable in any ecosystem model, when a (relatively) small number of state variables must be used to represent a system which comprises a complex nested hierarchy of components and processes. The horizontal and vertical spatial resolution of the model also necessarily represents a trade-off between detailed resolution and the need to run many simulations for model analysis and calibration. Finally, among the processes which are included in the model, there are some for which we have only limited scientific knowledge and understanding on which to base the model formulation. Some of the key uncertainties in model formulation highlighted in this chapter include the autecology and interactions among phytoplankton functional groups (especially dinoflagellates and *Nodularia)*, and the processes controlling N and P sequestration in sediments, especially on time scales of years to decades.

Even where we are confident about the process formulation, there may be uncertainty about the parameter values. For some of the model processes, we have sufficient knowledge and experience from field studies and model applications in other systems to be confident about *a priori* estimates of model parameters. For others, we have relied on calibration of the model against observations. The calibration process carried out here has been qualitative or semi-quantitative. We do not yet have robust statistical methods for estimating parameters and posterior probability distributions for parameters in models of this complexity and computational cost. Consequently, our assessment of the likely errors and degree of confidence in model predictions is necessarily qualitative, and based on experience.

In general, we think that the model captures the key processes controlling the biogeochemical cycles of nutrients and organic matter in Gippsland Lakes, and their response to changes in catchment loads, on time scales of days to years. The calibrated model reproduces the spatial contrasts among basins, and the event, seasonal and interannual contrasts over time, in key biogeochemical indicators (nutrients, chlorophyll, dissolved oxygen, light attenuation). Indicator statistics are generally reproduced to within a factor of two, and this accuracy is often achieved for individual observations. In some cases, where larger discrepancies are observed, it is plausible to interpret this as a problem of spatial aliasing: i.e., of comparing predicted average values over model compartments or basins with individual spot samples. Cell counts obtained during the *Nodularia* bloom of autumn 1996 vary by more than a factor of 10 in samples collected on the same day at different locations in L. Victoria and L. King (Keith Thomas, pers. comm.) In other cases, e.g. dinoflagellates, it is clear that the model fails to reproduce the timing of events in the real system, although it does reproduce the magnitude.

We discuss the implications of model uncertainties for scenarios in subsequent sections. The scenarios generally involve extrapolation: i.e., pushing the model outside the range over which it has been calibrated. Under these circumstances, confidence in prediction depends heavily on our confidence in the formulation and parameterisation of the model processes. Confidence can be reduced either because the model fails to reproduce observations in the calibration period (as in the case of dinoflagellates), or because confidence in the underlying formulation is low, so that a good fit to observations may simply represent over-tuning (as in the case of *Nodularia*).

In general, it is our opinion that the model provides a sound basis for assessing the likely relative effects of changes in catchment loads and flows, or marine exchanges, on physical exchanges and nutrient and carbon cycles. The model does not address other ecological indicators or values such as higher trophic levels (fish) or fringing vegetation. However, prediction of the degree of eutrophication, as measured by indicators such as phytoplankton biomass and bottom water hypoxia, provides an essential platform for considering effects on other ecological indicators. Phytoplankton bloom composition is represented, in a crude way, within the model, but a high degree of uncertainty attaches to predictions about bloom composition, due to a combination of process uncertainty, and problems in calibration. The best that could be said of predictions about dinoflagellates and *Nodularia*, as a fraction of total biomass, is that they are plausible in qualitative terms. However, it should be noted that in absolute terms, their biomass is still subject to the mass balance restrictions on total phytoplankton biomass.

4.   Analysis of Model Scenarios

This section presents results and conclusions from a set of model scenarios designed to simulate the effects of a set of changes to loads, flows and exchanges liable to impact water quality and ecological function in the Lakes. A more detailed and complete description of the scenario analyses is available in the "Gippsland Lakes Environmental Study - Integrated Model Scenarios Report" by Parslow et al. (2001b). The scenarios were agreed with stakeholders and are designed to provide information about the likely response of the Lakes to three classes of changes, namely changes in the river inputs of nutrients and suspended solids, alteration to river flows, and engineering modifications for altering the exchange with the sea.

In formulating the scenarios, there has been no attempt to predict the effects of specific actions in the catchments, such as changes in land use or water allocation, on the temporal variation of loads and flows, or in the composition of loads. The formulation and simulation of specific realistic scenarios could well be the next step in development of a management strategy for the Lakes. (This would require improved models of catchment processes and dynamics.)

The simulated response to each scenario was obtained by applying the hydrodynamic and biogeochemical models using specified inputs of nutrient and suspended sediment loads and freshwater inflows. The measured (or estimated) loads and flows for the 4-year period July 1995 to June 1999, used in model calibration and analysis of the current condition, defined the *baseline scenario*. Most of the other scenarios involved modifications to the flows and loads in the baseline scenario. A list of scenarios tested is presented in Table 4.1.

The scenarios involving the engineering modifications used the flows and loads for the baseline scenario, but the bathymetry used by the hydrodynamic model was modified to achieve the specified marine exchanges. For the two long-term scenarios, the model was run for 32 years using eight 4-year input sections. In one of these scenarios, nutrient loads were increased by 1% per year; in the other, loads were unchanged. For these scenarios, the output was analysed for two 4-year periods, starting at years 14 and 28. Results are presented for the scenario with the long-term increase in loads only and are labelled LT14 and LT28.

In presenting results from the scenarios in this report, we pay particular attention to indicators that demonstrate environmental impact in the Lakes, namely phytoplankton blooms and hypoxia. Chlorophyll concentration is related to the presence of phytoplankton; high chlorophyll concentrations are an indicator of a phytoplankton bloom. *Nodulari*a blooms can be toxic so blooms of these phytoplankton types are particularly undesirable. Low dissolved oxygen concentrations are an indicator of degraded ecosystem function. Not only do they adversely impact aquatic fauna, they also indicate a system which is overloaded with nutrients and which has a reduced capacity for 'eliminating' nutrients through burial and denitrification.

We also refer briefly to results for bottom ammonia and phytoplankton primary production, as they provide useful insight into changes in system function in response to

catchment loads. Parslow et al. (2001b) present results for a larger suite of indicators including nutrient concentrations.

Code	Description
Baseline Scenario	
B	Standard calibration run, with July 1995-June 1999 forcing.
Load Scenarios	
L-20	20% reduction in all loads
L-50	50% reduction in all loads
L-70	70% reduction in all loads
LE-20	20% reduction in eastern catchment loads
LW-20	20% reduction in western catchment loads
LW-40	40% reduction in western catchment loads
LMP-40	40% reduction in P loads from MID
LMPN-40	40% reduction in N and P loads from MID
Long-term scenarios	
LT14	years 14 - 18 from 32-year run, with 1% per annum growth in loads
LT28	years 28 - 32 from 32-year run, with 1% per annum growth in loads
Flow Scenarios	
F-20	Decrease flow by 20% in all rivers
FE-20	Decrease flow by 20% in eastern rivers
FW+20	Increase flow by 20% in western rivers
Exchange Scenarios	
E-	Exchanges from hydrodynamic model with existing Entrance shallower
E+	Exchanges from hydrodynamic model with existing Entrance deeper
E2	Exchanges from hydrodynamic model with second entrance at Ocean Grange

Table 4.1 List of scenarios tested.

There is very strong temporal variation in flows, loads, and system response within the 4-year baseline period on event, seasonal and inter-annual time scales. Parslow et al. (2001b) chose to summarise the temporal variation by constructing frequency histograms for each indicator, within each basin, over the designated 4-year output period. Here, we focus on the 95%ile concentration for the phytoplankton in the surface layer of the Lakes as being the statistic indicative of the presence of a bloom. We consider the 5%ile for oxygen as representing the low-oxygen condition in the bottom waters of L. Victoria and L. King. The scenario results for the 95%iles of chlorophyll and *Nodularia* relative to baseline are presented in Figure 4.1 together with the 5%ile for oxygen concentrations.

The load reduction scenarios involve reducing loads from both catchments, from eastern catchments alone, from western catchments alone, from the Macalister Irrigation District (MID) and from both catchments. These allow us to assess the relative sensitivity of model indicators in the Lakes to loads from differing sources. The percentage reductions were applied uniformly to all rivers flowing to the Lakes and did not differentiate between peak and base flows.

Load scenarios

Reducing loads from all catchments

We focus initially on the response to changes in loads to all catchments. The relevant scenarios involve reductions by 20% (L-20), 50% (L-50), 70% (L-70), and LT14 and LT28, that correspond approximately to 16 and 30% increases above baseline loads.
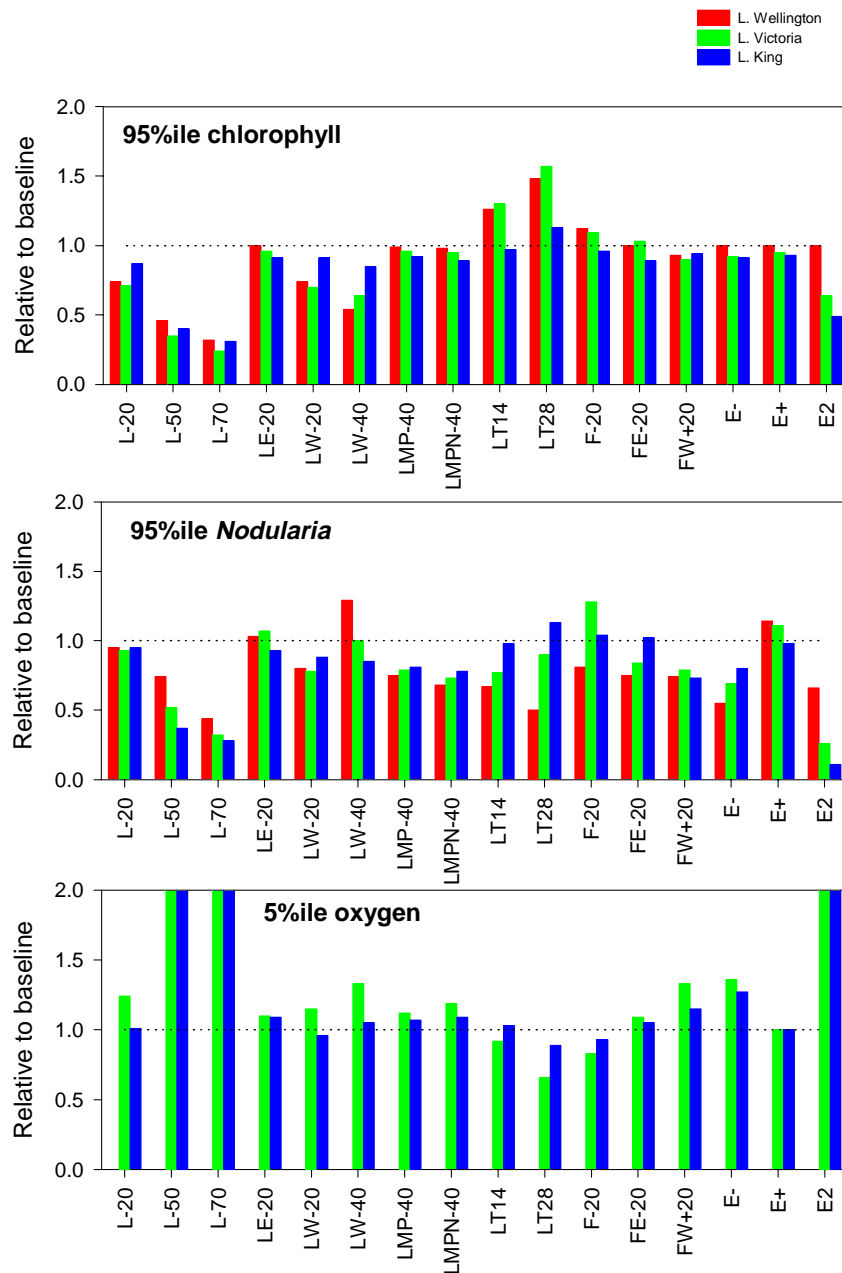
Figure 4.1 Results of scenario analysis relative to baseline for 95%iles chlorophyll, *Nodularia*, and for 5%ile oxygen.

For the uniform load reductions, it is instructive to compare results across scenarios and indicators by plotting indicators (relative to baseline) vs loads (relative to baseline). Phytoplankton biomass (chlorophyll concentration) in all three basins responds more or less linearly and proportionately to changes in catchment load (Figure 4.2), with some departures associated with bloom dynamics for individual functional groups (discussed below). Phytoplankton production in L. Wellington also responds linearly to loads, but phytoplankton production in L. Victoria and L. King shows a definite non-linear character, with an accelerated response to loads between 50 and 100% of current loads (Figure 4.3).
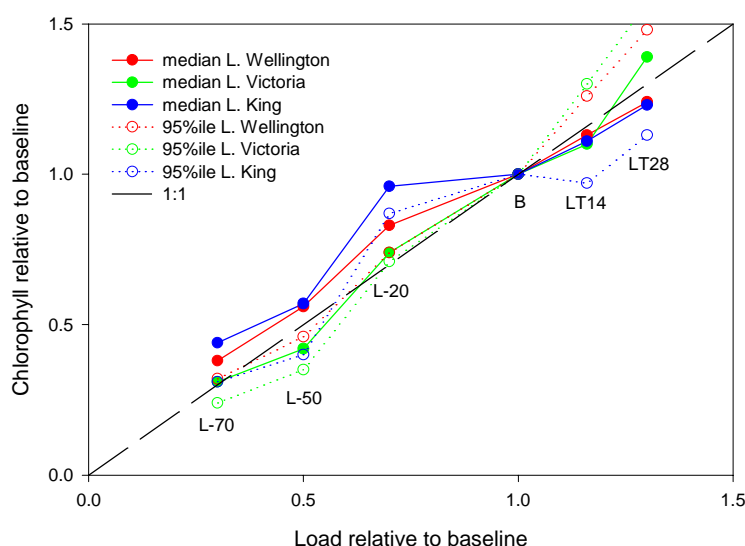
Figure 4.2 Median and 95%ile chlorophyll concentrations relative to baseline versus load relative to baseline.

This response in primary production is associated with the non-linear feedback involving bottom water hypoxia and nutrient release from sediments discussed in Section 3. This can be seen clearly in the responses of bottom water oxygen (Figure 4.4), and bottom water ammonia (Figure 4.5). Both show strongly non-linear responses to changes in loads, with a shift in response (recovery of bottom oxygen, reduction of ammonia concentrations) between 100% and 50% of baseline loads.

Even though overall phytoplankton biomass responds approximately linearly to changes in load, species composition alters in ways that reflect a complex set of biochemical and ecological interactions. With a 20% reduction in load, the 95%ile for *Nodularia* is only a little reduced from its baseline value, although at 50 and 70% load reductions, it decreases substantially (Figure 4.1). Because *Nodularia* is N-fixing it may receive a competitive advantage with reduced N loads compared to other species, but ultimately the reduction in P load would limit the growth of this cyanobacterium as well.
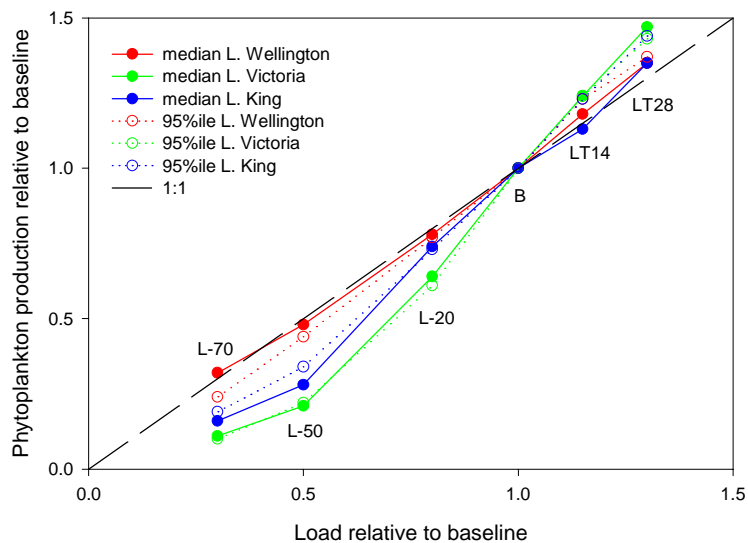
Figure 4.3 Median and 95%ile phytoplankton production relative to baseline versus loads relative to baseline.
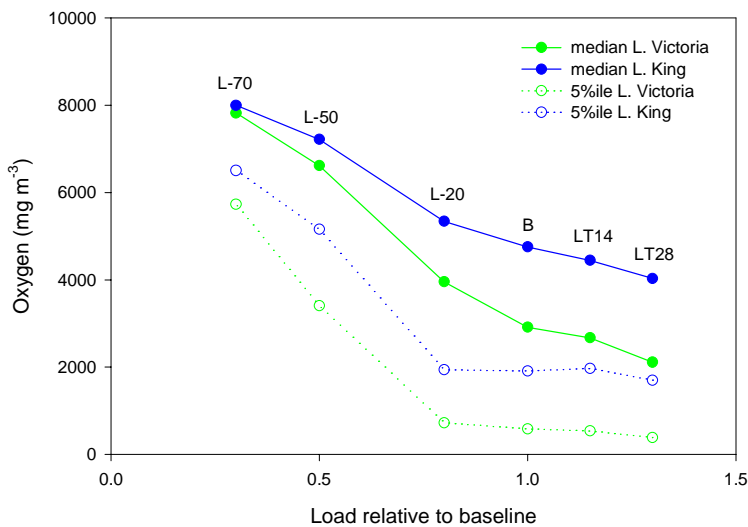
Figure 4.4 Bottom water oxygen concentrations in L. Victoria, L. King, versus loads relative to baseline.
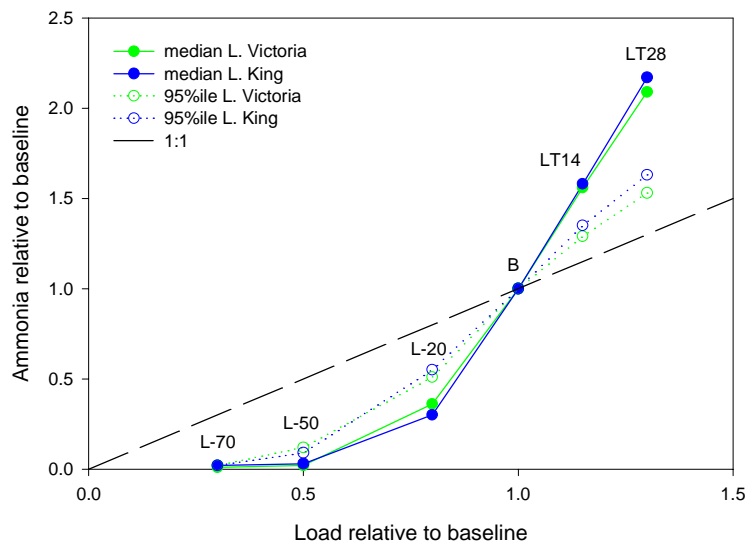
Figure 4.5 Bottom water ammonia in L. Victoria and L. King, relative to baseline concentrations, versus loads relative to baseline.

Figure 4.6 shows the time series of predicted chlorophyll concentrations in L. King for 0% (base scenario), 20%, 50%, and 70% reductions in sediment and nutrient loads to the Lakes. The peak concentrations of diatoms following the floods in June 1998 are reduced in approximate proportion to the reduction in input loads, but these concentrations are determined largely by the nutrients input by the flood and are little affected by other factors such as zooplankton grazing and biochemical cycling. By contrast, the chlorophyll occurring in autumn 1999 is reduced by considerably more than a factor of two when loads are reduced by half.

The effect of the 20% reduction in nutrient loads on the *Nodularia* blooms occurring in the autumns of 1996, 1997, and 1999 is quite different in the three years. In 1996, a 20% reduction in load has little impact on the size of the bloom; in 1999, the severity of the bloom is reduced by about 20%; but in 1997, the size and duration of the bloom is predicted to be substantially larger than for the baseline scenario.

For these scenarios, where TN and TP loads are subjected to the same percentage changes, the model predicts a tendency towards stronger N-limitation as loads are reduced. That is, the duration and magnitude of episodes of elevated surface ammonia decline as overall loads are reduced. This occurs because increases in denitrification efficiency at lower loads outweigh increases in the efficiency of P sequestration. However, given the uncertainties in the representation of P dynamics in the sediment, this prediction should be treated with caution.

A shift towards N-limitation might be expected to favour N-fixing cyanobacteria, and could partly explain the predicted increase in *Nodularia* bloom size in 1997 under a 20% load reduction. However, *Nodularia* bloom size is still limited by the available P,

and for large load reductions in N and P, decreases in the absolute concentrations of DIP are predicted to result in reductions in *Nodularia* bloom intensities.

The baseline 5%ile oxygen concentrations are about 500 and 2000 mg m^{-3} in L. Victoria and L. King respectively, low enough to seriously limit denitrification and lead to release of ammonia and phosphate from sediments, and to stress benthic fauna. The model predicts that very substantial reductions in loads, of order 50 to 70%, are required to eliminate episodes of bottom-water hypoxia, and the associated shutdown of denitrification and increased ammonia and phosphate release from sediments (Figure 4.4 and Figure 4.5). At these loads, the model indicates the Lakes could be classified as being in the upper mesotrophic range, with median chlorophyll levels around 5 mg Chl m^{-3}, and 95%ile (bloom) levels around 10 to 15 mg Chl m^{-3}. An important conclusion from this analysis is that even though chlorophyll concentrations are approximately proportional to loads, species composition can change dramatically and this also needs to be considered when assessing the desirability of a particular management strategy.
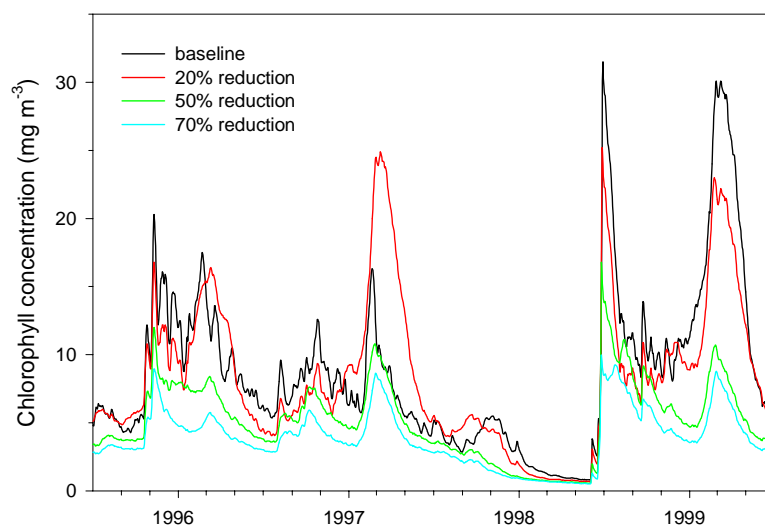
Figure 4.6 Time series of chlorophyll concentrations predicted for 0, 20, 50, and 70% reductions in nutrient loads and sediments from both eastern and western rivers.

Reduction in loads from eastern and western catchments

Loads from western catchments are 2 to 3 times those from eastern catchments in the 1995-99 period. Reducing eastern catchment loads by 20% (LE-20) has virtually no impact on 95%ile chlorophyll concentration in L. Wellington (as one might expect) and causes only a small decrease in L. Victoria and L. King (Figure 4.1). Load reductions of 20% and 40% from the western catchments alone (LW-20 and LW-40) have almost equal impact to the same percentage load reductions in all catchments in L. Wellington and L. Victoria, but are only half as effective for L. King. The reduction of 40% in

western loads alone has only a minor effect on bottom water oxygen in L. Victoria and negligible effect in L. King.

Reduction in loads from the Macalister Irrigation District

Two scenarios represent load reductions from the MID, which affect inputs through the western rivers. Scenario LMP-40 represents a 40% reduction in TP load from the MID, with no change in TN load. Scenario LMPN-40 represents a 40% reduction in TN and TP loads from the MID. Over the long term, the MID represents approximately 30% of the TP load and <10% of the TN load from western catchments. A 40% reduction in MID TP load reduces the western catchment TP load by approximately 10%. A 40% reduction in TN load reduces the western catchment TN load by only 3%. The TN:TP ratio by weight in western catchments is around 8.7 for baseline, 9.6 for LMP-40, and 9.3 for LMPN-40.

Given the results of scenarios representing the 20% reduction of western and total loads (LW-20 and L-20), one would not expect these relatively small reductions in western catchment loads to have major effects on the Lakes, and this is largely borne out by the results. Decreases in 95%ile chlorophyll are about 10% or less in L. Victoria and L. King, with quite small impacts in L. Wellington. The 95%ile *Nodularia* biomass is reduced in L. Wellington by more than 20%, and in L. King by about 20%, probably reflecting the dual effect of an absolute reduction in N and P loads, and an increase in N:P ratios. Bottom oxygen concentrations are increased slightly. Note that the nutrient loads delivered to the Lakes during the modelling period 1995-99 are estimated to be about 15% less than the long-term average loads (Appendix B). Consequently, the MID loads during 1995-99 may have a larger impact in relative terms during this time than they would over the long term.

We do not have independent information on the composition of TN and TP fractions in MID loads and have used the same rules, described in the calibration report (Parslow et al. 2001a), for allocating TN and TP to different inorganic and organic fractions in all load scenarios. Data from other irrigation areas show that nutrient loads in drains are comprised of more bio-available N and P, than is present in river flows. If this is the case for the MID, then load changes (expressed as changes in TN and TP) would be expected to have greater impact on nutrient and phytoplankton levels than equivalent changes in TN and TP loads from other subcatchments.

Long-term load increase scenarios

The long-term scenario having no change in loads is effectively an extended baseline scenario (not listed in Table 4.1). The indicators analysed for years 14-18 and for years 28-32 showed no change from the 1995-99 baseline run; that is, the model does not demonstrate a long-term trend. As discussed in the model calibration report (Parslow et al., 2001a), this should not be taken as a firm prediction. The model is quite limited in its ability to address long-term transients, because of our ignorance of the processes controlling long-term storage and release of N and P in sediments.

The results of the scenario representing a 1% increase in load per annum also need to be interpreted in this light. The model adapts to loads with a time scale of about 3 years and its behaviour between years 14-18 (LT14) and 28-32 (LT28) is consistent with the response of the model to loads that are approximately 16% and 30% larger than baseline

loads. The responses to these increased loads are shown in Figure 4.2 -Figure 4.5, and are broadly consistent with the linear responses (for chlorophyll) and non-linear responses (for primary production, bottom oxygen and bottom ammonia in L. Victoria and L. King) discussed above. Note that 5%ile bottom water oxygen concentrations are low at baseline, and while the absolute decrease in LT14 and LT28 is fairly small, this corresponds to a significant increase in the duration and potentially the impact of periods of bottom water hypoxia. There is a corresponding very significant increase in bottom water ammonia concentration, which could, at least in principle, lead to much larger dinoflagellate blooms. Scenarios LT14 and LT28 predict negative impacts on benthic plants due to reduced light penetration, with predicted loss of macroalgae from shallow parts of L. Victoria.

From an analysis of sediment core data collected throughout the Lakes by Longmore (2000a), Webster and Wallace (2001) estimate the time for continued release of P and N from the stored sediment pool after input loads of these nutrients to the Lakes are reduced. This time ranges from relatively short (~10 year) to quite long (~100 year) depending on which assumptions are made about the degradability of the solid-phase N and P within the sediments and the dominant transport mechanisms within the sediment. It is thought likely that the 'long' time scale is a substantial overestimate of the true depletion time.

The biogeochemical model considers nutrient cycling in the top few centimetres of the sediment layer only, and does not include release of dissolved nutrients from deeper in the sediment. Webster and Wallace calculated fluxes associated with degradation of organic matter 'deep' within the sediments (> 4 cm). The comparison between these fluxes and the 'surficial' sediment fluxes computed in the model provides a measure of the relative importance of long-term nutrient release to nutrient cycling in the Lakes. In all cases, deep fluxes are less than half the modelled sediment fluxes suggesting that nutrient fluxes would decline substantially if nutrient inputs to the Lakes were significantly decreased.

	L. Wellington		L. Victoria + L. King	
	N flux	P flux	N flux	P flux
Deep release	2.6	1.2	7.7	1.5
Model release	19	3.4	21	4.4

Table 4.2 Comparison between estimated flux of nutrients from 'deep' within sediments and model calculated sediment fluxes. Units are mg m^{-2}d^{-1}.

The drought year of 1997/98 within the calibration period does provide some evidence to support a short recovery time scale. There was a considerable improvement in water quality, especially in L. Victoria and L. King, during this period, and the model is calibrated so as to reproduce this improvement. It must be noted that the drought period corresponds to a reduction in flow and stratification, as well as a reduction in loads.

Implications of model uncertainty for load scenarios
A key limitation, stressed in the preceding section, is the limited knowledge and understanding of processes controlling long-term accumulation and release of nitrogen and phosphorus in sediments. The model is effectively calibrated to reproduce the observed environmental state under current forcing. We can be less sure that it correctly

predicts the long-term transient response to current or increasing loads (scenarios LT14, LT28). In addition, we can't be certain about the time scale on which the Lakes would respond if substantial decreases in load were achieved suddenly. It is possible that continuing release of nutrients from sediments would delay recovery. As noted above, the rapid improvement in the drought year of 1997 is encouraging, although it may partly reflect decreases in stratification, as well as reductions in nutrient load.

For phytoplankton bloom biomass at least, the predicted response to changes in loads is remarkably linear (Figure 4.2), despite the significant non-linearities incorporated in model processes. The response of other indicators (primary production, bottom oxygen and bottom nutrients) is more non-linear, reflecting largely the non-linear feedbacks associated with effects of anoxia on internal sinks of N and P (Figure 4.3 - Figure 4.5). The question is whether model errors substantially affect the predicted relationships between key indicators and load of the kind shown in Figure 4.2 to Figure 4.5.

There are some indications that the importance of the non-linear feedbacks in the model might be under or over-estimated at particular times and places. For example, it was noted that the efficiency of denitrification might be currently over-estimated in L. Wellington following run-off events. If this occurs because feedbacks related to organic matter flux are incorrectly represented, then reductions in nitrogen load might be more effective in reducing the duration of phytoplankton blooms in L. Wellington than the model scenarios suggest. However, denitrification efficiency in L. Wellington may be over-estimated because the model under-estimates resuspension of organic matter, and therefore over-estimates sediment respiration. This would not affect the linearity of the load-response relationship.

In the model, denitrification efficiency may be under-estimated, and bottom water hypoxia over-estimated, in western and central L. Victoria, possibly due to under-estimation of vertical mixing. This might mean that L. Victoria should behave more like L. King than the model suggests, so that load reductions of 50% rather than 70% might be sufficient to eliminate bottom water hypoxia in L. Victoria as in L. King.

In general, these kinds of errors might be expected to produce modest shifts in the inflexion points in the non-linear relationships shown in Figure 4.3 to Figure 4.5, and raise or lower the load reductions needed to eliminate hypoxia, for example, by about 20%. Given that load reductions are most likely to be implemented in a sequential manner, allowing opportunity for monitoring and feedback, these kinds of uncertainties should be easily managed.

It is important to note here that these load scenarios should all be interpreted relative to current loads. It is noted earlier, and in Appendix B, that there are considerable uncertainties in estimates of current daily loads, particularly during flood events. The model has been calibrated to reproduce (to the extent achievable) the observed time series of nutrients, chlorophyll and oxygen. We consider it unlikely that errors in calibration arising from errors in load estimates would change the qualitative response of model indicators to relative changes in loads, as reflected in Figure 4.2 to Figure 4.5. The uncertainty in load estimates does mean that managers need to think carefully about setting absolute rather than relative load reduction targets, and consider the consistency of methods used to compute loads. Errors in estimating loads, and differences that arise

from methods of estimation, along with the large interannual variability in loads, also have implications for how load reduction targets are set and monitored.

In all the load reduction scenarios, it was assumed that CDOM concentrations in river discharge remain constant. This has a significant effect on model predictions, as it means that light attenuation remains sufficiently high to prevent large macrophyte biomass establishing in any of the model basins. A switch from a planktonic to a benthic dominated system could result in substantial qualitative changes in model predictions.

The model used here does not include benthic filter-feeders. Dense populations of benthic filter feeders can exert very high clearance rates, and reduce plankton biomass substantially. Benthic filter feeders might be expected to be inhibited by both bottom water hypoxia and high suspended sediment concentrations. It is possible that feedbacks involving interactions of this kind could also play an important role in a planktonic to benthic switch, resulting in more rapid recovery than might otherwise be expected. It seems possible that the decline in water quality reported for L. Wellington in the late 1960s was exacerbated by a benthic to planktonic switch.

As discussed in Section 3, the model does not attempt to address ecological indicators such as fish and fringing wetlands. It does make some attempt at representing phytoplankton bloom composition, especially dinoflagellates and *Nodularia*. However, given limited process knowledge, predictions about changes in bloom composition in response to load scenarios are considered to be highly uncertain, and at best qualitatively plausible. However, it should be noted that phytoplankton biomass is subject to mass balance constraints, and so predictions about the response of overall biomass to load changes are considered to impose robust upper bounds on blooms.

River discharge scenarios

Alterations to river discharge

The flow scenarios involve a decrease in flow by 20% in all rivers (F-20), a decrease in flow by 20% in eastern rivers (FE-20), and an increase in flow by 20% in western rivers (FW+20). Changes in flow have been implemented without changes in loads, so increased flows correspond implicitly to decreased nutrient and sediment concentrations in flows.

Decreased flows in all rivers (F-20) produce modest increases in 95%ile chlorophyll concentrations in L. Wellington and L. Victoria (~10%) and little change in L. King. Decreasing flows in eastern catchments alone generally has similar qualitative effects on these chlorophyll statistics, but with smaller magnitude except in L. King where 95%ile concentrations decreased by ~10%. Increasing flows in the western rivers alone by 20% (FW+20) decreases 95%ile chlorophyll by ~10% in all Lakes. Bottom water oxygen concentrations are affected in a somewhat non-uniform way by changes in river discharge. The median and 5%ile oxygen concentrations in L. Victoria and L. King decrease with decreased discharges in F-20 and increase with increased discharges in FW+20, but when discharges are decreased in the eastern rivers alone (FE-20), bottom water oxygen concentrations increase slightly.

With loads fixed, the dominant effect of altering flows appears to be the effect on flushing: reduced flows result in reduced flushing and increased eutrophication, while increased flows have the opposite effect. Increased flows are likely to increase entrainment and ventilation of the bottom layer and improve oxygen concentrations there. One might expect that increasing flows into L. Wellington would reduce salinities in L. King, and so favour *Nodularia*. However, predicted 95%ile *Nodularia* biomass decreases in L. King under FW+20. Presumably, the reduction in phosphate in bottom waters and increased flushing outweigh any salinity reduction in the model.

Implications of model uncertainty for river discharge scenarios
The effects of river discharge scenarios are apparently dominated by the resulting changes in flushing rate. Given that changes in flushing rate have very similar effects to changes in loads, at least over the relatively small changes considered here, similar comments apply to those made above under load scenarios. In fact, the uncertainties involving large-scale non-linear feedbacks are unlikely to be important for these small relative changes in discharge and in predicted indicators.

Changes in river discharge might be expected to affect stratification as well as flushing. At least for these small changes in discharge, the model predicts that the effects on flushing outweigh effects on stratification. There is some uncertainty in the hydrodynamic model's ability to capture transients in stratification following flood events. However, the model does reproduce well the persistent stratification in L. Victoria and L. King over the months following runoff events, which is critical to the deterioration in bottom water quality. The model reproduces salinity and stratification over river flows which vary by orders of magnitude. This lends reasonable confidence to model predictions concerning the relative effects of flushing and stratification for a 20% change in flows. In practice, changes in flows may be accompanied by changes in loads, depending on how those flow changes were achieved. Such scenarios could be run as part of any further work.

Exchange scenarios

Existing Entrance modifications

Two scenarios involved altering the depth of the Entrance channel in such a way that the tidal exchange between the Lakes and Bass Strait was increased to 1.5 of its baseline level (E+) and reduced to 0.6 of its base level (E-). Walker and Andrewartha (2000) noted that these modifications resulted in quite small changes in salinity and flushing in the main basins. As has been pointed out in the hydrodynamics section (Section 2), this behaviour is due to the tidal excursion within Reeve Channel being small compared to the channel length. Nevertheless, increasing the tidal range increases the rate of horizontal mixing along Reeve Channel and increases the salinity at its western end where it joins L. King. This, in turn, causes a slight increase in salinity in the Lakes and an overall increase in stratification in L. Victoria and L. King with a consequent decrease in vertical exchange.

The effect on predicted chlorophyll concentrations proved to be modest also with both channel-modification scenarios showing a reduction in median and 95%ile chlorophyll concentrations of less than 10% in L. Victoria and L. King for both shallowing and deepening of the Entrance channel (Figure 4.1). Lake Wellington chlorophyll statistics

are little affected by either scenario. Shallowing the Entrance is predicted to cause a shift of the phytoplankton species composition away from *Nodularia*, but deepening the channel has a lesser effect. Bottom oxygen conditions are not much changed by scenario E+ in L. Victoria and L. King, but the 5%ile bottom oxygen concentration increases by ~30% with a more shallow Entrance channel.

Overall, the responses to E- and E+ reflect the opposing effects of reduced horizontal exchange and overall flushing, but increased vertical mixing, in E- compared with the baseline, and in the baseline compared with E+. For the most part, these responses are quite small, in keeping with small changes in average horizontal exchange rates of about 10%. However, for those model components and subsystems which are delicately poised, and/or particularly sensitive to vertical mixing, including the bottom water-sediment oxygen-denitrification coupling, and the balance between N and P-limitation in surface waters, the predicted effects of increases in vertical mixing outweigh the effects of reduced overall flushing in E-.

Second entrance

The 500-m grid used for the standard run in the hydrodynamic model was modified by inserting a second entrance to the Lakes near Ocean Grange (Figure 2.1). The entrance cell was given a depth of 1.6 m, similar to the existing opening at Lakes Entrance. A 4 m deep channel was inserted from the second entrance cell into the main body of the Lakes. On time scales of several days or more, water levels in the Lakes usually follow those in Bass Strait fairly closely in both the standard run, and the second entrance run. However, the second entrance does allow flood flows to exit the Lakes more easily, and so peak water levels during flood events are slightly lower. As well, the second entrance increases the tidal flows in and out of the Lakes, so that the tidal range inside the Lakes is almost doubled. These effects are illustrated in Figure 4.7.

The second entrance leads to significantly increased salinity values (3-4) in the main body of the Lakes (L. Victoria and L. King) and smaller increases (2-3) in L. Wellington. Figure 4.8 shows time series of surface passive tracer concentrations from the standard model run, and the second entrance run for L. King. With a second entrance, tracer disappears from the Lake about twice as fast as it does in the standard run. In general, the second entrance appears to about double the rate at which the main body of the Lakes exchanges with Bass Strait. The effect on L. Wellington is less significant.
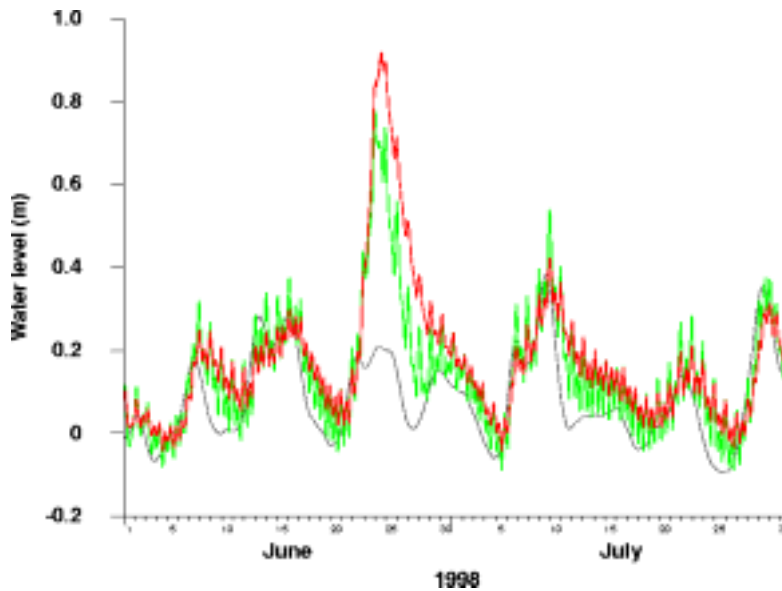
Figure 4.7 Lake King and Bass Strait water levels during June and July 1998. Lake King values are shown for the standard run (red) and the second entrance run (green). The grey plot shows low-pass filtered (48 hour) values in Bass Strait. Note that June 24, 1998 experienced the peak flows of the June 1998 flood event (see Figure 2.4).
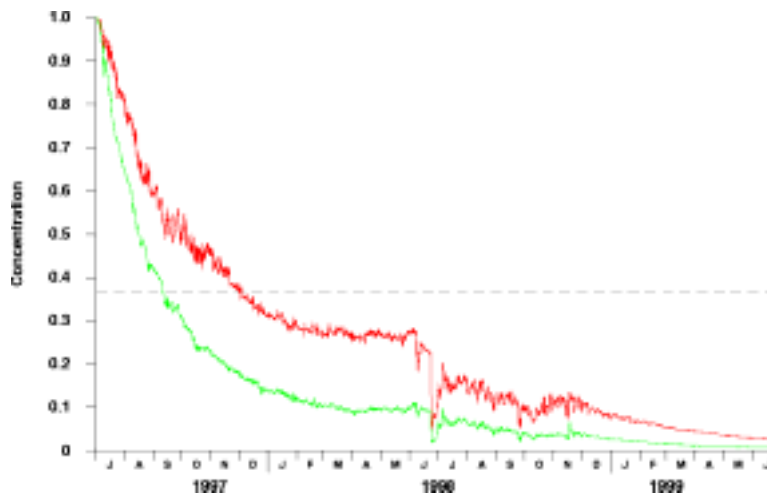
Figure 4.8 Flushing plots for L. King. Standard model run concentrations are shown in red, and modelled concentrations with the second entrance are shown in green.

The beneficial effects of the second entrance are mostly confined to L. Victoria and L. King and they are pronounced. Median and 95%ile chlorophyll concentrations are reduced by approximately a factor of two and the 5%ile of bottom water oxygen is increased by a factor of 3 to 4 (Figure 4.1). Note however that peak chlorophyll levels associated with large runoff events are virtually unchanged (Figure 4.9). The episodes of stratification, hypoxia, and nutrient accumulation in bottom waters still occur, but for shorter periods. In terms of impacts on nutrient cycling in L. Victoria and L. King, the

second entrance, which roughly halves flushing times, is comparable to a 50% reduction in loads (L-50). One would expect this to occur if the nutrient budgets are dominated by flushing rather than internal sinks. In fact, the analysis in Section 3 showed that, in the model under baseline conditions, export does dominate TN and TP budgets in L. Victoria and L. King, accounting for about 81% of TN loads and 87% of TP loads. This occurs, despite the long flushing times, because bottom water hypoxia reduces the efficiency of the internal sinks (denitrification, P burial). In fact, the second entrance has a dual beneficial effect in the model, as it not only increases flushing and export rates, but also increases the internal sinks, by reducing oxygen demand, and allowing increased ventilation of the bottom layer.

The temporal response of phytoplankton blooms in L. Victoria and L. King with a second entrance is quite different to that obtained with the 50% reduction scenario (L-50). Unlike the 50% load reduction, the second entrance does not reduce the intensity of the phytoplankton bloom immediately following flow events such as the flood of June 1998, but it virtually eliminates the *Nodularia* blooms in the autumn (Figure 4.9). The size of the bloom following the flow events is proportional to the mass of nutrient delivered to the Lake system, which has little to do with the presence or absence of a second entrance. It seems likely that the impact on *Nodularia* is due to increases in salinity resulting from the second entrance, which are certainly sufficient in the model to inhibit *Nodularia* growth. However this prediction needs to be confirmed by experimental salinity tolerance tests for strains of *Nodularia* from the Gippsland Lakes.
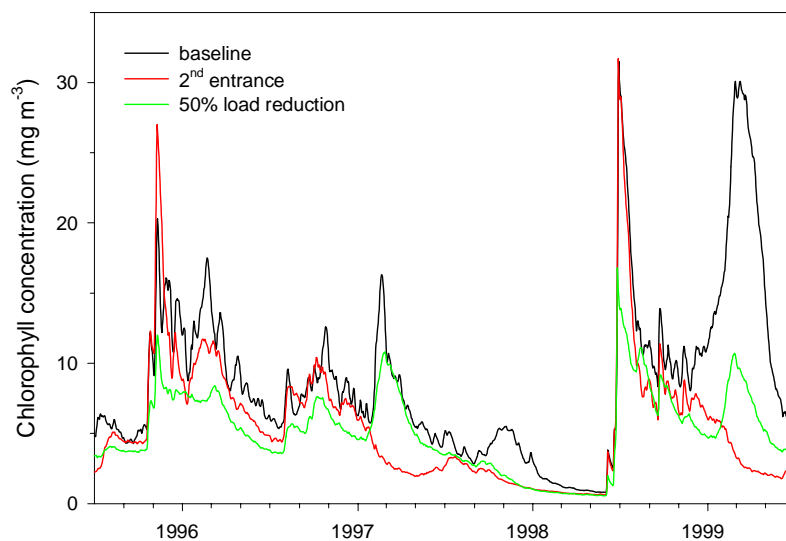
Figure 4.9 Time series of modelled chlorophyll concentrations in L. King for the baseline, 50% load reduction and second entrance scenarios.

Implications of model uncertainty for entrance scenarios

The key conclusion from the hydrodynamic model is that deepening or shallowing the existing Entrance so as to double or halve the tidal exchange will have very small (< 10%) effects on flushing rates. This is entirely consistent with our understanding of the mechanisms that make tidal exchange through a long narrow entrance ineffective at flushing, and is considered to be a robust conclusion. As a result, modifications to the existing Entrance are predicted to have small effects on nutrient cycling and water quality, and this is also considered to be a robust prediction. However, the predicted (small) effects depend on a balance between predicted changes in flushing and in stratification. The magnitude and direction of these predictions is considered to be much less certain. The effects of modifications to the Entrance on stratification will depend on patterns of longitudinal and vertical mixing along Reeve Channel, and the salinity of the water that ultimately penetrates into the bottom waters of L. King. This is likely to depend on the detailed channel bathymetry, which is only crudely captured in the Lake-wide model.

In contrast, the second entrance is assumed to be engineered so as to achieve a doubling of effective water exchange between Bass Strait and L. King, and therefore effectively a doubling of the flushing rate. (It would have course require further engineering assessments to establish whether this is technically feasible.) The model prediction that doubling the flushing rate in this way would substantially improve water quality inside L. Victoria and L. King is considered to be robust. However, there are two key areas of uncertainty in the details of this prediction.

The first concerns the hydrodynamic model prediction that vertical mixing would increase rather than decrease with a second entrance. As for the changes to the existing Entrance, this prediction would be expected to depend on the nature of the mixing in any 'new' channel connecting the Lakes to Bass Strait, and the salinity delivered to bottom water, and might therefore depend on the way in which such a channel was constructed. That said, it is true that salinities in both surface and bottom waters would be expected to approach marine salinities more rapidly with increased flushing, so that reductions in stratification would occur more rapidly during prolonged dry periods.

The second key uncertainty is associated with the model prediction that *Nodularia* blooms would be disproportionately reduced by a second entrance (Figure 4.9). This prediction depends on the assumed inhibition of *Nodularia* at high salinities, based on studies in Peel-Harvey Inlet. This assumption has not been confirmed in Gippsland Lakes and, until it is, the prediction must be treated with caution.

Lock scenarios

It has been suggested that the environmental condition of the Gippsland Lakes would be improved if they could be returned to freshwater systems. This objective could be accomplished by constructing locks across the two narrowest sections of the Lakes namely McLennans Strait and Lakes Entrance. A lock across McLennans Strait that allowed passage of freshwater flows but not return saline flows from L. Victoria would turn L. Wellington fresh. Similarly, a lock constructed across Lakes Entrance that prevented the ingress of seawater from Bass Strait would cause the entire Lakes' system to turn fresh.

The water level within a freshwater lake is maintained by a balance between the river discharge into the lake, evaporation from the lake's surface, precipitation, and the outflow to a neighbouring lake or the sea. The water levels within the Lakes that would have occurred with the lock scenarios can be simulated using available time series of river flows into the Lakes, evaporation and precipitation (Webster 2001). The simulated levels for a lock constructed across Lakes Entrance is shown in Figure 4.10 for the 23-year period between November, 1976 and September, 1999. Note that a level of zero represents the 'full' condition for the Lakes. Although the level in the Lakes would drop by less than 0.1 m in most years, during the summers of 1977/78, 1982/83, and 1997/98, the drop would exceed 0.3 m. The largest drop (0.73 m) occurs during the summer of 1982/83.
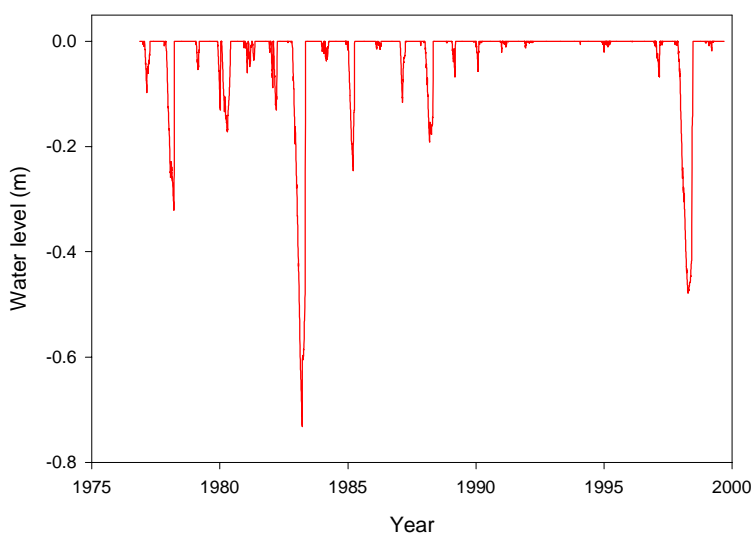
Figure 4.10 Simulated water levels in the Gippsland Lakes for the case of a lock constructed across Lakes Entrance.

A lock across McLennans Strait generally causes smaller water level changes within L. Wellington than those shown in Figure 4.10. In most summers, there would be at least a short period of time when input into L. Wellington due to river discharge and precipitation are not enough to replace the water lost by evaporation. This shortfall causes the water level to drop below its 'full' level, but the drop would be almost always less than 0.1 m. The water level would drop by more than 0.1 m only during the summers of 1977/78, 1982/83, and 1997/98. During the major drought of 1982/83, the water level drop of 0.61 m would be much larger than in the other years.

The range in water level throughout the Lakes associated with low-frequency sea level variations in Bass Strait is about 0.3 m, so water level drops caused by restricting water exchange by locks should probably not be considered significant unless they are larger than 0.3 m. By this criterion, the construction of a lock across McLennans Strait would cause significant water level changes in L. Wellington in one year out of 23, whereas

the Lakes Entrance lock would cause significant water level drops in three years out of the 23.

Inevitably, the construction of locks to prevent exchange with the sea would have huge impacts on the ecology of the Lakes. An obvious major impact is the replacement of the biota adapted to variable salinity by a freshwater ecology. The construction of locks would certainly reduce the flushing rate of the Lakes. However, it could be that the elimination of saline inflows through the Entrance would cause the Lakes to have the tendency to stratify in temperature only. Persistent stratification would not occur during autumn and winter and for the rest of the year it is possible that stratification would only occur for periods of limited duration. With the maintenance of the generally higher oxygen concentrations in bottom water that would accompany more sporadic stratification, we might expect denitrification efficiencies to be higher in L. Victoria and L. King. The reduction in flushing efficiency caused by lock construction could be more than compensated by an increased loss rate for nitrogen through denitrification. It is not known to what extent slow seepage from the ocean (or salty water from the MID groundwater area) would enable the formation of saline stratification and anoxic bottom waters. The prevention of saline exchange with the ocean is likely to impact on turbidity particularly in L. Victoria and L. King. If these two Lakes were fresh, then flocculation would be less effective at reducing their turbidity than is the present case.

The analysis has not considered the detailed effects the lock structures on the flows past them. Inevitably, the structures will impose some restriction of these flows. Any such constriction will exacerbate the problem of flooding during times of high discharge of the rivers flowing into the Lakes.

Uncertainty in lock scenarios

The principal uncertainty in the analysis of water levels for the lock scenarios arises from possible errors in the assumed evaporation rates. Evaporation was estimated from measured water level changes in an evaporation pan, which does not necessarily yield an accurate estimate of evaporation from the surface of a lake. The use of these measured evaporation rates in the hydrodynamic model yielded good agreement between modelled and measured temperatures in the Lakes for 1997/98, but modelled temperatures were significantly higher than measured during the following year (Walker et al. 2000). Using seasonal evaporation rates derived from 1997/98 in the model for the 1998/99 simulation resulted in a reasonable model fit in the second year. The difference in overall evaporation rates between the two years is ~20% and this could be considered to be an estimated upper bound in the uncertainty in yearly evaporation rates. This uncertainty translates into an approximately equivalent uncertainty in the calculation of water level depressions associated with the presence of locks; that is, the water level calculation is considered to be accurate to about 20%. The biogeochemical response of this scenario was not simulated.

5. Implications for Management

Under current conditions of catchment loads and marine exchanges, the Gippsland Lakes are subject to intense algal blooms, including toxic algal blooms, extended periods of bottom water hypoxia, and generally eutrophic conditions. The analysis and modelling conducted in GLES, and reported in Sections 2 and 3, identified a number of contributing factors, which can be summarised as follows:

- The Lakes are subject to large catchment loads of sediment and nutrient, often concentrated into flood events;
- The Lakes are shallow, so that moderate TN and TP loads per unit area of receiving waters translate, especially in L. Wellington, into very high loads per unit volume of receiving waters.
- Marine exchanges are inefficient, and the Lakes have long flushing times of 5 to 9 months between flood events.
- Lakes Victoria and King are persistently stratified, and high rates of organic matter sedimentation, following run-off events and algal blooms, lead to oxygen depletion in bottom waters. Bottom water and sediment anoxia in turn lead to large fluxes of ammonia and DIP from sediments, which stimulate further blooms of dinoflagellates and/or *Nodularia*.
- High levels of light attenuation, due to high turbidity and especially to CDOM, inhibit or prevent plant growth in bottom waters and deep sediments, which might otherwise take up nutrients and release oxygen.

The set of model scenarios described in Section 4 was designed to explore, in a fairly generic way, actions that might be expected to affect the environmental state of the Lakes. These fall into one of three broad categories: changes in loads, river flows and exchanges between the Lakes and the ocean. The GLES is intended to inform management decisions. Managers will need to take into account not only the predicted environmental impacts of various actions (and the associated uncertainty), but also the feasibility and cost of these actions, and the implied cost benefit trade-offs. In most cases, further studies would need to be undertaken to assess the cost and feasibility of the actions being considered, as well as impacts beyond those addressed in the GLES.

Management implications of load scenarios

Small decreases in loads lead to more or less proportional decreases in chlorophyll, and disproportionate increases in bottom oxygen, due to non-linear feedbacks in the modelled nutrient and carbon cycles. Thus, there is predicted environmental benefit in achieving small reductions in catchment loads, or in avoiding incremental increases in catchment loads of the kind simulated in the long-term growth scenario.

Reductions in western catchment loads have larger impacts on L. Wellington (unsurprisingly) and L. Victoria, than equivalent relative reductions in eastern catchment loads. Annual average loads from western catchments are about 3 times annual average loads from eastern catchments, and the western catchments arguably deserve first attention (see also discussion below on pre-European loads).

Relatively large (40%) reductions in MID loads correspond to only 10 and 3% reductions in western catchment TP and TN loads, respectively. Again, these have environmental benefits that are proportionally small (< 10% change in most indicators, with somewhat larger (20%) predicted reductions in *Nodularia* bloom intensity). The likelihood that the MID load may be more bioavailable than general catchment loads (something that was not modelled) may increase the benefits from changes to MID loads, although overall only by a modest amount.

All of the above scenarios produced what might be regarded as incremental changes in the environmental status of the modelled Lakes. Even a 40% reduction in western catchment loads alone produced a substantial reduction in chlorophyll biomass in L. Wellington, but only minor changes in bloom biomass and bottom oxygen in L. Victoria and L. King. Only the scenarios involving 50 or 70% reductions in loads from both eastern and western catchments led to substantial reductions in bloom intensity in all basins and to elimination of episodes of bottom water hypoxia / anoxia in L. Victoria and L. King.

Load reductions of order 50 or 70% are clearly ambitious, and it is not clear how they might be achieved, or indeed whether they are feasible. Following requests from the GLES Steering Committee, Grayson et al. (2001b) have estimated pre-European loads. They stress that these are preliminary crude estimates, and highly uncertain. They found that that pre-European loads of TSS, TN and TP were respectively 55%, 45%, and 75% less than current loads in western catchments, 10%, 10% and 25% less in eastern catchments and 50%, 35%, and 65% less overall. If correct, these results indicate that load reductions of around 40-50% for TN and 70 % for TP would be needed in the western rivers to approach pre-European conditions. In their analysis, Grayson et al. considered reductions in total nitrogen and phosphorus loads since it was not possible to consider bioavailability given the data available. It is likely that the composition, and in particular the bio-available and labile fractions of the loads, would have been different in pre-European times especially in the western catchments which have a higher level of agricultural activity than the eastern catchments.

The implications of model limitations and uncertainty for the load scenarios are dealt with at length in Section 4. As a general conclusion, model uncertainties are unlikely to change significantly the predictions about the relative effects of small (20% or less) changes in loads. There is increased uncertainty attached to predictions about larger relative changes, so that it is not possible to assert confidently (for example) that a 70% rather than a 50% reduction in loads is required to eliminate bottom water hypoxia. This can be partly attributed to uncertainty about the non-linear feedbacks represented in the model, and partly to uncertainty involving feedbacks and processes not represented in the model, e.g. benthic filter feeders.

There is of course uncertainty in estimates of loads into the Lakes during the calibration period. Again, this is not expected to substantially affect model predictions about the *relative* response of indicators to *relative* load changes. Uncertainty in load estimates, and the very large interannual variability in loads, does have significant implications for setting and monitoring load reduction targets. Managers should be very cautious about setting absolute load or load reduction targets, in tonnes per year, and be sure that methods for the estimation of loads are consistent.

It has only been possible within the scope of the present study to run a limited set of model scenarios, and there are many more which might be run. The scenarios have all involved fixed percentage reductions in loads, with no distinction between base and peak loads. All the load scenarios, with the exception of the MID scenarios, have involved equal percent reductions in TN, TP and TSS, and in all the component fractions. It is unlikely that any actions in the catchment to reduce loads would have effects of this kind, and it is possible that different patterns of reduction over time and across components, would produce different effects.

While many more "hypothetical" or "generic" scenarios might be run, it is clearly desirable that scenarios, and management decisions, be better informed by understanding and knowledge of the catchment. It is important to understand how loads into the Lakes are sourced across subcatchments, and how actions in the catchment might be expected to affect the delivery of loads to the Lakes. Ideally, the GLES model should be coupled to a catchment model based on at least comparable understanding and predictive power. The need for understanding of the catchment is highlighted by the pre-European load estimates discussed above.

Because actions in the catchment to reduce loads are likely to be sequential and incremental in nature, it is not so critical to reduce the uncertainty in prediction about the impacts of large reductions in loads. Sequential actions of this kind are ideally suited to an adaptive management program, in which monitoring of environmental indicators in the Lakes is used to provide ongoing feedback to management decisions. However, it would be highly desirable in such a program to have a better idea of the likely response time scale of the Lakes. If continued sediment nutrient efflux resulted in long (decadal) delays in improvement, this could lead to both frustration and/or inappropriate decisions in load management. It follows that investment in further process studies targeted at understanding the long-term response of sediment fluxes to load reductions could be justified.

Management implications of river discharge scenarios

Decreases in flow (keeping loads fixed) have similar, albeit proportionally smaller, impacts on chlorophyll and bottom oxygen to increases in load. There is environmental benefit in increasing flows in western catchments, and maintaining flows in eastern catchments.

Effects of model uncertainty on flow scenarios were also discussed in detail in Chapter 4. It was generally concluded that, for the small relative changes considered, the model predictions are likely to be qualitatively robust.

The flow scenarios are particularly limited in scope, involving fixed percent reductions in daily flows over base and flood flows, in some cases spanning 5 orders of magnitude. Changes in water use in the catchment are unlikely to produce these kinds of changes, and there might well be benefit in exploring separately the effects of changes to base and peak flows, and even to changes in intermediate runoff event relative to flood flows.

The flow scenarios were also run with unchanged loads, so that concentrations were assumed to increase as flow decreased. Changes in water use again might be expected to

alter the relationships between concentration and discharge. Specific catchment actions might be expected to modify flows and loads in diverse ways.

As discussed above for loads, before proceeding to consider a diverse set of hypothetical coupled flow and load scenarios, it is desirable that more realistic scenarios be developed based on understanding and modelling of catchment processes and potential management actions.

Management implications of entrance scenarios

The engineering scenarios involving modifications to the existing Entrance produced only marginal changes in environmental indicators in the modelled Lakes, primarily because the modifications produced only small changes in flushing rates. There is little encouragement in these results for deepening the existing Entrance, unless a way can be found to drastically increase the tidal exchange and the effective flushing rate of L. King and L. Victoria.

The second entrance scenario did produce a substantial improvement in simulated environmental health in L. Victoria and L. King. As modelled, this scenario effectively halved flushing times for these lakes. This resulted in an approximate halving of the median and 95%ile chlorophyll levels in L. Victoria and L. King, although it did not reduce peak post-flood algal bloom levels. It did result in substantial improvement in bottom water hypoxia in these lakes, comparable to that achieved by a 50% load reduction. The second entrance had largely negligible effects on water quality indicators in L. Wellington.

The extent of the impact of a second entrance on L. Victoria and L. King was not expected *a priori*, but is consistent with the dominant role of export in the TN budget for these lakes (Table 3.2), because denitrification is shutdown during periods of bottom water hypoxia. As discussed in Chapter 4, the prediction of substantial benefit is considered to be robust, although the reduction in bottom water hypoxia is enhanced by predicted decreases in stratification, which would need to be confirmed by further detailed modelling of entrance dynamics. The second entrance scenario is predicted to be particularly effective at reducing *Nodularia* blooms, because it increases salinities in L. Victoria and L. King to levels that are assumed to inhibit *Nodularia*. These assumptions would need to be tested through process studies.

Although the second entrance appears to be superficially attractive in terms of water quality indicators in L. Victoria and L. King, it would have little or no beneficial effect on L. Wellington, would involve a very substantial capital outlay, and probably considerable ongoing expenditure to maintain dredged channels into L. King proper. Further study, including detailed engineering modelling of any proposed channel, would certainly be required before committing to such expenditure.

The second entrance is predicted to increase salinities in L. Wellington by 2 to 3, and in L. Victoria and L. King by 3-4. Although this is small compared with the seasonal and interannual variation in salinity (of order 15), it could have significant ecological impacts in the Lakes, or in the lower reaches of waterways and wetlands adjacent to the Lakes. Assessment of these impacts is not within the scope of the current study. However it is likely that the increased salinity would lead to an upstream transition to

the more saline resistant species found near the Entrance. Changes to the aquatic and fringing terrestrial vegetation are likely to impact the faunal communities that form an integral part of the Lakes' ecosystem. Another concern is the potential for increased salinity to adversely affect the Lakes' bream fishery.

Management implications of the lock scenarios

The construction of locks to maintain the Lakes as freshwater systems faces considerations similar to those for a second entrance. Such works would also require substantial capital outlay and ongoing maintenance expenses. Turning the Lakes fresh would cause enormous ecological changes which are not addressed in this report, but would need to be evaluated very carefully by the affected communities and environmental managers if this strategy were to be considered.

Implications of scenarios for environmental objectives

The scenario results, in combination with the estimates of pre-European loads, raise some interesting issues for managers and for stakeholders. The results imply that, in their current (stratified) physical configuration, the Lakes are so susceptible to eutrophication that they would display "undesirable" environmental attributes (algal blooms, bottom water hypoxia), even if catchments were in a "pristine" or pre-European state. This is not necessarily surprising: there are certainly other estuaries in Australia with near-pristine catchments which exhibit bottom water hypoxia or anoxia.

These results raise questions about environmental objectives as well as the cost and feasibility of management actions. There is substantial debate as to whether, in modified systems such as the Gippsland Lakes, we should aim to return the system as near as possible to "pristine" state, or whether we should set environmental objectives which meet stakeholder needs, and manage to achieve them. If we conclude that episodes of bottom water hypoxia are "natural" events in the Gippsland Lakes, or other estuarine systems, should we still attempt to eliminate them?

Managing uncertainty

The appropriate management response to uncertainties in system response to management action will vary, depending on the nature of the actions being contemplated. Where management actions are sequential and incremental, as seems likely for reductions in catchment loads, an adaptive management approach seems most appropriate. This would involve the establishment of an appropriate monitoring program in the Lakes, and development of a strategy involving sequential implementation of load-reduction activities in the catchment, with feedback rules based on assessment of environmental outcomes.

Where proposed management actions involve major capital outlays, as for example with a second entrance, locks, or dams, there is obviously benefit (within constraints of cost and subject to managers' risk tolerance) in reducing uncertainties up front. The next step might involve engineering models, and further process studies leading to refinement of the biogeochemical model, as part of a targeted feasibility / environmental impact study.

6.   Recommendations for Continuing Work

Recommendations for a monitoring program

We suggest that a monitoring program be implemented with the prime purpose of assessing progress towards management objectives and to inform decisions on modification of management actions over time in response to system change. The challenge facing any monitoring program in the Gippsland Lakes is the large temporal variability of the system at time scales of weeks, seasons, and years. The linear regression of EPA chlorophyll data collected over the last 15 years at intervals of 1-2 months at only one or two sites in each Lake shows a small increasing trend, but the interannual variability evident in these data render this trend statistically meaningless. Assessing trends in Lakes condition based on such statistical analysis techniques is likely to require decades of measurements before the analysis yields statistically significant results.

We propose that a more effective alternative strategy is to use the hydrodynamic and biogeochemical models in diagnostic mode to continuously assess the cause and effect relationship between management actions and system biogeochemical response as evidenced in an ongoing monitoring program. For example, suppose a management action resulted in a decrease in load by a few percent per year, then the models would predict a particular response for the yearly median chlorophyll concentrations (say) that would inherently include the effects of interannual variability manifest through their forcing by loads, freshwater inflow, and meteorological conditions. A comparison between trend analyses undertaken on measurements and on model predictions would aid in establishing how much of the measured trend was due to statistical uncertainty arising from natural temporal variability in the Lakes and how much was due to an underlying change in their condition.

An important role for the ongoing application of the models is their use in fine-tuning management strategies as the condition of the Lakes evolves. For this purpose and for their potential use in the assessment of Lakes' condition it is important that the uncertainty in model simulations be reduced. The monitoring program will provide data for ongoing calibration, validation, and improvement of the models so that their effectiveness as management tools is enhanced

Key considerations for the design of a monitoring program are the choice of indicators, the choice of sampling frequency and location, and the manner in which the results are to be interpreted. The suggestions for a monitoring program presented here are designed specifically to support the ongoing assessment of the effects of management actions on key indicators of the biogeochemical function of the Lakes, taking into account the diagnostic use of the hydrodynamic and biogeochemical models. Of course, the monitoring strategy recommended does not preclude the need to monitor other indicators not connected directly with the biogeochemical function of the Lakes such as higher trophic levels such as invertebrates, fish, and birds.

Riverine loads

Accurate evaluation of riverine loads of nutrients, CDOM, and suspended particulate matter has the highest priority in our proposed monitoring strategy. It is absolutely essential for the direct assessment of the effectiveness of strategies that might be implemented for reducing loads from the catchments. It is also fundamental for the interpretation of observed changes in the Lakes, including the diagnostic use of the models. Management of loads requires knowledge of where they are coming from and what factors determine their composition. The acquisition of this knowledge is a major undertaking by itself and warrants significant investment of effort. Flow events deliver a major portion of the annual loads of nutrients to the Lakes and it is imperative that loads from these events, which may only last a week or two, are estimated accurately. It is important that the composition of the loads be determined, as this strongly affects their ecological impact. Parameters to be measured would include ammonia, nitrate, phosphate, organic nitrogen (both dissolved and particulate), total nitrogen, total phosphorus and particulate inorganic phosphorus.

Salinity

Salinity is a major indicator for flushing and stratification and will be affected by river discharge and exchange scenarios. Aliasing of individual salinity profiles is likely due to internal seiching of the halocline. This problem could be addressed using continuously recording salinity meters moored near the bottom in the centres of the major basins.

Dissolved oxygen

Dissolved oxygen (DO) is a critical water quality indicator. Low concentrations are conducive to fish deaths and hypoxic bottom waters lead to enhanced release of ammonia and phosphate from sediments. As with salinity, it is necessary to resolve the vertical profiles of DO and to make measurements sufficiently frequently to resolve the onset of hypoxia in bottom waters. Monthly monitoring is marginal for the resolution of hypoxic events and the use of moored continuously recording bottom DO sensors would be desirable. These could be combined with salinity sensors.

Chlorophyll a

Chlorophyll *a* is an indicator of the presence of algal blooms. It is apparent from this study and from others that blooms sometimes persist for only a few weeks and that they may be missed or poorly sampled by monthly sampling. Temporal variability of chlorophyll concentrations can be addressed using moored fluorometers. Blooms can also show considerable spatial heterogeneity. Remote sensing techniques using aircraft or satellites might be used to estimate the spatial distribution of near-surface chlorophyll concentrations, although this would require development and validation of algorithms that can discriminate among chlorophyll, CDOM and TSS. Samples need to be collected during blooms particularly for the analysis of the phytoplankton species composition.

Light attenuation

The light climate is crucial for the growth of phytoplankton and benthic plants and the clarity of the water is an important water quality indicator in its own right. More frequent measurements of Secchi depths and perhaps direct measurements of profiles of photosynthetically active radiation (PAR), turbidity and light transmissivity in the water

column (during surveys) are needed to establish the variation in light climate due to flow or wind events. Continuously measured PAR at the surface and the bottom would provide a direct measurement of the attenuation of light necessary for the growth of benthic flora as well as phytoplankton. Given the important role of CDOM in light attenuation, measurement of CDOM should be included in the suite of measurements in regular sampling programs in the Lakes, and in riverine inputs.

Particulate dynamics

The interplay between settling and resuspension determines the concentrations of particulate material in the water column. Resuspension of settled phytoplankton cells may be an important factor in maintaining phytoplankton concentrations in the water column in L. Wellington. Inorganic and organic particulates are important for determining the underwater light climate in the Lakes and adsorption/desorption to suspended particulate is important for phosphorus dynamics. Suspended sediment concentrations are likely to be driven by wind events, to fluctuate on short time scales, and to be highly aliased by fortnightly or monthly sampling. We recommend that a monitoring program for the Lakes should include continuous moored measurements of turbidity, along with fortnightly or monthly water column sampling for TSS. This would allow calibration of turbidity as a surrogate for TSS, and provide data to establish and model the relationship between wind events and suspended matter concentrations. It is important to differentiate between organic and inorganic suspended material, but this would be best accomplished from the analysis of collected water samples.

Nutrients

As key determinants of phytoplankton growth, the maintenance and extension of the current monitoring program for nutrients in the Lakes is essential. Changes in water column nutrient concentrations with time are an indicator of the impact of adopted management strategies. The sampling program should measure ammonia, nitrate, phosphate, total nitrogen, total phosphorus and particulate inorganic phosphorus in the Lakes on time and space scales that resolve the principal temporal and spatial variations of their concentrations sufficient to construct sensible budgets for estimating their transport, transformation and loss. For stratified L. Victoria and L. King, this requires measurements to be obtained in both the upper and lower layers. Variation in nutrient concentrations occur on the time scales of flow events, phytoplankton blooms, and of the onset of hypoxia in bottom waters, so sampling should occur on at least a fortnightly basis.

Recommendations for further process studies

While the hydrodynamic and biogeochemical models are generally able to reproduce most aspects of the observed system behaviour, at least qualitatively, the calibration process revealed a number of limitations and questions. Model limitations and uncertainties are discussed at length in the "Gippsland Lakes Integrated Model – Model Development and Calibration" (Parslow, 2001a). In this section, we summarise these limitations, focusing on their implications for the interpretation of scenario results. These limitations could be the foci of further process studies.

Sediment biogeochemistry

We do not understand processes controlling long-term storage of phosphorus and nitrogen 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 so the model cannot address the issue of how long will sediments continue to release nutrients from 'deep' stores after nutrient inputs to the Lakes are reduced. A range of long-term release time scales has been estimated from sediment cores, but these estimates are based on the indirect evidence of the shape of the concentration profiles of solid-phase nutrients (Webster and Wallace, 2001). We need to know how much of the large measured stores of N and P (Longmore, 2000a) is labile and what are the mechanisms and rates of material transport within the sediment column.

Improved understanding of long-term sediment dynamics has important practical implications for establishing any adaptive management program in the Lakes. It is critical that managers (and stakeholders) have realistic expectations of the time frame on which responses to load reductions or other actions should occur.

Phosphorus dynamics

The model does not properly buffer water column phosphate concentrations within the observed range in L. Wellington, and the model may be too easily pushed into P-limitation. More studies need to be undertaken on the adsorption/desorption processes between phosphorus on suspended and benthic sediments including the effects of pH and ionic strength. There seems to be a binding mechanism for P in the deep anoxic sediments in L. Wellington and possibly in the other Lakes that plays a role in the burial of this element, but which is not understood. The rates of P burial in L. Wellington sediments are under-estimated with the present model dynamics. Whether the phytoplankton are P or N limited has implications for the composition of algal blooms and their potential impact on the ecology and amenity of the Lakes. N-limitation tends to favour the growth of the toxic cyanobacterium, *Nodularia*.

Phytoplankton dynamics

Phytoplankton concentrations in the model 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. If this is the case, the model may under-estimate the sensitivity of L. Wellington, and over-estimate the sensitivity of L. Victoria, to changes in loads.

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 autecology and population dynamics would be required to improve model representation of phytoplankton composition and 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 behaviour is not consistent with observations, and suggests some key limiting process may be missing.

Hydrodynamics

The hydrodynamic- and box-model representations of overall flushing rates are 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. Much of the improved system function evident in the second entrance scenario appears to result from improved ventilation of the bottom layer in these two Lakes. If such a scenario is to be seriously considered, it is essential that the apparent benefit is demonstrated to be real and not an artefact of the model application. As has been pointed out in the hydrodynamic section of this report, the discrepancy between model simulation and measurement of bottom water salinities in L. Victoria and L. King may be due to aliasing. This possibility needs to be checked.

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Appendix A. The Hydrodynamic Model

The model used to simulate flow and mixing within the Lakes is MECO, a hydrodynamic model developed by CSIRO for estuaries and coastal oceans. The model represents the Lakes as a three-dimensional array of grid cells and solves equations for the conservation of mass, momentum, and advection/diffusion equations in each cell to provide three-dimensional distributions of velocity, temperature and salinity as well as concentrations of an arbitrary number of passive tracers (Figure. A1). For solution, the model requires input fluxes of water and tracers, and forcing by winds, atmospheric pressure gradients, sea-level (including tides) and surface heat fluxes. A full model description can be found in Walker and Waring (1998) and its application to the Gippsland Lakes is presented by Walker and Andrewartha (2000).

Each grid cell resolves T, S, ρ, other tracers
(T=temperature, S = salinity, ρ = density)
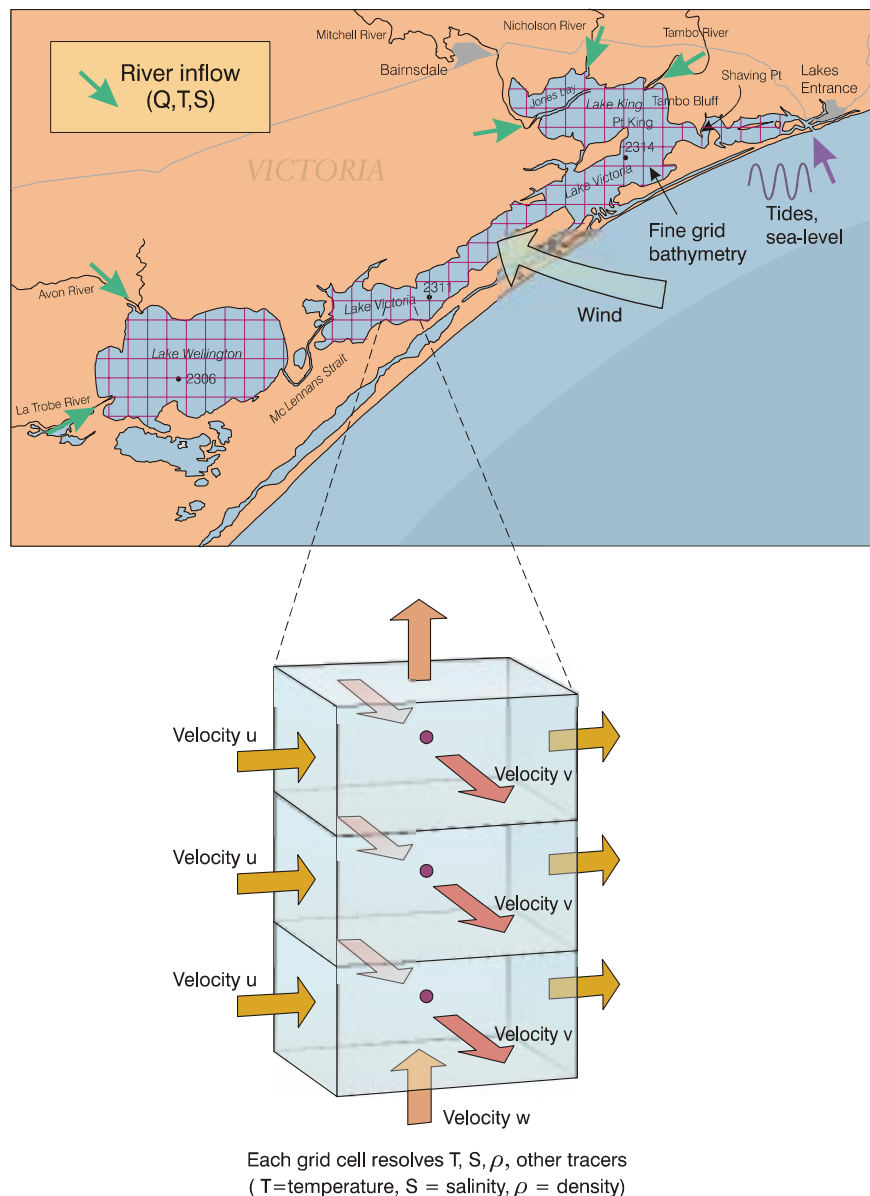
Figure A1 Schematic of hydrodynamic model application to the Gippsland Lakes.

For its application to the Lakes, the model used a horizontal grid of spacing 500 m and a vertical discretisation of 0.5 m (Figure A2). A 250-m grid was originally formulated for the model, but the time for simulations proved to be unfeasible. The model domain was extended beyond the Entrance to include a section of Bass Strait. Dye studies have shown that a significant fraction of the water that flowed out of the Entrance on the ebb tide re-entered on the flood so it was necessary to simulate the water movements in the coastal region outside the Entrance. Calculations are performed for all water (blue) cells within the Lakes and the included portion of Bass Strait, and boundary values are specified along the outer edge of the grid in Bass Strait.

A horizontal resolution of 500 m is not adequate to represent the narrower channels in the system such as those leading to the Entrance. In particular, the Entrance has an actual width of only 120 m. Consequently, the modelled depths in this region were adjusted to optimise the modelled tidal and salinity responses inside the Lakes. McLennans Strait was also modified to better represent exchange between L. Wellington and L. Victoria. The modelled Strait was made to run in a straight line, which then allowed the grid to be locally distorted to give the Strait more realistic width (and cross-sectional area). As a result of this modification, it was necessary to extend the grid and shift the position of L. Wellington, so that the Strait still had the correct length, and entered L. Wellington in the correct position.
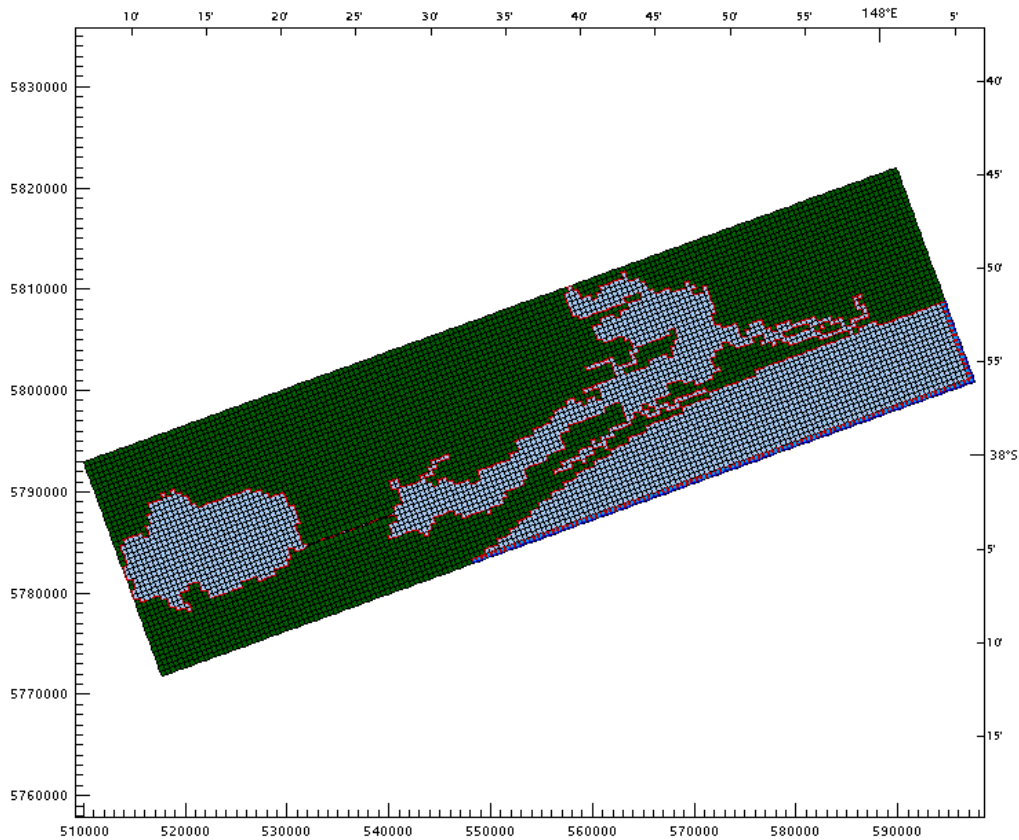
Figure A2 The 500-m model grid.

Appendix B. Computing Loads of Nutrients and Sediments

The following summarises the calculation of loads of total suspended solids (TSS), total phosphorous (TP) and total nitrogen (TN) into the Gippsland Lakes. Daily loads were estimated for the period 1975-1999 and used as inputs for the biogeochemical model for the period July 1995-June 1999. A full description of the procedure for calculating loads is provided by Grayson et al. (2001a).

The methodology employed in computing loads is based on deriving relationships between the concentrations of transported material and a combination of discharge and other variables. This enables equations for each river to be developed that predict daily loads for any period where flow data are available. This approach is needed because there are insufficient data available for direct computation of loads during the modelling period. It also enables loads to be easily computed for any other period if required, and provides a consistent approach to load estimation across all of the rivers. The initial approach formed regressions on the basis of a consistent data set from the EPA, dating from the mid 1970s, and following standard procedures for establishing the statistical validity of the regression relationships. In a second stage, for some sites, we stratified the data on the basis of discharge, separating out the lower flows and forming regressions on the higher flow data. For the lower flow data, a mean value of concentration was used. This approach was found to give more reliable load estimates for some rivers and constituents.

Comparisons were made between the observed daily loads computed directly from the data during the period 1977 to 1990 and the predicted daily loads based on the original and "two part" regression relationships. These comparisons indicated that the two-part approach produces similar estimates to the original approach but is slightly better at the higher flows. Comparisons were also made, where possible, with data collected for other studies. This procedure provided an independent check on methodology. In all cases, the fits to independent data were almost as good as to the main data set used for derivation of the relationships indicating that the relationships are reliable.

For each of the rivers, there is an area downstream of the gauging station that provides some load to the Lakes but is not monitored. This load was estimated as described below and added to the loads into L. Wellington, L. Victoria, and L. King. The method used was to determine the areas draining into each river system or directly to the Lakes, downstream of the gauging station at which the load relationships were computed. The mix of land uses in these areas is similar to those in the La Trobe River catchment, excluding the irrigation areas. Long-term load estimates from the La Trobe River were converted into areal loading factors. These were compared to published values and found to be equivalent to "typical" long-term loads summarised in the literature. These areal factors were used to compute long-term annual loads from the ungauged areas.

In order to allocate loads from the ungauged areas to particular years, the long-term averages were multiplied by the ratio of annual river loads computed as above for the particular year divided by the long-term average annual river loads for the full period of the load simulation (1975 to 1999). Daily loads were computed using a similar approach whereby the annual load was disaggregated on the basis of the proportion of annual river load occurring on a particular day. Loads from these ungauged areas make up

approximately 20% of the total loads into the Gippsland Lakes and so are quite important from a management perspective.

The approach described above does not include contributions from the Macalister Irrigation District (MID) that flow into the rivers downstream of the gauging stations. Sinclair Knight Merz (SKM) was contracted as part of another study to develop daily modelling of TP and TN loads in all of the MID drains. They have provided their data to us for inclusion in the final load estimates for the Lakes. Daily loads (1978 to 1999) from all drains entering the streams flowing to L. Wellington downstream of the gauging stations were added to the load estimates computed for the rivers to give a final set of loads for use in modelling.

The SKM work does not include estimates of TSS loads. We assessed several options for estimating TSS loads in the drains. Each was highly uncertain but generated loads that were always less than 10% (generally less than 5%) of the La Trobe River loads (i.e. only a few percent of total loads). Thus, the error introduced by ignoring TSS loads from the MID drains downstream of the gauging stations is considered to be negligible.

Table B.1 compares the annually averaged load estimates from all rivers and drains flowing into the Lakes for the four-year period used in the biogeochemical model, to the long-term average loads for 1975-1999. The average loads of TN and TP to L. Wellington and L. King for the modelling period are about 15% less than the long-term average. These two Lakes together account for the vast majority of the loads to the Lakes. The TSS load is 23% less for the modelling period than the long-term average for L. Wellington and 17% less for L. King. The below average loads experienced during the modelled period are primarily due to the very low loads from 1997.

	Period	Load TSS	Load TP	Load TN
L. Wellington	1975-1999	165,870	220	1,944
	1995-1999	127,660	186	1,626
L. Victoria	1975-1999	8,490	11	124
	1995-1999	7,210	8	118
L. King	1975-1999	45,270	71	731
	1995-1999	37,560	58	650

Table B.1 Comparison of average annual loads from 4-year baseline (1995-1999) to long-term loads (1975-1999). Units are tonnes/year

Based on comparisons between measured and simulated loads, an assessment of the data quality, statistical soundness of the regression relationships, and discussions with SKM regarding their modelling of the MID, we can make a qualitative judgment about the maximum and likely errors in loads. For non-extreme flow conditions, we expect the error in load estimates to be of the order of +/- 20% and unbiased over time periods of months to years. This may possibly increase to as high as 100% for individual events of very high magnitude (due to increasing uncertainty in flow estimates as well as concentration), but again the results should be unbiased in the longer term.

List of Study Reports

The following are reports prepared as part of this study. They can be accessed at the Gippsland Coastal Board web site: http://www.vcc.vic.gov.au/gcboard/

1. Estimation of Sediment and Nutrient Loads into the Gippsland Lakes. (Grayson et al. 2001a).
> This report describes the methods used to estimate nutrient and sediment loads to the Gippsland Lakes over the last 25 years. It presents summary statistics of these analyses and estimates of error.

2. Pre-European Load Estimates into the Gippsland Lakes. (Grayson et al. 2001b)
> This report describes the methods used and the results of an analysis to estimate pre-European nitrogen and phosphorus loads to the Lakes.

3. Integrated Model Implementation and Calibration – Final Report. (Parslow et al. 2001a)
> This report describes the application of the biogeochemical model to the Gippsland Lakes. It includes a description of the state variables used in the model, model calibration, and presents an extensive set of results from the application of the model to the Lakes. In its model description, it focuses particularly on novel model developments for this project.

4. Gippsland Lakes Environmental Study – Hydrodynamic Modelling. (Walker and Andrewartha 2000)
> This report details the application of the hydrodynamic model to the Gippsland Lakes. It includes discussions of grid selection, data sources, comparisons between model predictions and measurements, and the impacts of the exchange scenarios on the flushing characteristics of the Lakes.

5. Examination of Water Balance Scenarios for Locks Across Lakes Entrance and McLennans Strait. (Webster 2001)
> This report considers the estimated impacts on water levels within the Lakes over the last 23 years due to the (hypothetical) construction of locks across the Entrance and McLennans Strait designed to prevent the ingress of sea water.

6. Isotherm Analysis of Sediment Samples and Water Column Samples Collected in the Gippsland Lakes 2000 – 2001. (Webster and Grace 2001)
> This report presents the results of experiments designed to determine the phosphorus adsorption characteristics of sediments collected at sites throughout the Lakes. Freundlich isotherms are fitted to the results.

7. Further Analysis of Sediment Core Samples Collected in the Gippsland Lakes – 2000. (Webster and Wallace (2001).
> This report describes the results of further analyses on sediment cores collected throughout the Lakes by Longmore(2000a). The report mainly considers the likely long-term release of nitrogen and phosphorus from buried stores, but also addresses transport processes and sedimentation rates in the Lakes.

Glossary

Algal bloom: Microalgae occurring in dense numbers in a water body, as a result of favourable conditions (ie. nutrient enrichment).

Aliasing: A sampling error caused by inadequate sampling frequency.

Ammonia: Compound consisting of a single nitrogen atom coupled with three hydrogen atoms. It is a nitrogen source for algae.

Anaerobic respiration: Bacterial respiration supported by reduction of sulphate rather than oxygen consumption.

Anoxic: Devoid of oxygen.

Bathymetry: Depth characteristics of a water body.

Benthic: Belonging to the sea floor.

Biomass: The amount of living material (plants or animals).

Calibration: Use of measured field data to set up a mathematical model.

Catchment: Area of land from which run-off from rain enters a waterway.

CDOM: Colour dissolved organic matter.

Chlorophyll: The green pigments of plants that capture and use the energy from the sun to drive the photosynthesis process.

Coriolis force: The force due to the earth's rotation which moves air and water currents to the left in the southern hemisphere and right in the northern hemisphere.

Cycling/recycling: The movement of an element (like nitrogen) through various forms, living and non-living back to its starting form.

Denitrification: Conversion of bound nitrogen to elemental (gaseous) form.

Detritus: Non-living organic matter (e.g. dead algae).

Diatom: Variety of microalga that has a siliceous skeleton.

Dinoflagellates: A class of motile phytoplankton with two flagellae, which often form nuisance or toxic blooms in coastal waters.

DOM: Dissolved organic matter

Ecosystem: A community of plants or animals or both.

Epibenthos: Plants and animals living at the interface between the water column and the bed sediments.

Eutrophic: Having an unnaturally high content of algae due to excess nutrients.

Flagellates: Microscopic single-celled plants or animals, but here used to refer to small single-celled plants, typically less than 10 microns in diameter.

Flocculation: The process of particle aggregation in water.

Fluorometer: An instrument for measuring fluorescence (e.g. of chlorophyll).

Flushing: The rate at which a lake or bay changes its water content.

Flux: Rate of flow of material.

Geochemical: Relating to earth chemistry.

Grazing: Eating of plants by animals. In water the term is associated with zooplankton grazing on phytoplankton.

Haline: Salty.

Halocline: Zone of maximum salinity stratification.

Hydrodynamic: Related to movement of water.

Hypoxia: Low concentration of oxygen in the water column, stage prior to reaching anoxia (no oxygen).

Inputs: Substances entering a water-body.

Interstitial: Relating to interstices between grains in sediment.

Labile: Organic matter that is readily broken down by bacteria, on time scales of days.

Macroalgae: Multi-cellular plants that are visible to the eye, in this report used to describe seaweeds.

Macrophyte: Large plants that grow in an aquatic environment.

Mesotrophic: Water body that has a moderate algal population.

Microphytobenthos (MPB): Single-celled algae which live in and on the sea floor. They are mostly diatoms.

Nitrate: The NO_3 anion.

Nitrification: Formation of nitrate from reduced forms of nitrogen.

Nitrite: The NO_2 anion.

*Nodularia***:** Photosynthetic bacteria (cyanobacteria) that are capable of fixing dissolved nitrogen gas (N_2), and forming toxic blooms in brackish water.

Nutrients: Substances (eg. nitrogen and phosphorus in various forms) required for the growth of plants (like fertiliser).

Oxic: Having oxygen present.

PAR: Photosynthetically Active Radiation. The spectrum of light required by plants for photosynthesis.

Particulates: Particles suspended in water.

Photosynthesis: Transformation of carbon dioxide and water to organic matter and oxygen by means of light energy.

Phytoplankton: Microalgae that live in the water column.

Redfield ratio: The atomic ratio of the major constituents of phytoplankton and aquatic plants.

Refractory: Organic matter that is not readily broken down by bacteria, and is remineralized on time scales of months to years.

Salinity: The salt content of water. Seawater has a salinity of approximately 35.

Seagrass: A group of flowering plants which live rooted in the sea floor.

Secchi Depth: A measure of water transparency.

Seiching: Oscillations of water in an enclosed body.

Stoichiometry: Term in chemistry that describes a balance of elements in reactions.

Stratification: Layering (usually due to temperature or salinity differences).

Turbidity: Cloudiness of water.

Zooplankton: Small animals living in the water column, usually drifting with the water.