Sensitivity analysis of the CSIRO model for the Gippsland Lakes.

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1 INTRODUCTION........................................................................................................................................2

2 APPROACH ...........................................................................................................................................3
  2.1 SEARCHING FOR ALTERNATIVE PARAMETER SETS .................................................................3
  2.2 CHOICE OF SCENARIOS.............................................................................................................4
  2.3 ASSESSING THE SCENARIOS .......................................................................................................6

3 VERTICAL EXCHANGE ....................................................................................................................8
  3.1 BACKGROUND............................................................................................................................8
  3.2 APPROACH ....................................................................................................................................9
  3.3 RESULTS.......................................................................................................................................10
     3.3.1 Effect of parameter variation on baseline dynamics ........................................................10
     3.3.2 Impact on Scenarios............................................................................................................13

4 NODULARIA .......................................................................................................................................15
  4.1 BACKGROUND...........................................................................................................................15
  4.2 APPROACH ...................................................................................................................................16
  4.3 RESULTS.......................................................................................................................................18
     4.3.1 Effect of parameter variation on baseline dynamics ........................................................18
     4.3.2 Impact on Scenarios............................................................................................................22

5 SEDIMENTS .......................................................................................................................................25
  5.1 BACKGROUND............................................................................................................................25
  5.2 APPROACH ....................................................................................................................................25
  5.3 RESULTS.......................................................................................................................................26
     5.3.1 Effect of parameter variation on baseline dynamics ........................................................26
     5.3.2 Impact on Scenarios............................................................................................................30

6 SUMMARY..........................................................................................................................................36

7 CONCLUSIONS AND RECOMMENDATIONS ..............................................................................38

8 REFERENCES.......................................................................................................................................40
1 Introduction

The Gippsland Lakes have suffered considerable water quality degradation due to algal blooms over past decades (Harris et al. (1998)). The Gippsland Lakes Environmental Study (GLES) was a modelling study intended to investigate a series of hypothetical management strategies in the Lakes (Webster et al. (2001)). Modelled scenarios included nutrient and river flow reductions and exchange modifications. A strategy to ameliorate conditions in the Lakes has been developed by the Gippsland Coastal Board (GCB) partly based on the model predictions (Gippsland Lakes Future Directions and Actions Plan (2002)). An important issue that has been raised is the validity and robustness of the model predictions. This project addresses aspects of model uncertainty.

The GLES combined the application of a hydrodynamic model and a biogeochemical model to the Gippsland Lakes. The hydrodynamic model was used to simulate flow, mixing, stratification and salinity throughout the Lakes, whereas the biogeochemical model was used to predict the transport and transformations of nutrients and organic matter and the resultant impacts on primary production including algal growth. These models are at best greatly simplified approximations to reality. There are many uncertainties and possible sources of error in their predictions. Biogeochemical models represent the complicated interactions between many species in real systems by using a relatively small number of functional groups and simple relationships between them. Models represent real systems using coarse spatial grids, whereas real systems exist as a continuum. Other sources of uncertainty are the forcing time series (e.g., nutrient loads), and the representation of the processes as equations. One component of the process representation is the choice of parameters in the equations, mainly rate constants, that are needed. In the biogeochemical model applied to the Gippsland Lakes, there are many processes represented so the specification of a whole suite of such parameters is required. This project treats model uncertainty that arises from uncertainty in the chosen parameter values. Other sources of uncertainty described above are not addressed by this project.

The choice of parameters is constrained by a range of feasible values, usually deduced from other published studies and experiments. For application to the Gippsland Lakes system, the parameters were tuned by calibration against field observations; that is, specific parameter values were chosen so that the model predictions approximately reproduced the observed response of the system (Parslow et al. (2001a)). If observations were sufficient to uniquely define model parameters, there would be no need for a sensitivity analysis. However, there were limited field data with which to calibrate the model, and so alternative sets of parameters can yield an equally acceptable match to the field data. In effect, simulations obtained with these alternative parameter sets are equally legitimate as far as the calibration process goes. The essential question addressed by this project is how might the conclusions drawn from the scenario analysis undertaken in the GLES be affected by parameter uncertainty.

The first task of this project was to identify alternative parameter sets. There are a large number of parameters required by the biogeochemical model and it is not possible to examine the impact of uncertainty in all of them. Rather, we selected parameters from processes that were flagged as being particularly important or uncertain in the GLES modelling work. These processes were: the representation of *Nodularia* (with particular emphasis on temperature and salinity tolerances); the representation of sediment processes (particularly denitrification and phosphorus dynamics); and the amount of
vertical exchange in the water column. Strictly, concern with vertical exchange does not derive from parameter uncertainty, but it is of major importance.

There are many aspects of model behaviour whose sensitivity to parameters we could assess. The original model study, and this sensitivity analysis, have both been undertaken in an applied management context, where stakeholders might use model scenarios to inform management decisions. Accordingly, we have, in consultation with the Gippsland Coastal Board, identified four management scenarios of particular interest, and have assessed the sensitivity of key model indicators (identified in the original study) to variations in model parameters.

This report is structured so that each process is addressed in turn. For each process, we describe the search for alternative calibration parameters and the implications for scenario predictions. The concluding section of this report discusses overall findings and makes specific recommendations.

2 Approach

2.1 Searching for Alternative Parameter Sets

The search for alternative parameter sets is an inverse or inference problem; it effectively requires a search through parameter space for parameter combinations that meet specified criteria. The published literature on this kind of searching problem is extensive and many efficient search techniques have been developed (e.g. Sambridge (1998), Mosegaard K. and Sambridge (2002)). Unfortunately, these approaches are best suited to models with rapid run-times, and with well-defined search criteria.

The search for parameter sets requires two steps: a sampling step which generates parameter sets to test, and a selection step which identifies ‘acceptable’ parameter sets from all those tested.

Ideally, the first step should thoroughly sample the space of possible parameter combinations. Unfortunately, this sampling problem rapidly becomes intractable as the number of parameters increases; this is known as ‘the curse of dimensionality’. For example, if there are 10 parameters to be varied simultaneously, and four values of each parameter are to be tested, a systematic grid search of that space would require $4^{10}$ (just over one million) model runs. The model runtime for the Gippsland Lakes model is approximately 13 minutes, so to conduct one million runs would require roughly 25 years of computing time. For this reason, most of the parameter searches are limited to testing variations in a single parameter at a time, and at most 4 parameters are varied simultaneously. A range of feasible values was identified for each parameter from the results of other studies, and values were drawn from this range. The random number generator in Matlab was used to sample from a uniform distribution over the range for each parameter; that is, all parameters within the range were treated as equally likely.

The second step requires a procedure for comparing model results with the available data from the Lakes. One approach could be to define automated measures of model match to data for making this comparison (e.g. requiring that the median or 95th percentile of model and data time series do not differ by more than a prescribed amount). Such an approach has obvious appeal if an appropriate measure of model-data fit can be found.
In our problem, there are many quantities being modelled (e.g. nutrients, phytoplankton), in many different locations (e.g. sediment or water column, different Lake basins) and varying across time. Consequently, there is no one ideal measure of “model accuracy” that we can use to guide a search algorithm. In addition, there are relatively scarce data against which to compare the model output. These characteristics of the problem make the definition of an appropriate measure of model performance – one that can be automated and used to guide the search through parameter space – very difficult.

Rather than pursue this approach, we chose to visually scrutinise the model results against field observations and manually identify ‘observation consistent’ parameter sets from the multiple runs. This has the advantage of being more cautious and flexible, and reduces the risk of parameter combinations being accepted or rejected unnecessarily due to inflexible automated selection criteria. A disadvantage is that it is more time-consuming, however in our case the model run-time already limits number of model runs to be relatively few (a few thousands). A further disadvantage is that decisions about which parameter sets to include or exclude have an element of subjectivity. The MAFRI and EPA datasets used for this comparison are the same as those used in the original calibration (Parslow et al. (2001a)).

A Matlab graphical user interface (GUI) was written for the purpose of making the visual model-data comparisons. Within the GUI, the time series produced by a set of model runs were viewed simultaneously, with the original baseline time series and field data plotted on the same graph for comparison. For any given indicator, or parameter range, model runs could be identified and removed.

### 2.2 Choice of scenarios

The GLES modelling work conducted simulations for eighteen different scenarios. Four scenarios have been selected for the sensitivity analysis, and two of these have been newly constructed after consultation with the Gippsland Coastal Board.

<table>
<thead>
<tr>
<th>Code</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>B</td>
<td>Baseline calibration run</td>
</tr>
<tr>
<td>SE</td>
<td>Exchanges from hydrodynamic model with second entrance at Ocean Grange</td>
</tr>
<tr>
<td>R50</td>
<td>50% reduction in all loads</td>
</tr>
<tr>
<td>EWNP</td>
<td>'Best practice' scenario, with different N and P reductions in east and west catchments. Western catchments: decrease P by 12% and N by 8% Eastern catchments: decrease P by 5% and N by 4%</td>
</tr>
<tr>
<td>RN20</td>
<td>20% reduction in all nitrogen loads</td>
</tr>
</tbody>
</table>

Scenario SE models the effect of a second entrance at Ocean Grange. It is one of the GLES scenarios, and a more detailed description can be found in the GLES reports (Parslow et al. (2001b) and Webster et al. (2001)). In this scenario, the physical exchanges in the box model were derived from a modified hydrodynamic model.
incorporating the second entrance. The second entrance has the effect of increasing salinity values in Lakes Victoria and King, with a smaller increase in salinity in Lake Wellington and approximately doubles the rate at which Lakes King and Victoria exchange with Bass Strait. In the original GLES work, the beneficial effects of the entrance were mainly limited to Lakes Victoria and King, where median and 95th percentile chlorophyll concentrations were approximately halved and 5th percentile bottom water oxygen levels were increased by a factor of 3 to 4. The benefits were due to two main effects: the second entrance increased nutrient removal due to flushing and export rates, and also increased the internal nutrient sinks by increasing oxygen levels in bottom waters. This scenario did not reduce the bloom peaks which immediately follow flow events, however it had a disproportionately large impact on *Nodularia* blooms. This appears to be due to the increased salinity levels, as *Nodularia* growth is assumed to be suppressed above a certain salinity level. Webster et al. (2001) warned that this prediction must be treated with caution, as relatively little was known about the salinity tolerances of *Nodularia* strains in the Gippsland Lakes system.

The R50 scenario is one of the GLES load scenarios and it models the effect of reducing all catchment N and P loads by 50%. It represents an ambitious load reduction target, and one that is unlikely to be achieved in the near future. The GLES modelling work suggested that load reductions of this order would be required to produce a significant reduction in periods of bottom water hypoxia. The TP and TN loads were both reduced by the same proportion, however the model predicted stronger N-limitation. This effect was due to the fact that denitrification efficiency was increased relatively more than the efficiency of P-sequestration. It was noted that a shift towards N-limitation might be expected to favour N-fixing *Nodularia*, but that bloom sizes were still limited by the available P. Some effects of this scenario were comparable to the SE scenario, with an important difference: the R50 scenario effectively reduced bloom peaks that directly followed flood events, where the SE scenario had little impact on these peaks.

The EWNP scenario is a newly constructed scenario, and allows differential reductions in Eastern and Western catchments and differential reductions in N and P. According to the Gippsland Coastal Board, the adoption of ‘best practice’ nutrient reduction strategies in the catchment is likely to produce higher reductions in western catchments, and phosphorus reductions are expected to be slightly higher than nitrogen reductions. Specifically, western catchments are expected to yield a 12% reduction in P loads and 8% reduction in N loads. Eastern catchments are expected to yield 5% and 4% reductions in P and N respectively.

The RN20 scenario is a second newly constructed scenario. It reduces catchment-wide nitrogen loads by 20%. The GLES modelling work suggested that a shift towards stronger N-limitation in the lakes might be expected to favour N-fixing cyanobacteria such as *Nodularia*. This scenario can be used to investigate this situation, as well as testing whether the results from a scenario with preferential N or P reduction would be sensitive to the choice of sediment parameters. It should be noted that in reducing all N fractions in loads by 20%, there were associated small reductions in TP (3% in Western catchments and 7% in Eastern catchments). This is because a significant portion of the N load is labile detrital material, which is assumed to contain N and P in a fixed ratio according to Atkinson stoichiometry (N:P of 30:1, as discussed in the original GLES modelling work, Parslow et al. (2001a)).

A concise summary of the impacts of the four scenarios is shown in Figure 1. As expected, the two new scenarios (EWNP and RN20) have relatively little impact on 95th percentile chlorophyll levels and 5th percentile oxygen levels, relative to the baseline
levels. The predictions for *Nodularia* 95\textsuperscript{th} percentiles are that the EWP scenario would reduce *Nodularia* levels slightly, relative to baseline, while the RN20 scenario would increase *Nodularia* levels slightly. This latter result is consistent with suggestions made in the GLES work that stronger N-limitation may favour *Nodularia* growth. Results suggest a significant increase in dinoflagellate 95\textsuperscript{th} percentiles under the RN20 scenario in Lake King, with little impact on levels in Lake Victoria. Increases relative to baseline are suggested for Lake Wellington, but model dinoflagellate levels are very low in Lake Wellington, and the large increase relative to baseline still corresponds to very low dinoflagellate levels in Lake Wellington.

![Graphs showing impacts of scenarios relative to baseline](image)

**Figure 1** Impacts of scenarios relative to baseline for four indicators (95\textsuperscript{th} surface chlorophyll, surface *Nodularia* and surface dinoflagellates, and 5\textsuperscript{th} bottom oxygen). Red bars are Lake Wellington, green bars are central Lake Victoria and blue bars are south Lake King.

### 2.3 Assessing the scenarios

When assessing the scenarios the motivating question is: are the conclusions drawn from the GLES modelling work sensitive to the choice of calibration parameters?

In the GLES modelling work, scenario assessment compared percentiles for a selection of indicators. Nine water column and six sediment indicators were identified as being most useful for assessing the impact of each scenario (Parslow et al. (2001b)). In this report we also refer to indicator percentiles, but our interest now is the sensitivity of these percentiles to different parameter sets.
Figure 2 (a) Examples of original baseline time series (blue line) and set of alternative baseline time series from different parameter sets (grey lines). Units are mg m$^{-3}$. (b) Example absolute sensitivities for chlorophyll 50th, 80th and 95th percentiles; (c) Example ratio sensitivities for chlorophyll 50th, 80th and 95th percentiles. In (b) and (c) horizontal black lines represent the original GLES values.

For each indicator, there is a set of alternative baseline time series (eg. Figure 2(a)) and a set of alternative scenario time series for each scenario. A set of percentiles can be calculated for each of these time series. In this report we will refer to both absolute and ratio sensitivities for these percentiles. The absolute sensitivity for a given indicator percentile is the range encompassing all the values for that indicator percentile (eg. Figure 2(b)). Most of the conclusions drawn from the scenario analysis in the GLES were presented as a predicted change in an indicator percentile relative to baseline. The question being addressed in this work is: under a different parameter set, would this prediction of scenario result relative to the new baseline be substantially changed? A particular indicator may show considerable baseline sensitivity to parameter choice,
but the predicted scenario effects relative to their (altered) baselines may be quite insensitive to parameter choice, in which case it could be argued that the management conclusions about change relative to baseline are quite robust. An example of this ratio sensitivity is given in Figure 2(c). Note that in this figure, the ratio sensitivity for the baseline scenario (i.e. the 'B' column in Figure 2(c)) presents the new baseline percentiles relative to the original baseline scenarios, and the ratio sensitivity for the other scenarios is the new scenario percentiles relative to the new baseline percentiles. Where the 5th and 20th percentiles are close to zero, ratio sensitivities can be very large and it is more useful to refer to absolute rather than ratio sensitivities. This is particularly true for bottom oxygen levels.

It should be stressed that this approach is likely to over-state the sensitivity as we have shown only the range for any indicator percentile and have not tried to characterise its distribution (eg. to quantify the proportion of runs in each part of the range).

3 Vertical exchange

3.1 Background
The biogeochemical model divides the Lakes into 8 boxes, and a ninth box represents the ocean (Figure 3). Each box has two water-column layers, with the exception of Lake Wellington and the northern-most box in Lake King (boxes 6 and 8 in Figure 3). Time-varying horizontal and vertical exchanges between all the boxes were determined from the hydrodynamic model developed for the GLES (Walker and Andrewartha (2000)). It has been suggested that the vertical exchanges in the box model may be overestimated in Lake Victoria and Lake King (Webster et al. (2001)). There are indications that that the model underestimates the strength of thermal stratification (suggesting more vigorous vertical mixing than in reality), but the comparison with measurements is certainly not conclusive.

![Figure 3 Box model geometry.](image-url)
3.2  **Approach**

Here we alter vertical diffusive exchanges and assess the effect on the biogeochemical simulations. Advective exchanges also occur but these can’t be altered without re-running the hydrodynamic model.

Consider two layers, layer A and layer B, within a single box of the model (Figure 4). The advective exchange is the net exchange between the two layers (shown here as a surface layer to bottom layer exchange). The diffusive exchange can be thought of as the two-way exchange; it transports volume equally in both directions. The diffusive exchange component can be increased or decreased without affecting the net exchange between the two layers. Altering the advective exchange is more problematic as any changes need to maintain volume conservation requirements (reducing or increasing advective exchange between layers in one box requires compensating changes in other boxes).

![Figure 4](image)

**Figure 4** Vertical exchanges between two layers in one box of the biogeochemical model.

To test sensitivity to vertical exchange, the baseline scenario vertical diffusive exchanges were decreased by 50% and 90% as the GLES modelling work suggested that exchanges were over-estimated in the box model. An example of how vertical exchanges in south Lake King have been modified is shown in Figure 5. Blue lines show the original exchanges and green lines show the modified exchanges (in this case, a reduction in diffusive exchange by 50%). The diffusive exchanges are all reduced by 50%, and the advective exchanges remain unchanged. Table 2 shows the actual percentage changes in upward and downward exchanges for given changes in diffusive exchange.

**Table 2** Percentage changes to total upward and total downward exchanges for a given percentage change in the diffusive exchange. (Upward exchange is the upper value and downward exchange is the lower value in each table cell.)

<table>
<thead>
<tr>
<th>% change to diffusive exchange</th>
<th>South King</th>
<th>North King</th>
<th>East Victoria</th>
<th>Central Victoria</th>
<th>West Victoria</th>
</tr>
</thead>
<tbody>
<tr>
<td>50%</td>
<td>42.0%</td>
<td>10.8%</td>
<td>16.9%</td>
<td>4.8%</td>
<td>8.2%</td>
</tr>
<tr>
<td></td>
<td>14.0%</td>
<td>44.0%</td>
<td>41.3%</td>
<td>38.7%</td>
<td>16.2%</td>
</tr>
<tr>
<td>90%</td>
<td>75.7%</td>
<td>19.4%</td>
<td>30.4%</td>
<td>8.6%</td>
<td>14.7%</td>
</tr>
<tr>
<td></td>
<td>25.2%</td>
<td>79.1%</td>
<td>74.3%</td>
<td>69.6%</td>
<td>29.1%</td>
</tr>
</tbody>
</table>
Figure 5 Sample vertical exchange file time series for south Lake King.

3.3 Results

3.3.1 Effect of parameter variation on baseline dynamics

All the alternative exchanges produced surface and bottom water salinity time series that are virtually unchanged from the original baseline exchanges. Surface chlorophyll levels are relatively insensitive to the changes to vertical diffusive exchange (eg. Figure 6).

Figure 6 Baseline chlorophyll time series for south Lake King. The dark blue line is the original baseline time series, and the shaded blue region is the range covered by the model runs with all alternative diffusive exchange values. Square and asterisk markers are EPA and MAFRI data respectively. Units are mg m$^{-3}$. 
Larger differences are observed in bottom oxygen levels (Figure 7) and denitrification efficiency (Figure 8), particularly in the eastern boxes of the model (Lake King and eastern Lake Victoria). Of particular interest is the predicted occurrence of a more extended period of low oxygen in Lake Victoria in late 1998 and early 1999 (Figure 7) which corresponds to periods of lower denitrification efficiency.

Figure 7 Bottom water oxygen levels for east Lake Victoria (left) and south Lake King (right). The dark blue line is the original baseline time series, and the shaded blue region is the range covered by the model runs with the alternative diffusive exchange values. Units are mg O m$^{-3}$.

Figure 8 Denitrification efficiency (%) in east Lake Victoria and south Lake King sediments. The dark blue line is the original baseline time series, and the shaded blue region is the range covered by the model runs with the alternative diffusive exchange values.
Dinoflagellate concentrations were found to be sensitive to vertical exchange (Figure 8). *Nodularia* levels remained virtually unchanged across changes to exchange values.

**Figure 9** Surface dinoflagellate levels for east Lake Victoria and south Lake King. The dark blue line is the original baseline time series, and the shaded blue region is the range covered by the model runs with all alternative diffusive exchange values. Units are mg N m$^{-3}$.

There are follow-on consequences for bottom and surface water ammonia levels: periods of P-limitation are more pronounced as the decreased efficiency of the sediment nitrogen sink allows more ammonia in the water column (Figure 10(a)); an earlier dinoflagellate bloom produces ammonia drawdown in the bottom waters at times corresponding to dinoflagellate blooms (Figure 10(b)).

**Figure 10** Surface (left) and bottom water (right) ammonia levels in south Lake King. Units are mg N m$^{-3}$.
3.3.2 Impact on Scenarios
In this section results are only shown for Lakes Victoria and King as Wellington is relatively unaffected by changes to vertical exchange (it is only represented by one vertical layer in the model). The 50% and 90%-reduction exchange files were applied directly to the three load scenarios. The SE scenario requires its own exchange file, so 50% and 90% reductions were applied to the vertical diffusive exchanges in the SE exchange file.

3.3.2.1 Chlorophyll
Baseline chlorophyll levels were relatively insensitive to changes in diffusive exchange (Figure 6) and this remained true for the scenario predictions (Figure 11).

![Figure 11](image-url) Absolute (left) and ratio (right) sensitivities for surface chlorophyll in south Lake King. Absolute units are mg m$^{-3}$.

3.3.2.2 Nodularia and dinoflagellates
The dinoflagellate response differs from the chlorophyll response. Here a relatively large sensitivity is evident in the baseline 95th percentiles, which remains high for the EWNP and RN20 scenarios but is reduced in the SE and R50 scenarios.

The Nodularia response is similar, with the exception that ratio sensitivity is increased for the EWNP and RN20 scenarios (and the R50 scenario to a lesser extent). There is a tendency for higher 95th percentiles in DIP for the EWNP and RN20 scenarios, which is indicative of periods of stronger N-limitation than in the baseline scenario. Stronger N-limitation can favour the growth of Nodularia because of its nitrogen fixing ability. It’s not clear why the EWNP scenario can produce periods of stronger N-limitation, as P-loads are reduced more than N-loads in this scenario.
### 3.3.2.3 Bottom oxygen

A large effect was seen in baseline bottom oxygen levels (Figure 7), and this remained true for scenario predictions, although with an important variation (Figure 13). The sensitivity to diffusive exchange in the baseline time series was similar to the sensitivity for the EWNP and RN20 scenarios, but the sensitivity of the 5th percentile was considerably larger in the SE and R50 scenarios. In particular, for a 90% reduction in diffusive exchange, the bottom oxygen 5th percentiles for the SE and R50 scenarios drop by 1570 mg O m\(^{-3}\) and 1610 mg O m\(^{-3}\) respectively in east Lake Victoria. While this is a significant quantitative difference, the qualitative result remains unchanged from the original GLES work, in that the SE and R50 scenarios still predict a large improvement in bottom oxygen levels relative to baseline.

![Figure 12](image12.png)

**Figure 12** Ratio sensitivities for surface dinoflagellate (left) and *Nodularia* (right) levels in south Lake King.

![Figure 13](image13.png)

**Figure 13** Absolute sensitivities for 5th and 20th percentiles for bottom oxygen levels in east Lake Victoria. Units are mg O m\(^{-3}\).
The uncertainty in bottom oxygen levels has consequences for denitrification (Figure 14). While denitrification levels are still lifted relative to baseline in the SE and R50 scenarios, the 5th percentile dropped from 16% to 4% for the SE scenario and from 24% to 9% in the R50 scenario in the sediments of Lake Victoria. Denitrification efficiency percentiles for the EWNP and RN20 scenarios remain similar to the baseline percentiles. Again, although the quantitative results are significantly different, they remain qualitatively consistent with the original conclusions that denitrification efficiency would be increased by the SE and R50 scenarios.

Figure 14 Absolute sensitivities for sediment denitrification efficiency (%) in east Lake Victoria.

4 Nodularia

4.1 Background
The Gippsland Lakes are subject to recurring blooms of *Nodularia*, a toxic blue-green cyanobacterium. Their representation in the model is subject to significant uncertainty, and it was one of the processes flagged in the GLES work as requiring further research and model development. In particular, the model assumed temperature and salinity limits on *Nodularia* growth that were derived from studies in other locations. During the original model development there was limited information specific to *Nodularia* strains found in the Gippsland Lakes, and yet in the calibration process the model behaviour was observed to be sensitive to the choice of these temperature and salinity tolerances. *Nodularia* germinates each summer from akinetes in the sediments. This process is poorly understood and is represented very simply in the model; a small population of *Nodularia* is seeded in Lake King in January each year. Very little is known about *Nodularia* mortality, and again this is represented very simply in the model by empirical loss rates (this is in contrast to other phytoplankton species in the model, which are explicitly grazed by zooplankton). When calibrating the model it was found that the model tended to produce large blooms in Lake Wellington, which are not observed in reality. The prescription that *Nodularia* seeds in January and the selection of mortality rates were crucial parameters in preventing Lake Wellington blooms.
4.2 Approach

*Nodularia* was represented in the model in the same manner as the other phytoplankton species, using the Chemical Reaction (CR) scheme of Baird and Emsley (1999). In the CR scheme, phytoplankton size and surrounding nutrient concentrations are used to calculate nutrient uptake rates, which are derived from the physical processes of diffusive flux to the cell membrane. For *Nodularia*, the CR scheme was modified to represent nitrogen fixation. The specified maximum growth rate of *Nodularia* accounted for temperature and salinity tolerances; growth rates were prescribed to fall off exponentially for temperatures in the vicinity of TempgrowBG (set to 19°C) and for salinities in the vicinity of SaltgrowBG (set to 25). A particular focus of the sensitivity analysis was to investigate these tolerances more closely.

Holland and Beardall (2004) described experiments on *Nodularia* strains from the Gippsland Lakes. These experiments confirmed that both *Nodularia* akinete germination and growth rates are subject to salinity controls (experiments did not extend to investigating temperature controls). Of particular relevance is the observation that germination and growth were inhibited at very low salinities (Figure 15 and Figure 16). Low germination rates at low salinities provide some justification for the representation of *Nodularia* germination in the model, where *Nodularia* populations are seeded in January each year in Lake King only.

![Figure 15](image-url)

*Figure 15* Germinating *Nodularia* akinetes per mL of sediment over a range of salinities (0-45gL⁻¹) in A) N-free BG-11and B) N-replete (NH₄) BG-11. Error bars equal 1 standard error. Taken from Holland and Beardall (2004).
The experimental results reported by Holland and Beardall (2004) showed very little germination and growth in freshwater, whereas experiments at a NaCl concentration of 10 g/L yielded the highest *Nodularia* growth rate. This suggests there is germination and growth inhibition at very low salinities, and the model has been modified to include another salinity threshold, SaltgrowBG_min, and salinity drops exponentially in the vicinity of this threshold. The *Nodularia* experiments suggest that this salinity value ought to be below 10. It should be noted that *Nodularia* growth in the model does not follow the pattern in Holland and Beardall (2004). In particular, maximum *Nodularia* growth rates in the model extend between SaltgrowBG_min and SaltgrowBG, but according to the experimental results there ought to be a peak in growth at a salinity of approximately 10, and lower growth rate at higher salinities. The current version is likely to over-estimate *Nodularia* growth, and so is a more cautious approach. The modification to minimise growth in water of low salinity was included to weaken the model’s tendency to produce *Nodularia* blooms in Lake Wellington, which are not observed.
The parameters listed in Table 3 were tested in the search for alternative *Nodularia* parameter sets. Each parameter was tested in isolation, however most of the runs were devoted to testing combinations of some or all of the following four parameters: SaltgrowBG, SaltgrowBG_min, TempgrowBG and BGumax.

**Table 3** Parameters varied in search of alternative *Nodularia* parameter sets. Parameters coloured in blue were varied simultaneously, and remaining parameters were varied individually.

<table>
<thead>
<tr>
<th>Parameter name</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>SaltgrowBG</td>
<td>Upper salinity tolerance: <em>Nodularia</em> growth rate decreases at salinities above this threshold</td>
</tr>
<tr>
<td>SaltgrowBG_min</td>
<td>Lower salinity tolerance: <em>Nodularia</em> growth rate decreases at salinities below this threshold</td>
</tr>
<tr>
<td>TempgrowBG</td>
<td>Temperature tolerance: <em>Nodularia</em> growth rate decreases at temperatures below this threshold</td>
</tr>
<tr>
<td>BGumax</td>
<td>Maximum <em>Nodularia</em> growth rate</td>
</tr>
<tr>
<td>ml_BG_T15</td>
<td>Linear <em>Nodularia</em> mortality rate</td>
</tr>
<tr>
<td>BGrad</td>
<td><em>Nodularia</em> cell radius</td>
</tr>
<tr>
<td>Nodseed</td>
<td>A forcing time-series prescribing <em>Nodularia</em> germination in Lake King in January. The magnitude (not the timing) of the seed population in this time series was altered.</td>
</tr>
</tbody>
</table>

### 4.3 Results

#### 4.3.1 Effect of parameter variation on baseline dynamics

The model has a tendency to produce large *Nodularia* blooms in Lake Wellington which are not observed in reality. When varying *Nodularia* parameters and searching for alternative calibration sets, the most important selection criterion was that the *Nodularia* concentrations in Lake Wellington must not exceed those considered acceptable in the GLES modelling work. A second selection requirement was that blooms do occur in Lake King (otherwise runs could be selected that eliminate *Nodularia* from the Lakes altogether). Although these were the two main selection rules, the chlorophyll, nutrients and dinoflagellate time series were also checked to ensure that their levels were acceptable.

As discussed in Section 2.1, varying parameters in combination can allow a greater range of ‘acceptable’ values. This is particularly true for TempgrowBG and BGumax, and a plot of the two against each other shows why (Figure 17). The parameter BGumax is the maximum growth rate for *Nodularia* and it is multiplied by a factor derived from TempgrowBG in the model to create a temperature-affected growth rate that is used by the CR scheme when calculating *Nodularia* growth. A particular temperature-affected growth rate is possible by many TempgrowBG and BGumax combinations. Without solid experimental evidence to constrain one of them, a large range of both BGumax and TempgrowBG are possible so long as they lie on the curve shown in Figure 17. By contrast, no such relationship is seen between BGumax and SaltgrowBG. A factor derived from SaltgrowBG is also multiplied with BGumax, however a similar pattern is not observed because in general the salinities are too low for the salinity tolerance to take effect (with the exception of the first half of 1998, where high salinities suppress blooms in Lake King in the model).
**Figure 17** The relationship between accepted TempgrowBG and BGumax parameter values. Grey circles show all the combinations tested, and the blue crosses show the parameters that remained after rejecting unacceptable results (eg. removing high blooms in Lake Wellington, but requiring blooms to exist in Lake King). The red asterisk marks the original calibration parameter values.

**Figure 18** Modelled Lake King (a and b) and Lake Wellington (c and d) *Nodularia* and chlorophyll time series, using the modified *Nodularia* representation. The blue solid lines show the original model time series, and the blue shaded regions show the range produced by alternative *Nodularia* parameter sets, where SaltgrowBG, SaltgrowBG_min, TempgrowBG and BGumax are varied together.
Table 4 Sampled and accepted *Nodularia* parameter ranges. The accepted ranges are larger than ranges produced by one-by-one parameter variations. 1000 combinations of these parameters were tested, and 53 were accepted for testing with the scenarios. The original calibration values are given in parentheses next to each parameter, with the exception of SaltgrowBG_min, which didn’t exist in the original model formulation. Note that negative values of SaltgrowBG_min represent a case where *Nodularia* growth is reduced only slightly in freshwater; this case is little different from the unmodified model representation.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Range sampled</th>
<th>Range accepted</th>
</tr>
</thead>
<tbody>
<tr>
<td>BGumax (0.4 d⁻¹)</td>
<td>0.2 - 0.6</td>
<td>0.23 - 0.60</td>
</tr>
<tr>
<td>TempgrowBG (19.0 °C)</td>
<td>16 – 21</td>
<td>16.8 - 20.1</td>
</tr>
<tr>
<td>SaltgrowBG (25)</td>
<td>22 – 30</td>
<td>22.1 - 28.3</td>
</tr>
<tr>
<td>SaltgrowBG_min (N/A)</td>
<td>-1 – 6</td>
<td>-1 – 6</td>
</tr>
</tbody>
</table>

Figure 18 shows the range of accepted *Nodularia* and chlorophyll time series and Table 4 shows the range of parameters tested and accepted.

The aim of modifying the model to include SaltgrowBG_min was to seek more parameter combinations that allow comparable *Nodularia* levels in Lake King while effectively eliminating them in Lake Wellington in a way that was not possible using the original model. This parameter set will be referred to as the ‘best’ *Nodularia* parameter set (Figure 19).

Figure 19 Modelled Lake King (a) and Lake Wellington (b) *Nodularia* time series, using the modified *Nodularia* equation and ‘best’ *Nodularia* parameter set. The blue solid lines show the original model time series, and the blue shaded regions show the range produced by alternative *Nodularia* parameter sets.

In the GLES modelling work, the second entrance raised salinity levels in Lake King. These were enough to exceed SaltgrowBG, and so *Nodularia* blooms were suppressed in Lake King because of this salinity inhibition. Holland and Beardall (2004) suggested that there is reduced growth at higher salinities, but it is not eliminated altogether. In the model, *Nodularia* growth drops off exponentially above SaltgrowBG, whereas the
experimental data seems to suggest there may be a long tail where growth still occurs, albeit at a reduced rate. Given this information, we sought out parameter sets with high SaltgrowBG values (exceeding 27, where the original value was 25) with the aim of testing impacts on *Nodularia* predictions for the SE scenario.

The remaining *Nodularia* parameters tested were the cell radius (BGrad), the linear mortality rate (mL_BG_T15) and the seeding rate in Lake King each year (nodseed). These parameters were varied in a one-by-one manner.

BGrad, the cell radius for *Nodularia*, was varied between 18 and 24 µm (the baseline value was 20 µm). All values were found to be acceptable, and only the largest value was propagated through for testing with the scenarios. For BGrad = 24 µm, there is a slight reduction in *Nodularia* in Lake Wellington, with relatively little effect on the magnitude of Lake King blooms.

Only a small variation in mL_BG_T15 was found to be acceptable: the baseline value was 0.1, and an alternative value of 0.11 was accepted after testing a range from 0.05 to 0.15. Values higher than 0.11 reduce blooms in Lake King too much, and values below 0.1 produce large blooms in Lake Wellington.

Germination is represented in the model by introducing a seed population into south Lake King at a rate of 200 mg s\(^{-1}\) in January. An alternative of 100 mg s\(^{-1}\) was accepted after testing seed rates between 100 and 400 mg s\(^{-1}\). This value was found to produce lower bloom magnitudes in Lake Wellington while retaining relatively unchanged bloom magnitudes in Lake King.

Aside from the effect on *Nodularia* and chlorophyll levels, changes to *Nodularia* parameters affect organic N levels in the system (because *Nodularia* fixes nitrogen). There are also interactions with dinoflagellates and DIP (Figure 20). In particular, a slight reduction in *Nodularia* growth can trigger higher surface DIP levels, an indication of stronger N-limitation and suggests that factors other than P availability are limiting *Nodularia* growth in these cases.

![Figure 20 Surface DIP (left, units mg P m\(^{-3}\)) and dinoflagellate levels (right, units mg N m\(^{-3}\)) in south Lake King. Blue shaded regions represent range of results from the combination parameter set in Table 4.](image-url)
4.3.2 Impact on scenarios

4.3.2.1 Chlorophyll

A large absolute sensitivity in baseline chlorophyll 95th percentiles corresponded to similar absolute sensitivities in the EWNp and RN20 scenarios, but low absolute sensitivities in the SE and R50 scenarios (Figure 21).

![Figure 21](image)

**Figure 21** Absolute (left, units mg m\(^{-3}\)) and ratio (right) sensitivities for surface chlorophyll in south Lake King for the parameter ranges given in Table 4.

4.3.2.2 Nodularia and dinoflagellates

Varying salinity and temperature tolerances yielded a relatively large sensitivity in the 95th percentiles for baseline *Nodularia* levels. The absolute sensitivity increased for the RN20 scenario, remained similar for the EWNp scenario and reduced for the SE and R50 scenarios (Figure 22). This pattern parallels the sensitivity in peak DIP levels in the scenarios. An increase in DIP peaks suggests stronger N-limitation, which can favour *Nodularia* growth.

The ratio sensitivities are largest for the SE scenario. These large values are for the parameter sets with the highest salinity tolerances (SaltgrowBG > 27). When these parameter sets are applied, the SE scenario still predicts a substantial reduction in *Nodularia* levels in Lake King, but the relative-to-baseline change is not as large as the GLES result (relative to baseline ratios of 0.46 and 0.21, compared to the original GLES result of 0.09 in south Lake King). This confirms concerns raised in the GLES work that *Nodularia* predictions in for the SE scenario may be particularly sensitive to the choice of salinity tolerance.
The dinoflagellate response followed a similar, but more exaggerated pattern (Figure 23). The baseline absolute sensitivity in 95th percentiles was small, and the absolute sensitivity was reduced for the SE and R50 scenarios. Both the range and magnitude of the 95th percentiles for the RN20 scenario were larger than the baseline; one of the few cases identified where the baseline absolute sensitivity is substantially smaller than a scenario absolute sensitivity. It is interesting that this only occurs for a relatively small incremental change in loads, but where N and P have been reduced differentially. This suggests that results are sensitive to the N:P ratio of incoming loads, and demonstrates that the dinoflagellate response can be sensitive to the choice of *Nodularia* parameters under these circumstances. These interactions between dinoflagellates and *Nodularia*, and their sensitivity to these parameters suggest that more process work is needed to better understand the phytoplankton autecology.
Variations to the values of BGrad, mL_BG_T15 and *Nodularia* seeding rate share a common pattern: the relatively small differences in baseline percentiles were retained for RN20 and EWNP scenarios, and the ranges were considerably smaller in the R50 and SE scenarios.

Figure 24 shows that by applying the ‘best’ *Nodularia* parameter set to the scenarios it was possible to eliminate blooms from Lake Wellington without substantially altering the *Nodularia* predictions in Lake King. More work could be done here to investigate interactions with the modified version of the model and *Nodularia* mortality parameters; it is possible that the mortality could be weakened so allowing the possibility of larger blooms in Lake King.

![Figure 24 Ratio sensitivities for *Nodularia* 95th percentiles in Lake Wellington (left) and south Lake King (right), using the ‘best’ *Nodularia* parameter set](image)

4.3.2.3 **Bottom oxygen**

Altering *Nodularia* parameters had little effect on bottom oxygen 5th percentiles (Figure 25).

![Figure 25 Bottom oxygen 5th percentiles in central Lake Victoria. Units are mg O m⁻³.](image)


5 Sediments

5.1 Background

Sediments take up and release nutrients in the Gippsland Lakes, and the fate of nitrogen and phosphorus entering the Lakes is strongly affected by sediment processes. Two sediment-related processes are of particular interest: phosphorus adsorption and desorption in the water column and sediments; and nitrification and denitrification. Inorganic phosphorus adsors to particles in the water column and sediments. The adsorption-desorption dynamics are important for regulating levels of inorganic phosphorus available for phytoplankton growth. An important nitrogen sink in the Lakes is the sequential nitrification of ammonia into nitrate, and then denitrification of nitrate into nitrogen gas (which escapes to the atmosphere). The strength of the denitrification sink is important for regulating levels of nitrogen available for phytoplankton growth. The GLES modelling work found that the model predictions were sensitive to sediment denitrification efficiency, and it strongly influenced the balance between N and P limitation in the Lakes. The physical exchange between water column and sediments affects the rates at which oxygen and nutrients exchange with the sediment, so affects both phosphorus and denitrification dynamics.

5.2 Approach

The model represents the adsorption and desorption of dissolved inorganic phosphorus (DIP) in the water column and the sediments. Webster and Grace (2001) recommended a particular functional form, a Freundlich isotherm, after conducting P-adsorption experiments. The Freundlich isotherm assumes that inorganic phosphorus is divided between DIP and particulate inorganic phosphorus (PIP) according to the following relationship:

\[
\frac{\text{PIP}}{\text{TSS}} = A(D\text{IP})^B
\]

where TSS is the total suspended solids and A and B are parameters. From their experiments Webster and Grace (2001) concluded that A should take the value of 74 m$^3$ kg$^{-1}$ in the sediments and 107 m$^3$ kg$^{-1}$ in the water column, but noted a wide range of possible values. They recommended that B should be approximately 0.34. Within the model the parameters corresponding to A are P_ads_coeff_wc for the water column, and P_ads_coeff_sed for the sediment. The parameter corresponding to B is P_ads_exp.

Nitrification and denitrification are represented in the model by assuming that all inorganic nitrogen produced by benthic respiration is converted to ammonia. Pore water ammonia is converted to nitrate at a rate that depends on the ammonia and oxygen concentration (and the rate parameter r_nit_sed). Nitrate is converted to N$_2$ gas at a rate that depends on the nitrate concentration and the oxygen concentration (and the rate parameter r_den).

Physical exchange between sediment pore water and overlying water is governed by two exchange velocities, bi_exchange and bi_exchange_O2. The parameter bi_exchange_O2 sets an upper limit to the physical rate at which oxygen can be supplied to the sediment. The parameter bi_exchange is the physical exchange velocity for other dissolved pore water nutrients, and is set to be much lower than bi_exchange_O2.
Table 5 Parameters varied in search of alternative sediment process parameter sets. Parameters coloured alike were varied simultaneously.

<table>
<thead>
<tr>
<th>Parameter name</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>P_abs_coeff_wc</td>
<td>A constant in the phosphorus adsorption-desorption equation for the water column.</td>
</tr>
<tr>
<td>P_abs_coeff_sed</td>
<td>A constant in the phosphorus adsorption-desorption equation for sediments.</td>
</tr>
<tr>
<td>P_abs_exp</td>
<td>Exponent in the phosphorus adsorption-desorption equation.</td>
</tr>
<tr>
<td>r_nit_sed</td>
<td>Rate constant in nitrification equation.</td>
</tr>
<tr>
<td>r_den</td>
<td>Rate constant in denitrification equation.</td>
</tr>
<tr>
<td>bi_exchange</td>
<td>Exchange velocity between sediment and water column</td>
</tr>
<tr>
<td>bi_exchange_O2</td>
<td>Exchange velocity between sediment and water column for oxygen.</td>
</tr>
</tbody>
</table>

5.3 Results

5.3.1 Effect of parameter variation on baseline dynamics

5.3.1.1 Phosphorus isotherms

1000 different combinations of the phosphorus isotherm parameters were tested, with values drawn from ranges given in Table 6. The ranges of accepted parameters are the same as the ranges tested, with the exception of P_ads_coeff_wc where lower values have been rejected. Plotting the accepted parameter values against one another again shows the relationship between them (Figure 26).

Table 6 P isotherm parameter searches. These three parameters were varied in combination, drawing values at random from the ranges specified. 1000 parameter combinations were tested; 195 were accepted and applied to the scenarios.

<table>
<thead>
<tr>
<th>Parameter name</th>
<th>Range tested</th>
<th>Range of accepted values</th>
</tr>
</thead>
<tbody>
<tr>
<td>P_ads_coeff_wc (300 m³kg⁻¹)</td>
<td>55 - 500</td>
<td>187.1 - 498.8</td>
</tr>
<tr>
<td>P_ads_sed (74 m³kg⁻¹)</td>
<td>40 - 100</td>
<td>40.0 - 99.4</td>
</tr>
<tr>
<td>P_ads_exp (0.34)</td>
<td>0.28 - 0.45</td>
<td>0.28 - 0.45</td>
</tr>
</tbody>
</table>

The phosphorus isotherm governs how much DIP there is in the water column and sediments, so finding parameter combinations that yield acceptable DIP levels in the Lakes is important. The original parameter set yields DIP values that are too high in Lake Wellington. Increasing P_ads_coeff_wc can reduce the value of DIP, but then P becomes limiting in Lakes Victoria and King and surface ammonia levels build up. The main selection criteria for observation consistent parameter sets were that DIP levels in Lake Wellington did not exceed those considered acceptable in the original calibration, and that DIP levels in the other lakes remained high enough to avoid large ammonia peaks induced by P-limitation. Observed sediment DIP concentrations are approximately 1000 mg m⁻³ in Lake Wellington, so another check made was to ensure that lower water column DIP concentrations did not correspond with significantly reduced sediment DIP concentrations (Figure 27).
Figure 26 Tested and accepted P isotherm parameters. Tested values are in grey, accepted values in blue.

Figure 27 Baseline and calibration set model surface DIP levels in Lakes Wellington, Victoria and King (units mg P m$^{-3}$).
Some parameter sets that produced periods of P-limitation have been retained, as seen in the ammonia time series from Lakes Victoria and King (Figure 28).

![Figure 28](image)

**Figure 28** Surface ammonia levels for east Lake Victoria (left) and south Lake King (right). Units are mg N m$^{-3}$.

### 5.3.1.2 Nitrification and denitrification parameters

Important indicators to guide the selection of acceptable parameter combinations include phosphate and ammonia levels. The GLES modelling work suggested that if $r_{\text{nit\_sed}}$ is set too low, the Lakes become P-limited and high surface ammonia levels result. If $r_{\text{nit\_sed}}$ is set too high, the Lakes become strongly N-limited and high surface phosphate levels result. Avoiding both extremes was the main selection criterion used to find acceptable denitrification parameters.

**Table 7** Denitrification parameter searches. These two parameters were varied in combination, drawing values at random from the ranges specified. 250 parameter combinations were tested; 101 were accepted and applied to the scenarios. Original calibration values are given in parentheses.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Range of tested values</th>
<th>Range of accepted values</th>
</tr>
</thead>
<tbody>
<tr>
<td>$r_{\text{nit_sed}}$ (0.4 d$^{-1}$)</td>
<td>0.2 - 0.8</td>
<td>0.34 - 0.74</td>
</tr>
<tr>
<td>$r_{\text{den}}$ (5.0 d$^{-1}$)</td>
<td>2 - 6</td>
<td>2.02 - 5.84</td>
</tr>
</tbody>
</table>

After selecting runs according to these criteria, the resulting ranges in denitrification efficiency are relatively small (Figure 30). An interesting observation is that the impact of this range in denitrification efficiency on chlorophyll levels is relatively small, but the composition of the blooms is affected (eg. see the dinoflagellate concentrations in Figure 31). The GLES modelling work suggested that denitrification efficiencies in Lake Wellington were overestimated, but reducing $r_{\text{nit\_sed}}$ had the effect of producing high surface ammonia levels in Lakes Victoria and King. In the search for parameters, we found that a small reduction in Lake Wellington denitrification efficiencies was possible without causing the high ammonia levels in Lakes Victoria and King, so long as $r_{\text{den}}$ was reduced to a much lower level.
Figure 29 Tested and accepted values for denitrification parameters.

Figure 30 Sediment denitrification efficiency (%) in Lake Wellington and south Lake King.

Figure 31 Surface chlorophyll (left, units are mg m$^{-3}$) and dinoflagellates (right, units are mg N m$^{-3}$) in south Lake King.
5.3.1.3 Sediment exchange velocities

Varying the sediment exchange velocity has a large effect on sediment denitrification efficiencies, and so again balance between N and P-limitation is very sensitive to the choice of these parameters. The selection criteria for acceptable sediment exchange velocities were much the same as for the P isotherm and denitrification parameters: avoid high surface ammonia and phosphate levels in Lakes Victoria and King. The tested and accepted parameter ranges are given in Table 8 and plotted in Figure 32.

Table 8 Exchange velocity parameter searches. These two parameters were varied in combination, drawing values at random from the ranges specified. 250 parameter combinations were tested. 37 parameter sets were accepted and applied to the scenarios. Note that the model assumes there are 100 animals/m².

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Range of tested values</th>
<th>Range of accepted values</th>
</tr>
</thead>
<tbody>
<tr>
<td>bi_exchange (5×10⁻¹⁰ m s⁻¹/(animals m⁻²))</td>
<td>2×10⁻¹⁰ - 10×10⁻¹⁰</td>
<td>4.3×10⁻¹⁰ - 7.2×10⁻¹⁰</td>
</tr>
<tr>
<td>bi_exchange_O₂ (5×10⁻⁸ m s⁻¹/(animals m⁻²))</td>
<td>2×10⁻⁸ - 10×10⁻⁸</td>
<td>4.0×10⁻⁸ - 9.0×10⁻⁸</td>
</tr>
</tbody>
</table>

The patterns observed are similar to those discussed in the previous section: the range in denitrification efficiency is similar, the effect on chlorophyll is small, but again the composition of blooms is sensitive to these parameters, with earlier dinoflagellate blooms possible. The accepted calibration sets include sets that produce more pronounced periods of P-limitation and sets that produce stronger N-limitation.

5.3.2 Impact on scenarios

5.3.2.1 Phosphorus isotherms

5.3.2.1.1 Chlorophyll

Varying phosphorus isotherm parameters had a larger effect on Lake Victoria and Lake King chlorophyll levels than on levels in Lake Wellington (Figure 33). The absolute chlorophyll sensitivity is greatest for the baseline, EWNP and RN20 scenarios in Lake King.
5.3.2.1.2 *Nodularia* and dinoflagellates

Scenario RN20 showed the largest ratio sensitivity for *Nodularia* and dinoflagellates, demonstrating that the sensitivity of bloom composition to phosphorous adsorption processes is particularly acute when the N:P ratio of loads is altered significantly.

A related observation is that median surface DIP levels are substantially elevated in the SE scenario for Lakes King and Victoria (Figure 35), and to a lesser extent in the R50 scenario. This is indicative of stronger N-limitation, due to a larger increase in denitrification efficiency than phosphorus removal efficiency. Given the availability of DIP in the SE scenario, it suggests that *Nodularia* levels are not limited by P availability but by other mortality and growth constraints (eg. the salinity tolerance and the prescribed mortality rates).
5.3.2.1.3 Bottom oxygen
Absolute sensitivities for bottom oxygen 5\textsuperscript{th} percentiles are highest for the SE and R50 scenarios. It suggests that sensitivity to phosphorus dynamics can be sufficient to affect bottom oxygen levels indirectly through primary production (lowest bottom oxygen levels correspond with chlorophyll peaks), although the effect is not strong.

5.3.2.2 Nitrification and denitrification parameters
5.3.2.2.1 Chlorophyll
Chlorophyll percentiles are relatively insensitive to changes in nitrification and denitrification parameters (Figure 37).
5.3.2.2.2 Nodularia and dinoflagellates
Dinoflagellate and *Nodularia* sensitivities to nitrification and denitrification parameters are highest for the EWNP and RN20 scenarios, and are small for the SE and R50 scenarios.

![Figure 37](image)

Figure 37 Ratio sensitivity for surface chlorophyll levels in Lake Wellington (left) and south Lake King (right).

5.3.2.2.3 Bottom Oxygen
Altering nitrification and denitrification parameters had little effect on bottom oxygen 5th percentiles (Figure 39).

![Figure 38](image)

Figure 38 Ratio sensitivities for surface dinoflagellate (left) and *Nodularia* (right) 95th percentiles for south Lake King.
Figure 39 Bottom oxygen 5th percentiles in central Lake Victoria. Units are mg O m$^{-3}$.

5.3.2.3 Sediment exchange velocities

5.3.2.3.1 Chlorophyll
Chlorophyll percentiles are relatively insensitive to changes in sediment exchange velocity parameters (Figure 40).

Figure 40 Ratio sensitivity for surface chlorophyll percentiles in Lake Wellington (left) and south Lake King (right).

5.3.2.3.2 Nodularia and dinoflagellates
A large baseline sensitivity in dinoflagellate 95th percentiles corresponds to comparable ratio sensitivities in the EWNP and RN20 scenarios. The ratio sensitivity is smaller for SE and R50 scenarios. A similar pattern is observed for Nodularia 95th percentiles, but the ratio sensitivities are much smaller.
5.3.2.3.3 Bottom Oxygen

Bottom oxygen 5th percentiles are little affected by changes to sediment exchange velocity, with the exception of the R50 scenario (Figure 42).

Figure 41 Ratio sensitivities for surface dinoflagellate (left) and *Nodularia* (right) 95th percentiles for south Lake King.

Figure 42 Absolute sensitivity for bottom oxygen 5th percentiles in central Lake Victoria. Units are mg O m$^{-3}$. 
6 Summary

The sensitivity analysis conducted in this work is by no means comprehensive, and considerably more work could be done in this regard. In particular, by focusing individually on three process groups (diffusive exchange, *Nodularia* and sediments) we have not been able to explore interactions between parameters from different processes. A Bayesian approach to this problem would be to combine prior knowledge about parameter values with the observed data to generate a posterior probability density function for each parameter. Well-established methods exist to sample this posterior probability density function and so estimate parameter means and errors (Harmon and Challenor (1997)). Unfortunately these methods are best suited to models with rapid run-times, and with well-defined measures of model-data comparison. In this work we have taken a less sophisticated approach by assuming uniform prior distributions for parameters and visually scrutinising model-data comparisons to find ensembles of ‘observation consistent’ parameter sets. Nevertheless, our approach has been sufficient to illuminate important aspects of model response and its sensitivity to parameter choices.

![Figure 43](image_url)

**Figure 43** Summary table of maximum range in absolute sensitivities for each process and each scenario. The fifteen rows represent the five different scenarios, each with three process groupings (diffusive exchange, *Nodularia* parameters and sediment parameters). The four columns represent four key indicators (95%ile surface chlorophyll, 95%ile surface *Nodularia*, 95%ile surface dinoflagellates and 5%ile bottom oxygen). The marker colours correspond to the range in absolute sensitivity (as shown on the scale bar provided for each indicator). The marker sizes within each column indicate the magnitude of the range in absolute sensitivities (note that marker size should only be compared within a single column, and not compared across columns).

We searched across all model boxes and identified the maximum absolute sensitivities for four indicators and the three process groupings; this allowed the identification of broad patterns of response to parameter variation. Figure 43 tabulates the maximum
absolute sensitivity for each process grouping and scenario. A clear pattern to emerge across the chlorophyll, \textit{Nodularia} and dinoflagellate indicators is that absolute sensitivity is typically largest for the baseline, EWNP and RN20 scenarios, and considerably smaller for the SE and R50 scenarios. By contrast, the absolute sensitivities for oxygen are largest for the SE and R50 scenarios in their response to lower vertical diffusive exchange.

![Figure 44](image)

**Figure 44** Summary table of maximum ratio sensitivity ranges for each process and each scenario. The fifteen rows represent the five different scenarios, each with three process groupings (diffusive exchange, \textit{Nodularia} parameters and sediment parameters). The three columns represent three key indicators (95\%ile surface chlorophyll, 95\%ile surface \textit{Nodularia} and 95\%ile surface dinoflagellates). The marker colours correspond to the range in ratio sensitivity (as shown on the scale bar). The marker sizes also indicate the magnitude of the range in ratio sensitivities (note that colours and marker sizes are directly comparable across columns).

Summarising the ranges in ratio sensitivities in a similar manner highlights a different pattern. In Figure 44 the maximum ranges in ratio sensitivity are shown; i.e. each marker represents the difference between the maximum and minimum scenario:baseline ratio for that indicator percentile. Ratio sensitivity ranges for chlorophyll levels are low to moderate (0.06 and 0.38), suggesting that model predictions about chlorophyll changes relative to baseline are robust to parameter variations. Ratio sensitivities for dinoflagellate levels are higher (0.13 to 0.99), and are consistently affected by sediment and \textit{Nodularia} parameters across all scenarios (although to a lesser extent in the SE scenario). Changes to diffusive exchange have a smaller impact on ratio sensitivities for dinoflagellates.

The largest ratio sensitivities are observed for \textit{Nodularia}. The RN20 scenario \textit{Nodularia} predictions are particularly sensitive to changes in sediment process parameters, and reflects important links between N:P ratios in loads, sediment processes and bloom
composition. For example, when altering P isotherm parameters the RN20 scenario responded with periods of stronger N-limitation (apparent in the 95th percentiles of surface DIP levels). The strength of the periods of N-limitation was very sensitive to choice of P isotherm parameters, and consequently yielded a large sensitivity in *Nodularia* predictions.

A general conclusion is that while predicted chlorophyll levels are robust to parameter variation, the specifics of bloom composition are sensitive to interactions between N:P ratios in loads and sediment process parameters.

The baseline absolute sensitivity to parameters rarely translates into larger sensitivity in the other scenarios. Important exceptions are the dinoflagellate and to a lesser extent *Nodularia* levels, where sensitivities for the EWNP and particularly RN20 scenario can be higher than baseline sensitivities. We conclude that predictions specific to dinoflagellates and *Nodularia* are highly uncertain (a point also stressed in the GLES modelling work). Stephens et al. (2004) reported that a large dinoflagellate bloom occurred in August 1998, following a flood: “The dinoflagellate bloom went largely unnoticed by the wider community, as there were no bright green surface scums that are a feature of blue/green *Nodularia* blooms which have occurred in the area. During the dinoflagellate bloom the water was a dark olive green progressing to a turbid chocolate brown at the height of the bloom.” Two dinoflagellates were identified in the report: *Gymnodinium aureolum* and *Heterocapsa triquetra*. *Gymnodinium aureolum* is often confused with *Gymnodinium mikimotoi*, which kills fish. *Gymnodinium aureolum* does not appear to be toxic, but Stephens et al. (2004) recommend further investigation given its affinity to a known toxic species. The dinoflagellate populations in the model are very sensitive to parameters specific to dinoflagellate growth and mortality (not discussed in this report), as well as the *Nodularia* and sediment parameters.

For most other indicators, parameter sensitivity for the EWNP and RN20 scenarios is similar baseline parameter sensitivity, and sensitivity to parameters is reduced in the R50 and SE scenarios. It should be noted that this is the conclusion for sensitivity to parameters only. Other uncertainties for the R50 and SE predictions (unknown) may very well be much higher than for scenarios with only small incremental changes to loads, especially given that they extrapolate to conditions for which we have no field data for model validation.

### 7 Conclusions and recommendations

The broad conclusions drawn from this work are that predicted peak chlorophyll levels are relatively insensitive to parameter variation. This was anticipated in the original GLES modelling work, and is a consequence of the fact that 95% chlorophyll percentiles are dominated by nutrient inputs during flow events and not significantly affected by internal biogeochemical dynamics between times.

Bottom oxygen 5th percentiles are also relatively insensitive to parameter variations, however they are very sensitive to reductions in vertical diffusive exchange in the SE and R50 scenarios. Bloom composition, particularly the response of the dinoflagellates and *Nodularia*, is much more sensitive to model parameters, and complicated interactions between sediment processes, N:P ratios in loads and phytoplankton process parameters make predictions of these species particularly challenging.

An aim of this work was to assess whether changes to parameters would lead to a different interpretation of model results, and so lead to different management
conclusions. The sensitivity assessment does not change the qualitative conclusions about scenarios reached in GLES; for example, the second entrance still leads to a large decrease in primary production and a large improvement in bottom oxygen under any of the mixing assumptions. In those instances where parameter variation significantly changes quantitative conclusions, it would need discussion with stakeholders to decide whether they would form different conclusions about management actions.

Another aim of the sensitivity analysis was to make recommendations for further process study and model development. The conclusion from this work is that these recommendations will depend on management priorities.

If a second entrance to the Gippsland Lakes were to be considered further, the results of the sensitivity analysis suggest that more work is needed to better understand vertical exchange. Reductions in vertical diffusive exchange yielded large reductions in predicted bottom oxygen levels for the second entrance scenario. We also note that the second entrance scenario predicts a substantial increase in surface DIP levels (a result of stronger N-limitation in the system as denitrification efficiency is boosted). These are conditions that can promote *Nodularia* growth ahead of other species, because of its N-fixing ability. It is likely that the salinity tolerance in the model is preventing *Nodularia* blooms in the second entrance second entrance scenario. Variation in assumed salinity tolerances yielded little change to this prediction, and the experiments by Holland and Beardall (2004) suggest lower growth rates at high salinities for *Nodularia* strains from the Gippsland Lakes. Nevertheless, given the significance of this species, its history in the Lakes, and model assumptions for *Nodularia* germination, growth and mortality, a more comprehensive analysis of the interactions between salinity, N:P ratios and *Nodularia* processes is warranted if a second entrance is to be considered.

The model predictions about peak chlorophyll levels are robust to parameter variation, however if understanding bloom composition is the main priority, the results suggest that more work is needed to understand phytoplankton autecology. Only *Nodularia* parameters were varied in this report and we made no changes to other phytoplankton species’ parameters. However both *Nodularia* and dinoflagellate 95th percentiles were sensitive to these changes, suggesting that the interactions between these species need to be better understood. The model fails to capture the timing of the dinoflagellate blooms, and currently has them occurring later than observed (and coinciding with *Nodularia* blooms). These problems cannot be addressed with parameter changes alone, and require further modifications to process representation in the model.

The balance between N and P limitation in the model is sensitive to many of the sediment parameters, and at the moment these have been constrained to avoid periods of P-limitation and to prevent the system from being too strongly N-limited. This in turn has a strong effect on bloom composition. Thus, while we’ve suggested that phytoplankton autecology needs to be better understood if bloom composition is of particular concern, the sediment processes play a key role in interacting with the phytoplankton dynamics and deserve further attention.

A key concern expressed in the GLES modelling work was that we do not understand the processes controlling long-term P and N storage in the system. A particular concern raised was that if N and P loads are reduced by similar fractions, but sinks in the system respond differently (eg. denitrification is boosted, but P-release from sediment continues), this may exacerbate N-limitation in the system, and so potentially favour *Nodularia* growth. The RN20 scenario was included as a proxy for this situation, and demonstrated that bloom responses are sensitive to N:P ratios.
Our conclusions can be summarised as a list of recommendations conditional on management priorities:

1. If the priority is the long-term response to nutrient loads, then further investment in sediment process studies is recommended;

2. If the priority is to understand bloom composition, then phytoplankton autecology and sediment processes are most in need of further work;

3. If the priority is to consider a second entrance, then further work in understanding physical transport and phytoplankton autecology (particularly *Nodularia*) is most important.

8 References


Environmental Study Technical Report, CSIRO.
