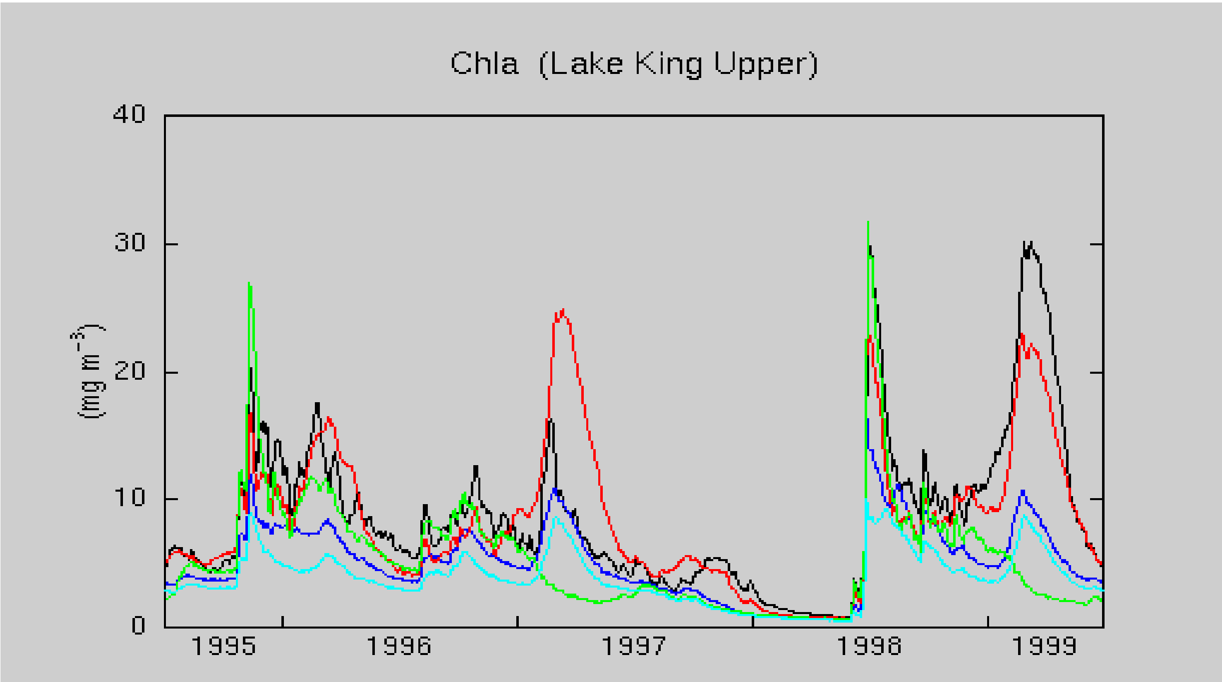
Gippsland Lakes Environmental Study

Integrated Model Scenarios

Technical Report



Prepared by:

John Parslow, Pavel Sakov, and John Andrewartha
CSIRO Marine Research

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Executive Summary

This report provides results and conclusions from a set of model scenarios carried out as part of the Gippsland Lakes Environmental Study, using an integrated model of nutrient and sediment cycling and impacts within the Lakes. The development and calibration of the model is described in a separate report.

The scenarios were agreed with stakeholders, and designed to provide information about the likely response of the Lakes to changes in flows, loads and exchanges.

The scenarios fall into various classes:

- Load scenarios were designed to examine the response of the Lakes to reductions in loads in eastern and/or western catchments.
- Flow scenarios were designed to examine the impacts of increases or decreases in flows in eastern or western catchments, independent of loads.
- Exchange scenarios involve modifications in ocean exchanges resulting from shallowing or deepening the existing entrance or creating a second entrance.
- The long-term scenarios simulate the behaviour over periods of 32 years with or without growth in loads.

Scenario simulations were conducted by running the model for 12 years, with July 1995 to June 1999 forcing repeated 3 times, and saving model output for the last 4 years. In the long-term scenarios, the model was run for 40 years, with output saved for the last 32 years. In the long-term growth scenario, loads increased by 1% for year over the last 32 years only.

Nine water column indicators and six benthic indicators have been selected to characterise the model response. These capture phytoplankton biomass and production, dinoflagellate and *Nodularia* blooms, ammonia and phosphate, oxygen, TSS and Secchi depth in the water column, and sediment respiration, denitrification efficiency, microphytobenthos, settled phytoplankton, macroalgae and benthic primary production in sediments.

Model output has been aggregated spatially into the three principal basins, L. Wellington, L. Victoria and L. King. Temporal variation is summarised statistically by computing minimum, 5%ile, 20%ile, 50%ile, 80%ile, 95%ile and maximum values for each indicator over a 4-year period.

The key conclusions are as follows:

Loads from western catchments are 2 to 3 times those from eastern catchments in the 1995-99 period: 20% reductions in western catchment loads have much more impact on L. Wellington and L. Victoria than 20% reductions in eastern catchments.

Phytoplankton biomass responds approximately linearly to 20% load reductions, but nonlinear positive feedbacks involving bottom water hypoxia, reductions in denitrification efficiency, and increases in ammonia and phosphate flux from sediments, amplify the impact on bottom water ammonia and phosphate in L. Victoria and L. King.

Despite this amplification, large reductions in loads (50 to 70%) are needed to eliminate bottom water hypoxia and associated accumulation of ammonia and phosphate in L. Victoria and L. King. Load reductions of 70% eliminate bottom water hypoxia in both lakes (95%ile dissolved oxygen above 5000 mg O m^{-3}), and almost eliminate episodes of bottom water accumulation of ammonia and phosphate. Phytoplankton biomass is still substantial, with median levels about 4 mg Chl a m^{-3}, and 95%ile levels around 15 mg Chl a m^{-3}. Under these loads, the system might be classed as upper mesotrophic.

The Lakes are predicted to be more strongly N-limited under load reductions, but *Nodularia* bloom densities are predicted to fall, due to reductions in phosphate supply.

Light attenuation is predicted to remain high, with relatively little recovery in benthic plants. This is partly because coloured dissolved organic matter (CDOM) concentrations in rivers are assumed to remain fixed.

Reductions in N and P loads from the MID by 40% represent only 10 and 3% reductions in western catchment N and P loads respectively, and have proportionally small effects on Lakes indicators.

The dominant effect of changes in flows in the model is their contribution to flushing. Increased flows result in increased flushing and improved water quality, while flow reductions lead to poorer water quality. The 20% changes in flows generally have smaller impacts than 20% changes in loads.

In the hydrodynamic model, creation of a second entrance approximately halves flushing times. It has a surprisingly large impact on water quality in L. Victoria and L. King in the integrated model, equivalent to a 50% reduction in catchment loads. This is surprising because sediment and nutrient budgets are thought to be dominated by internal sinks rather than flushing.

Positive feedbacks involving increases in bottom water oxygen amplify the impact of the second entrance on nutrient cycles. The hydrodynamic and box-model exchanges in this scenario involve enhanced vertical mixing, and this may help to reduce bottom water hypoxia.

The second entrance increases salinities by about 3 to 4 PSU, and consequently reduces light attenuation due to CDOM. The increase in salinity is sufficient to inhibit *Nodularia* blooms substantially in the model, although the assumed salinity tolerance needs to be confirmed. Assessment of other ecological impacts of increased salinity in the Lakes, and possibly within the lower reaches of the waterways and wetlands of the system, are not within the scope of the current study.

Modifications to the existing entrance that approximately double or halve tidal exchanges through the entrance have small effects on salinity and flushing rates in the hydrodynamic model, and relatively small effects on water quality indicators in the integrated model. Predicted changes in vertical mixing rates resulting from shallowing or deepening the entrance tend to oppose and in some cases outweigh effects of changes in flushing rate.

We do not understand processes controlling accumulation of N or P in sediments on decadal time scales sufficiently to represent these in the model, and so the model cannot be used to answer questions about transients on these scales. The model adapts to loads on time scales of a few years, and there is consequently no change in indicators over 32 years with repeated baseline loads.

In the growth scenario, with loads increasing by 1% p.a., the model response at any point in the growth trajectory is essentially the same as its response would be if the loads at that time were fixed. After 30 years, the loads have increased by just over 30%. The model again shows evidence of non-linear positive feedbacks, with bottom water quality indicators such as ammonia increasing by as much as 200%. While long term transient responses are uncertain, the general conclusion of a non-linear response to increasing loads should be robust.

Key model limitations and uncertainties that affect model predictions include:

The model lacks processes needed to address long-term transients. However, the observed response to the 1997/98 drought year suggests improvement in water quality following load reductions could be rapid.

The model may overpredict denitrification efficiency in L. Wellington, and underpredict it in L. Victoria. If so, the sensitivity of L. Wellington to load changes may be underestimated, and that of L. Victoria over-estimated.

The model does not buffer phosphate in L. Wellington as tightly as observed, and may be too easily pushed into P-limitation.

The model is unable to reproduce the timing of dinoflagellate blooms, possibly because it misrepresents interactions with diatoms through an assumed common grazer. Consequently, scenario predictions concerning dinoflagellates should be treated as highly uncertain.

Model predictions about the impacts of reductions in phosphate and increases in salinity on *Nodularia* are plausible, but would need to be confirmed by focused process studies.

While model predictions of phytoplankton composition are therefore subject to considerable uncertainty, the model representation of phytoplankton as a whole, and their role in nutrient cycling and eutrophication, should be more robust.

The physical representation of flushing is well-calibrated against salinity, and should be robust. However, the representation of vertical mixing is more uncertain, and this does affect predictions of bottom-water oxygen and nutrient concentrations. The model does over-estimate oxygen depletion and ammonia accumulation in bottom waters in L. Victoria, and this could indicate that vertical mixing rates there are underestimated. If so, the model may over-estimate the sensitivity of L. Victoria to changes in loads.

There is some uncertainty about the composition of TN and TP loads in rivers. Changes in composition (e.g. increases in the refractory fraction) could affect model predictions. In load scenarios, we have assumed that TSS varies along with TN and TP, but CDOM concentrations remain fixed. This has significant implications for predicted light attenuation. Substantial reductions in CDOM would favour recovery of benthic plants or microphytobenthos in deeper basins. However, high CDOM concentrations can occur in runoff from pristine catchments: it is not clear that management actions are likely to reduce CDOM concentrations in rivers.

Model scenario predictions and the associated uncertainties need to be considered, along with management objectives and the feasibility and costs of alternative actions, in choosing management strategies. It is important in any case to first establish management objectives and performance indicators.

Where management strategies involve a series of incremental actions, as seems likely in reducing catchment loads, short-term improvements in understanding and reductions in model uncertainty provide less benefit. It is more important to establish a long-term monitoring program that allows managers to assess progress towards objectives.

If management strategies involving large one-off capital works were to be contemplated, then further upfront investment in process studies and modelling to reduce uncertainties are recommended.

1 Introduction

The CSIRO Gippsland Lakes Environmental Study (GLES) 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 management options to address these problems.

The development of a prognostic, process-based model of nutrient cycling and algal blooms is a key part of the Study. We need to predict the impacts of management options that fall outside the range of past experience. We can only do this with confidence to the extent that our predictions are based on a solid understanding of the fundamental processes that determine the response of the Lakes to changes in circulation and nutrient and sediment loads.

The development and calibration of an integrated model of nutrient cycling and algal blooms for the Gippsland Lakes is described by Parslow et. al. (2001). The reader should refer to that report for details of the model formulation and calibration, and for insights into the behaviour of the Lakes, and their dynamical response to time-varying loads and flows.

This report presents the results obtained by subjecting the integrated model to a series of scenarios involving changes in loads, flows or exchanges with Bass Strait. These scenarios, described in detail in the next section, were arrived at after discussion with the GLES Steering Committee. They are designed to inform managers about the likely response of the Lakes to generic classes of management actions, and to help managers to choose among classes and/or levels of management action.

The report begins by defining the scenarios, and then describes the indicators used to characterize model outputs. The results are then presented as a series of figures and tables, with discussion and conclusions.

2 Scenarios.

For the purposes of presentation and analysis, the scenarios have been divided into three groups: load scenarios, flow and exchange scenarios, and long-term scenarios. All modified scenarios are compared to a baseline scenario.

2.1 The Baseline Scenario

The baseline scenario represents the response of the calibrated model to the estimated forcing and loads for the July 1995 to June 1999 period. Details of the baseline forcing and loads, and the model response, are given by Parslow et al. (2001). For the baseline scenario, and for other scenarios except where specified, the model is run for a period of 12 years, starting nominally in July 1987, with the 1995-99 forcing repeated 3 times. Indicators are based on the last 4 years, so as to minimize the impact of initial conditions on model predictions.

2.2 Load Scenarios.

These scenarios represent changes in nutrient and sediment loads from the catchments. In all these scenarios, where loads from a particular catchment are increased or decreased by a nominated percentage, the daily loads have been multiplied by a constant factor. There has been no attempt to distinguish between base loads and peak loads. In all cases except the Macallister Irrigation District (MID) scenarios, loads of TSS, TN and TP, and the various forms of N and P, have all been scaled equally. In the MID scenarios, TP and/or TN, and the associated TP and TN constituents, have been modified.

The load scenarios were chosen in order to separate the effects of load changes in the western catchments (into L. Wellington), and the eastern catchments (into L. Victoria and L. King) respectively. The western catchments include the Latrobe, Thomson and Avon Rivers. The eastern catchments are dominated by the Mitchell, Tambo and Nicholson Rivers, but also include coastal catchment inputs into L. King south and L. Victoria (see Parslow et al. 2001).

In formulating these load scenarios, as well as the flow scenarios described below, 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 management scenarios could well be the next step in development of a management strategy for the Lakes.

There are 8 load scenarios, which involve different levels of reduction in eastern and western catchments separately or together. Loads for the MID scenarios were provided by Grayson et al. (2001), and involve reductions in loads from the Irrigation District alone.

2.3 Flow and Exchange Scenarios.

The flow scenarios represent effects of changes in flow that might result from increased or decreased water diversion. Again, it has been assumed that daily flow will be increased or decreased by a constant fraction: there is no distinction made between base and peak flow. It is further assumed that catchment loads remain fixed, so that concentrations of nutrients and sediment increase or decrease as flows respectively decrease or increase. We acknowledge that this assumption may not hold true in specific sections of the catchment, as there is significant variation in the source of nutrient and sediment inputs into streams, and in how nutrients and sediments are processed in streams and reservoirs. However keeping loads fixed in flow scenarios allows us to contrast the response of the model to changes in loads and changes in flows on a catchment wide basis.

For the flow scenarios, the hydrodynamic model was rerun, with modified flows, but other forcing unchanged. The box model exchanges were derived from these modified hydrodynamic model runs.

The exchange scenarios represent the effect of modifying the exchanges between the Lakes and Bass Strait. These scenarios are described in more detail in the Hydrodynamic Modelling Report (Walker and Andrewartha, 2000). Scenario E1 involves the imposition of a second entrance between the Gippsland Lakes and Bass Strait, at Ocean Grange. Walker and Andrewartha found that the second entrance resulted in substantially increased salinities in L. King and L. Victoria, and approximately halved the flushing time for these basins.

Scenarios E2 and E3 retain the existing entrance, but this entrance is respectively shallowed and deepened, so as to approximately halve or double the tidal flow through the entrance. Walker and Andrewartha found that these modifications had negligible influence on salinities and flushing rates for the Lakes. They concluded that tidal exchanges play very little role in flushing the Lakes, at least within the range of modifications considered.

2.4 Long-term Scenarios

In the first long-term scenario, the model is run for 40 years, nominally starting in July 1987, with the standard baseline loads for July 1995 to June 1999 repeated 10 times. The output from years 9 to 12 corresponds to the baseline output B. The output 14 years later (years 23 to 26) is referred to as B14, and the output a further 14 years later (years 37 to 40) is referred to as B28.

In the second long-term scenario, the model is run for 40 years, but starting after 8 years, loads are increased by 1% per annum. This increase is implemented as a continuous small daily growth rate. Again, outputs from years 23 to 26 and years 37 to 40 are presented as G14 and G28 respectively.

Model Scenarios Presented in this Report - the scenario code is used in subsequent figures and tables.

Code	Description
2.4.1 Baseline Scenario	
B	Standard calibration run, with Jul 95-Jun 99 forcing.
Load Scenarios	
L1	20% reduction in eastern catchment loads.
L2	20% reduction in western catchment loads.
L3	20% reduction in all loads
L4	40% reduction in western catchment loads
L5	40% reduction in P loads from MID
L6	40% reduction in N and P loads from MID
L7	50% reduction in all loads
L8	70% reduction in all loads
2.4.2 Flow, Exchange Scenarios	
F1	Decrease flow by 20% in all rivers
F2	Decrease flow by 20% in eastern rivers
F3	Increase flow by 20% in western rivers
E1	Exchanges from hydrodynamic model with 2nd entrance at Ocean Grange
E2	Exchanges from hydrodynamic model with existing entrance shallower.
E3	Exchanges from hydrodynamic model with existing entrance deeper.
2.4.3 Long-term scenarios	
B14	years 23-26 from 40 y baseline run, with repeated 95-99 forcing.
B28	years 37 to 40 from 32 year baseline run.
G14	years 23-26 from 40 y run, with 1% p.a. growth in loads after year 8.
G28	years 37 to 40 from 40 y run, with 1% p.a. growth in loads after year 8.

3 Scenario Indicators.

The model output includes a large suite of state variables, and a considerable number of derived or diagnostic variables (approximately 60 in all). Each variable is output daily, in each of 14 model water column cells, or 6 model sediment cells. Reproducing the output time series for all variables, for all locations, for all 19 scenarios, is clearly not feasible, and would in any case be counter-productive. The challenge is to choose an informative subset of indicator variables, and to statistically summarise the spatial and temporal variation in these variables.

We have chosen to consolidate the spatial variation into the three principal basins: L. Wellington, L. Victoria and L. King. This has meant combining output for model boxes 3, 4 and 5 in L. Victoria, and model boxes 2 and 7 in L. King. We have omitted Jones Bay (model box 8), as it is shallow, close to major river inputs, and we have no calibration data in Jones Bay. Similarly, we have omitted the entrance box 1, as it is strongly affected by exchange with Bass Strait.

L. Wellington includes only one water column layer, but L. Victoria and L. King include surface and bottom layers, and we have calculated separate statistics for these. Similarly, separate statistics are computed for sediment layers in all basins.

We have chosen to summarise the temporal variation by constructing frequency histograms for each variable, within each basin, over the designated 4-year output period. The temporal variation in forcing and baseline calibration simulations is discussed in more detail in the model implementation and calibration report (Parslow et al., 2001). There is very strong temporal variation in flows and loads within the 4-year baseline period on event, seasonal and inter-annual time scales. A substantial fraction of the total cumulative 1995-99 load into L. King was delivered in one flood event, lasting a few days, in June 1998. Loads are generally concentrated in the winter and spring of each year, but loads in the winter and spring of 1997 were particularly low, resulting in a protracted 18-month period of unusually low loads and flows, especially into L. King.

Many of the model variables undergo large changes in response to these changes in loads. The frequency histograms capture the extent of this variation, although of course the details of the dynamical response are lost. The histograms are sufficient for broad comparisons across scenarios. If a single scenario was singled out for further investigation, analysis of the time series output for key indicators might be warranted (although one might then want to estimate and use more realistic time series of loads corresponding to specific management actions in the catchment).

The histograms are further summarised here as a series of percentile values. The minimum, 5%ile, 20%ile, 50%ile, 80%ile, 95%ile and maximum, have been computed for each variable. These are presented in plots as box and whisker diagrams (Fig. 1-51). **In all these figures, the vertical line joins minimum and maximum, the thin box joins 5%ile and 95%ile, the thick box 20%ile and 80%ile, and the horizontal bar marks the median.** The medians and 95%ile values, or where appropriate the 5%ile values, are also provided in Tables 1-30. These summary plots and tables allow easy comparison of results across scenarios.

We have selected a subset of the model variables and diagnostics as indicators for the scenarios. These indicators, shown below, represent common water, sediment or ecological indicators, or provide insight into the response of those indicators to the scenarios. Chlorophyll, ammonia, phosphate and dissolved oxygen are widely used water quality indicators. Ammonia, phosphate and dissolved oxygen in bottom waters in L. Victoria and L.

King are particularly useful indicators of eutrophication: current eutrophic conditions are characterised by bottom-water hypoxia or anoxia, and increases in bottom-water ammonia and phosphate. Ammonia and phosphate concentrations in surface waters provide some indication of switches between N-limitation (low ammonia) and P-limitation (low phosphate).

Model scenario indicators.

Name	Description and units
Water Column Indicators	
Chl a	Chlorophyll a concentration in water column surface layer (mg Chl a m^{-3})
Dinoflagellates	Dinoflagellate biomass (mg Chl a m^{-3}) in surface layer
Nodularia	*Nodularia* biomass (mg Chl a m^{-3}) in surface layer
NH_4	Ammonia concentration in water column surface or bottom layer (mg N m^{-3})
PO_4	Phosphate concentration in water column surface or bottom layer (mg P m^{-3})
TSS	Suspended sediment concentration in surface layer (g m^{-3})
Oxygen	Concentration of dissolved oxygen in bottom layer (mg O m^{-3})
Secchi depth	Index of light penetration (m), approximately 2/Kd
PPProd	Pelagic primary production (mg C m^{-2} d^{-1})
Benthic Indicators	
Sediment Respiration	Sediment oxygen consumption rate (mmol O_2 m^{-2} d^{-1})
Denitrification Efficiency	Ratio of N_2 production to NH_4 release by remineralization in sediments.
MPB	Microphytobenthos biomass (mg Chl a m^{-2})
Chl a	Total microalgal biomass in sediment (mg Chl a m^{-2}) (includes settled diatoms)
Macroalgae	Macroalgal biomass (mg N m^{-2})
BPProd	Benthic primary prodn (MPB + macrophytes) (mg C m^{-2} d^{-1})

Chlorophyll a is used as a measure of plankton biomass. Dinoflagellate and cyanobacterial (*Nodularia*) biomass are presented separately, although these should be treated cautiously, given the model's limitations (see Parslow et. al., 2001, and discussion and conclusions below). Light attenuation is given not only as a control on phytoplankton biomass, but also as a control on the potential depth range for macroalgae and seagrass. Pelagic primary production is provided as a complementary eutrophication measure to biomass (Chlorophyll a): it measures the autotrophic organic matter flux through the plankton and allows the turnover rate of the phytoplankton to be deduced.

Sediment respiration rate and denitrification efficiency reflect the degree of eutrophication and organic loading of the sediment. Sediment oxygen consumption is also strongly controlled by bottom-water oxygen concentrations in L. Victoria and L. King. In model calibration runs, it was found that the biomass of settled phytoplankton greatly outweighed the predicted biomass of benthic microalgae (microphytobenthos or MPB), so both are presented. Macroalgal biomass is also provided although it was predicted to be very low under baseline conditions. Benthic primary production includes contributions from benthic microalgae and macrophytes, and is included for comparison with pelagic primary production.

4 Results and Discussion

4.1 Load Scenarios.

4.1.1 20% Load Reductions

The first three load scenarios L1 to L3 involve 20% reductions respectively in loads from eastern catchments alone, from western catchments alone, and from both catchments. These allow us to assess the relative sensitivity of model indicators in L. Wellington, L. Victoria and L. King to relatively small changes in loads from the eastern and western catchments.

Most indicators in L. Wellington show little or no response to reductions in loads in the eastern catchments, as one might expect. Exceptions include 95%ile dinoflagellates (Fig. 2, Table 3), 95%ile ammonia (Fig. 4, Table 6), and 95%ile MPB and benthic primary production. We return to these shortly. Indicators in L. Victoria are generally more responsive to reductions in western than eastern catchments. We see a common pattern in L. Victoria that effects of reductions increase from L1 to L3. Results for L. King are more variable: in some cases, L1 shows more impact than L2, and in other cases the converse is true.

There is considerable variation among indicators in the sensitivity to load reductions. Median surface Chl a in L. Wellington and L. Victoria (Fig. 1, Table 1) shows a close to linear response to load reductions, with 20% load reductions in western or both catchments producing reductions in Chl a of 17 to 26%. The reduction in 95%ile Chl a (bloom levels) is greater, 26 to 30%. Reductions in Chl a in L. King are less than linear, <5% in median and <13% in 95%ile Chl a.

Median ammonia levels in surface waters in all basins show a close to linear response to load reductions, but there are large reductions, up to 80%, in 95%ile ammonia values (Fig. 4, Tables 5 and 6). Surface ammonia is only elevated in the baseline runs during periods of P depletion, so these results reflect a tipping of the balance towards N limitation, reducing the extent of periods of P depletion. Bottom-water ammonia values in L. King and L. Victoria respond very nonlinearly to reductions in loads, with up to 84% reductions in 50%ile ammonia, and 50% reductions in 95%ile ammonia. Ammonia concentrations in bottom waters tend to have a bimodal distribution, with very low values alternating with high values associated with bottom-water hypoxia. Thus, the very large reductions in median bottom-water ammonia reflect a reduction in the duration of periods of hypoxia and high bottom-water ammonia to less than 50% of the time.

Phosphate concentrations in surface waters in L. Victoria and L. King show unexpected increases in surface water concentrations, especially in L1 where loads are reduced in the eastern catchments alone (Fig. 5, Tables 9 and 10). Again, this suggests that reductions in loads are pushing the system in the direction of N limitation. Eastern catchment loads have higher N:P ratios than loads into L. Wellington, so reductions in the former do tend to favour N limitation. Bottom-water phosphate concentrations show weaker responses than ammonia to load reductions, but still respond much more than linearly, with up to 36% reductions in median DIP, and 41% reductions in 95%ile DIP. Again, these reflect reductions in the duration and extent of periods of hypoxia and enhanced DIP release from bottom sediments.

Median bottom-water oxygen concentrations increase by 36% in L. Victoria under reductions in western catchment loads, but by only 12% in L. King (Fig. 8, Tables 13 and 14). It appears that the principal effect of reduced loads on sediment release of ammonia and phosphate may occur in L. Victoria. This is also evident in sediment denitrification efficiencies (Fig. 13, Tables 23 and 24): under L3, median denitrification efficiency is unchanged in L. Wellington, and increases by only 4% in L. King, but increases by 96% in L. Victoria.

Primary production rates in surface waters (Fig. 11, Tables 19 and 20) show a qualitatively similar response pattern to Chl a, but reductions in primary production are larger: that is, reductions in loads are leading to reductions in both biomass and turnover rate of phytoplankton. These reductions in organic matter production result in comparatively small reductions in sediment respiration (oxygen consumption) rates, at least in L. Victoria and L. King. This is because increases in bottom-water oxygen have a compensating effect, increasing aerobic respiration as a fraction of total respiration.

Suspended sediment concentrations decrease almost in proportion to loads (Fig. 9, Tables 15 and 16), as one might expect. Median Secchi depths increase by only 8 to 12% in L3, and 5%ile Secchi depths increase by only 10 to 15%. CDOM loads are unchanged, and TSS contributes only a part of light attenuation, making a bigger relative contribution in L. Wellington, and at times of minimum Secchi depth. The fact that phytoplankton turnover rates decrease indicates that N limitation outweighs light limitation most of the time in surface waters.

Median dinoflagellate and *Nodularia* biomasses are very low (Fig. 2 and 3), so we have presented Tables only for 95%ile (bloom) values (Tables 3 and 4). Predicted dinoflagellate densities are very low in L. Wellington. The model predicts increases under load reductions, but these values are still negligible. The model also predicts small increases in L. Victoria and L. King under 20% load reductions. These predictions may be due to reductions in zooplankton biomass and grazing pressure and, given the model limitations, should be treated with some skepticism. Predicted *Nodularia* biomass shows reductions under reduced loads, but these reductions are generally less than linear.

Benthic plants are severely light limited in all three basins in the model. Microphytobenthos biomass (Fig. 14, Tables 25 and 26) increases, especially in L. Wellington, but this increase takes place against a very small baseline. MPB biomass is only a small fraction of total settled phytoplankton in the surface sediment. The latter decreases along with surface Chl a (Fig. 15, Tables 27 and 28). The 95%ile macroalgal biomass is essentially zero in L. Wellington and L. King, but increases substantially in L. Victoria. Benthic primary production is also negligible in L. Wellington: it undergoes a large relative increase for L2 and L3, but absolute values are still negligible. Increases in benthic primary production are more than 20% in L. Victoria, but less than 20% in L. King.

The key lessons from scenarios L1 to L3 are:

- 20% load reductions in western catchments alone have larger effects on L. Wellington and L. Victoria, while L. King is roughly equally sensitive to 20% reductions in either eastern or western catchments;
- 20% load reductions result in similar percent reductions in phytoplankton biomass, but larger percent reductions in primary production, and in bottom-water ammonia and phosphate;
- there is evidence for non-linear positive feedbacks, with increases in bottom-water oxygen and decreases in sediment respiration leading to increased denitrification efficiency, and less phosphate release;
- these load reductions shift the system away from P and towards N limitation;
- there is a less than linear response in the biomass of *Nodularia* i.e reductions occur, but they are less than 20%
- the system remains dominated by pelagic production, with low benthic plant biomass and production.

4.1.2 40 to 70% Load Reductions

Under scenario L3, the system is still subject to intense phytoplankton blooms, with extended periods of bottom-water hypoxia and elevated nutrients. Scenarios L4, L7 and L8 consider larger reductions in catchment loads, with respectively a 40% reduction in western catchment loads, a 50% reduction in all loads, and a 70% reduction in all loads.

Scenario L4 shows larger (almost proportional) reductions in Chl a and median ammonia concentrations in L. Wellington and L. Victoria, with smaller reductions, about 20%, in L. King (Fig. 1 and 4, Tables 1, 2 and 5). Reductions in pelagic primary production are again larger (Tables 19 and 20). Median ammonia concentrations in bottom-waters in L. Victoria and L. King are reduced by more than 80%, and 95%ile ammonia concentrations in L. Victoria by 56% (Tables 7 and 8). Bottom-water PO_4 concentrations are decreased by more than 40% in both L. Victoria and L. King (Table 11). Median oxygen concentrations have increased by 45% in L. Victoria, but only 10% in L. King, and 5%ile oxygen concentrations are still very low in both lakes (Table 13 and 14). MPB biomass is still low. Predicted reductions in 95%ile *Nodularia* biomass in L. King are only about 15%, presumably because eastern catchment loads are not reduced.

Scenario L7 and L8 represent very substantial reductions in loads from all catchments. Under these scenarios, median chlorophyll levels are reduced by around 50% and 60% respectively, to around 5 and 4 mg m^{-3} in all basins (Fig. 1, Table 1). The 95%ile chlorophyll values are reduced to around 10 to 15, and 8 to 10, mg Chl a m^{-3}, or by around 60% and 70% respectively. These still represent quite substantial blooms associated with runoff events. For these scenarios, all lakes are strongly N-limited: 95%ile ammonia values in surface waters are low (Table 8). Median surface phosphate is reduced in L. Wellington, but increased in L. Victoria and L. King (Tables 9 and 10) . Median primary production rates in surface waters are reduced by 60% and 70% in L. Wellington, and by 80% and 90% in L. Victoria (Tables 19 and 20).

Median bottom-water ammonia values are low in both L7 and L8, and 95%ile bottom-water ammonia is low in L8, indicating that release of ammonia into bottom waters has shut down (Tables 7 and 8). The 95%ile phosphate in bottom waters is low for both L7 and L8. This is consistent with higher bottom-water oxygen concentrations: the 5%ile oxygen concentrations are above 5000 mg O m^{-3} in L. King for L7, and in L. King and L. Victoria for L8 (Table 13). Median denitrification efficiencies have increased to around 35% in both Lake Victoria and L. King (Table 23).

TSS values have again decreased approximately in proportion to loads, and Secchi depths have increased, although only by 30 to 50%. Increases in light penetration are not sufficient to increase MPB biomass substantially: there are some large relative increases in L. Wellington, but absolute values are still very low. Macroalgal biomass actually decreases in L. Victoria: the predicted increase in light penetration is outweighed by a decrease in nutrient concentrations.

Dinoflagellate bloom densities are reduced substantially, by around 50% and 60%, in L. Victoria and L. King (Table 3). *Nodularia* bloom densities are reduced by around 50% and 70%, roughly in proportion to loads (Table 4).

The model suggests that a 70% load reduction is sufficient to eliminate episodes of bottom-water hypoxia and associated shut-down of denitrification in all basins, while for a 50% load reduction, there are still (limited) episodes of hypoxia in L. Victoria. The Lakes are predicted to remain dominated by phytoplankton production, with moderate chlorophyll levels except in bloom periods associated with runoff events. The model predicts substantial reductions in the intensity of dinoflagellate and *Nodularia* blooms, although confidence attached to these predictions is lower.

4.1.3 Macalister Irrigation District (MID) Load Reductions

Scenarios L5 and L6 represent load reductions from the MID, which affect inputs through the western rivers. Scenario L5 represents a 40% reduction in TP load from the MID, with no change in TN load. Scenario L6 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 L5, and 9.3 for L6.

Given the results of scenarios L2 and L3, 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 (Fig. 1-17, Tables 1-30). Decreases in surface chlorophyll and primary production are about 10% or less, with quite small impacts in L. Wellington (Tables 1, 2, 19 and 20). Median ammonia also changes by less than 10%, but 95%ile surface ammonia increases substantially, particularly in L. Wellington, because the shift in N:P ratios in loads is sufficient to extend periods of P-limitation. Surface DIP concentrations are reduced by up to 20% in L. Wellington, again reflecting a shift in N:P balance, as well as the reduction in P loads. Bottom ammonia and phosphate concentrations are reduced by up to 20%. Bottom oxygen concentrations are increased slightly, while suspended sediment concentrations and Secchi depths are left virtually unchanged.

Sediment respiration rates are reduced only marginally. Median denitrification efficiencies are increased substantially in L. Victoria, but this again reflects the bimodal distribution of this variable, and the sensitivity of the median to small changes in loads. MPB and macroalgal biomass and benthic primary production are affected only slightly. *Nodularia* biomass is reduced in Lake 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.

Overall, MID loads represent only a small fraction of TN and TP loads, and reductions in MID loads have impacts which are commensurate with their magnitude. One caveat should be noted here: we do not have independent information on the composition of TN and TP fractions in MID loads as opposed to other sources, and have used the same rules, described in the Calibration Report (Parslow et al. 2001), for allocating TN and TP to different inorganic and organic fractions in all load scenarios. It is probable that MID loads include more available N and P, and therefore have a more immediate impact on nutrient and phytoplankton levels, than loads from other subcatchments. If this is true, the model may underestimate the impact of MID reductions, but only slightly. Given that nitrate and inorganic P (PIP+DIP) represent close to 50% of total western catchment TN and TP loads into L. Wellington, it seems unlikely that changes in the assumed MID composition would radically change these results.

4.2 Flow Scenarios

The flow scenarios F1 to F3 involve respectively a decrease in flow by 20% in all rivers, a decrease in flow by 20% in eastern rivers, and an increase in flow by 20% in western rivers.

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 (F1) produce increased median Chl a concentrations in all basins (Fig. 18, Table 1), and increased 95%ile Chl a concentrations in L. Wellington and L. Victoria. Median ammonia concentrations increase, and 95%ile surface ammonia concentrations increase very substantially in L. Wellington, suggesting increased periods of P-limitation. Median bottom ammonia concentrations also increase disproportionately, by around 40%, in both L. Victoria and L. King (Table 7). Surface and bottom phosphate concentrations also increase, but by smaller percentages. Median bottom oxygen is reduced by around 10% in L. Victoria and L. King. Median primary production in surface waters increases by around 10 to 15%: again, phytoplankton biomass is not only increased, but is turning over faster. Median denitrification efficiencies are reduced by 35% relative to baseline in L. Victoria.

Decreasing flows in eastern catchments alone generally has similar qualitative effects, but with smaller magnitude.

Increasing flows by 20% in western catchments (F3) reduces median chlorophyll by about 10% in L. Wellington and L. Victoria, and 1% in L. King (Table 1). Median ammonia concentrations decline by about 10% in surface waters, but by 30 to 40% in bottom waters. Median phosphate concentrations decline by about 20% in bottom-waters (Table 11), while median bottom oxygen values there increase by 15% in L. Victoria, but only 3% in L. King (Table 13). Median denitrification efficiencies increase by 55% in L. Victoria, but only 8% in L. King (Table 23).

Median TSS concentrations decline by about 10% in L. Wellington, but increase by 10% in L. Victoria and L. King. This presumably reflects increased export of suspended sediment from L. Wellington to L. Victoria. Secchi depths increase by 4% in L. Wellington, and decrease by only 1 to 2% in L. Victoria and L. King. Consistently, MPB biomass increases in L. Wellington, but decreases in L. Victoria and L. King, as does macroalgal biomass in L. Victoria.

One might expect increasing flows in F3 to reduce salinities in L. King, and possibly favour *Nodularia*. However, predicted 95%ile *Nodularia* biomass decreases in L. King under F3 (Table 4). Presumably, the reduction in phosphate in bottom waters and increased flushing outweigh any salinity reduction in the model.

With loads fixed, the principal effect of changes in flow seems likely to be the associated change in flushing rates. The results are broadly consistent with this: decreases in flow decrease flushing rates and lead to increases in nutrient and chlorophyll concentration, while increases in flow have the opposite effect. As was the case for loads, bottom-water and sediment properties in L. Victoria and L. King prove most sensitive to flows. Increased flows are likely to increase entrainment and ventilation of the bottom layer. This, combined with the positive feedbacks among organic matter production, oxygen consumption, denitrification efficiency and phosphate release, lead to disproportionate impacts in bottom waters.

Increasing flows into L. Wellington by 20% (F3) has positive effects on nutrients, oxygen and phytoplankton, but the model does predict increased export of suspended sediment into L. Victoria and L. King, and small decreases in Secchi depth in these systems. This results in some further disadvantage for benthic plants.

4.3 Exchange Scenarios

4.3.1 2nd Entrance

The three exchange scenarios, E1, E2 and E3 correspond respectively to a second entrance at Ocean Grange, reduced tidal exchange through the existing entrance, and increased tidal exchange through the existing entrance.

Of these three scenarios, the second entrance has by far the most dramatic effect, at least for L. Victoria and L. King. The model predicts a reduction in Chl a in L. Victoria and L. King in E1 by almost 50% (Tables 1 and 2). Median surface ammonia in these Lakes is reduced by 60%, 95%ile surface ammonia by 90%, and median bottom ammonia by 96% (Tables 5, 6 and 7). The 95%ile bottom ammonia is reduced by about 60%, but is still high (Table 8), indicating that episodes of stratification and accumulation still occur, but for shorter periods. Surface phosphate concentrations are increased slightly (less P-limitation), but median bottom-water phosphate concentrations are reduced by 40%, and 95%ile bottom-water phosphate concentrations by 65% (Tables 11 and 12). The 95%ile bottom-water phosphate concentrations fall to background levels in L. King.

Bottom-water oxygen values are increased very substantially in both Lakes. The 5%ile bottom oxygen concentration in L. Victoria is still low (2380 mg O m^{-3}), but 5%ile bottom oxygen in L. King is above 5000 mg O m^{-3}. Median denitrification efficiencies are increased about 300% in L. Victoria, and by 26% in L. King. Median primary production rates in both Lakes are decreased by about 70%.

Median suspended sediment concentrations are reduced by 30% in L. King, but less than 10% in surface waters in L. Victoria and L. Wellington. Median Secchi depths are increased by 30% and 47% in L. Victoria and L. King. This increase is likely primarily due to increased salinities and decreased CDOM concentrations, due to additional flushing. Despite the increased light penetration, predicted MPB biomass is still very low (Tables 25 and 26), and macroalgal biomass in L. Victoria increases by 30% above baseline.

As one might expect, the 2nd entrance has comparatively little effect on L. Wellington. Phytoplankton biomass and primary production are virtually unaffected, as are median ammonia and phosphate concentrations. Median salinity and Secchi depth are increased slightly.

In general, the beneficial effects of the 2nd entrance are confined to L. Victoria and L. King. For these Lakes, in terms of impacts on nutrient cycling, the 2nd entrance is comparable to a 50% reduction in loads (scenario L7). Walker and Andrewartha (2000) noted that the 2nd entrance approximately halved flushing times for the main body of the Lakes, but resulted in less pronounced reduction in flushing times for Lake Wellington. If the budgets for N and P were dominated by export, rather than internal sinks, one might expect halving flushing time to be equivalent to halving loads. Given that the mass balance for nitrogen and phosphorus is dominated by internal sinks rather than flushing and export, it seems likely that other factors may play a role. In particular, the increased exchange, by allowing increased ventilation of the bottom layer, may indirectly increase the effectiveness of the internal sinks. The integrated model, using exchanges derived from the hydrodynamic model, shows no evidence of the adverse impacts that might be expected to follow if the 2nd entrance caused increased stratification or reduced ventilation of the bottom layer. In fact, the opposite appears to be the case: in the box model, mean vertical mixing rates in L. King are increased rather than decreased.

The model predicts very substantial reductions in *Nodularia* blooms in L. King for E1. These reductions, to 11% of baseline, are much larger than those produced by the 50% load reduction L7 (to 37%), or even the 70% load reduction L8 (to 28%). It seems likely that this increased impact on *Nodularia* is due to increases in salinity resulting from the 2nd entrance. In this scenario, 95%ile salinities exceed 30 in L. Victoria and 34 in L. King, compared with 27 and 30 respectively in the baseline. These are certainly sufficient in the model to inhibit *Nodularia* growth. However this prediction would need to be confirmed by experimental salinity tolerance tests for strains of *Nodularia* from the Gippsland Lakes.

4.3.2 Altering tidal exchanges at the existing entrance.

Scenarios E2 and E3 are intended to simulate the effects of respectively shallowing and deepening the existing entrance, so as to reduce or increase tidal exchange. The box-model exchanges used in these runs are derived from hydrodynamic model simulations, in which the geometry of the channel at Lakes entrance was modified so as to achieve approximately a halving and doubling of the tidal prism. Walker and Andrewartha (2000) noted that this resulted in quite small changes in predicted salinity and flushing times in the main basins. They attributed this to the fact that, because of the small tidal prism, and the length of the channel joining Lakes Entrance to L. King, the tide contributes relatively little to flushing, which is dominated by freshwater outflow and estuarine circulation under high-flow conditions, and by low-frequency changes in sea-level under low-flow conditions.

The box-model exchanges derived from the hydrodynamic model via inverse techniques generally support this conclusion. Mean exchange rates between bottom-waters in southern L. King (box 2) and the entrance box (1) are respectively decreased or increased by about 10% in scenarios E2 and E3. Median salinities in L. King and L. Victoria are changed by less than 2%. However, the vertical exchange between surface and bottom layers in L. King is increased on average in scenario E2 (reduced tidal exchange), and decreased on average in scenario E3 (increased tidal exchange). It seems likely that this is due to periods of reduced stratification under E2. (This is not the case for the 2nd entrance scenario E1; there, increased tidal flushing leads to increased vertical mixing in L. King.)

The results for scenarios E2 and E3 appear to be explicable in terms of the interaction between horizontal and vertical exchanges. Changes in Chl a, surface phosphate, suspended sediment, Secchi depth, surface primary production, sediment respiration rates, MPB biomass are all small (< 10%), in keeping with the small percentage change in flushing rates. However, not all of these are in the expected direction: e.g. Chl a in L. King decreases under scenario E3, as we might expect for increased flushing, but also decreases under E2. Reasons for this become apparent when we consider some other variables that show larger responses to E2.

The 5%ile bottom oxygen concentrations increase substantially under E2, while bottom ammonia and phosphate concentrations decrease by up to 20%, and median denitrification efficiencies in L. Victoria increase by 40%. These variables all show much less response to E3. It seems likely that these responses reflect the impact of increased vertical mixing in E2. Surface phosphate concentrations also tend to increase in E2, but 95%ile surface ammonia shows a large relative decrease, reflecting a decrease in the extent of P-limitation.

Overall, the responses to E2 and E3 reflect the opposing effects of reduced horizontal exchange and overall flushing, but increased vertical mixing, in E2 compared with the baseline, and in the baseline compared with E3. For the most part, these responses are quite small, in keeping with the small change 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 increases in vertical mixing outweigh the effects of reduced overall flushing in E2.

4.4 Long-term Scenarios

Two long-term (40 year) simulations were carried out, nominally starting in July 1987, with output saved after the first 8 years (i.e. starting in July 95). Output statistics were computed for 4-year periods in years 23-26 (nominally July 2009 to June 2013), and years 37-40 (nominally July 2023 to June 2027). In the extended baseline run, the 1995-99 loads were repeated unchanged and these outputs are referred to as B14 and B28. In the long-term growth scenario, loads grew at a constant rate equivalent to 1% p.a after year 8, and the corresponding outputs are referred to as G14 and G28. These scenario results are plotted in Fig. 35-51.

The baseline run shows no long-term trends: almost all indicators show zero change in B14 and B28 from the 1995-99 baseline B (Fig. 35-51, Tables 1-30). As discussed in the model calibration report (Parslow et al., 2001), this should not be taken as a firm prediction that there will be no change in the Lakes under constant existing loads. Rather, it reflects the fact that the model does not include key processes in the sediments that might result in such long-term changes. We do not understand the processes affecting long-term accumulation of N and P in sediments well enough to model these with any confidence. The model is formulated and calibrated so as to reproduce the observed behaviour in the 1995-99 period, and to maintain this behaviour in the long-term.

The growth scenarios G14 and G28 also need to be interpreted in this light. As the model adapts to loads with a time scale of about 3 years, the model behaviour at G14 and G28 is similar to the response of the model to loads that are approximately 15% and 32% larger than baseline loads. Median chlorophyll concentrations at G14 and G28 have increased by about 12 and 30% respectively (Table 1). Primary production rates have increased by 13% to 24% at G14, and 35% to 47% at G28.

Median ammonium concentrations in surface waters (Table 5) have increased by only 1% to 15% at G14, but by 23% to 50% at G28. However, maximum ammonium concentrations in surface waters have increased by 4 times or more, suggesting extended periods of P-limitation in surface waters. Median bottom-water ammonium concentrations have increased by almost 60% at G14, and over 200% at G28. Median surface phosphate concentrations have decreased slightly, again suggesting more P-limitation, but bottom-water phosphate concentrations in L. Victoria have increased by 24% at G14, and 60% at G28.

Median bottom-water oxygen concentrations have declined by less than 10% at G14, and less than 30% at G28. Median denitrification efficiencies decline drastically in L. Victoria, to 62% of baseline at G14, and 27% of baseline at G28.

Suspended sediment concentrations have increased slightly less than loads, and Secchi depths have decreased by around 10% at G14, and 15% at G28. These result in small declines in MPB in L. Victoria and L. King, but in loss of macroalgae in L. Victoria.

This response to increases in loads more or less mirrors the response to decreases in loads in the load scenario L3. The positive feedback involving oxygen depletion in bottom waters, denitrification, and ammonia and phosphate release from sediments into bottom waters, amplifies the response of bottom-water and sediment indicators, especially in L. Victoria but also in L. King, to increases in loads. This results in turn in disproportionate increases in primary production. There are also negative impacts on benthic plants, due to reduced light penetration, with predicted loss of macroalgae from shallow parts of L. Victoria.

5 Conclusions

5.1 The Current Condition

The scenarios considered here all represent changes to the existing (baseline) condition. The baseline condition is analysed in considerable detail in the Implementation and Calibration Final Report (Parslow et al., 2001). We repeat some of the key conclusions from that report here, as they provide important background for assessment and interpretation of the scenario results.

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

During low flow periods, in summer and autumn, or in drought years, flushing rates are very low, with flushing times of order 6 months or longer. A large fraction of the load is retained and recycled within the Lakes, and internal sinks are as or more important than export in nutrient and sediment budgets.

Loads are highly variable in time. Peak events in L. King and L. Victoria flood surface waters with high nutrient concentrations, which result in large blooms. Base loads into L. Wellington are higher, and phytoplankton blooms there are more persistent.

While L. Wellington is vertically well-mixed, L. Victoria and L. King are stratified for much of the time. This makes those basins particularly vulnerable to eutrophication. There is a positive feedback loop in which organic matter settling into bottom waters drives oxygen consumption and bottom-water hypoxia, which leads to shut down of denitrification and reduction of Fe in sediments, leading in turn to high rates of ammonia and phosphate release, and increased organic matter production.

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

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

5.2 The response to loads.

Over the 1995-99 period, loads into L. Wellington were about 2 to 3 times those into L. King. Small (20%) reductions in western catchment loads have much more effect on L. Wellington and L. Victoria than reductions in eastern catchments, whereas reductions in western and in eastern catchments have comparable effects on L. King.

The response to small (20%) reductions in loads shows evidence of the positive feedback loop involving bottom-water hypoxia described above. Reductions in surface chlorophyll, TSS, and median ammonia concentrations are more or less proportional to load reductions, while bottom-water ammonia and phosphate, sediment denitrification efficiencies, and planktonic primary production, show amplified responses.

Despite this amplification, very substantial reductions in loads, of order 50 to 70%, would be required to eliminate episodes of bottom-water hypoxia, and associated shutdown of denitrification, and increased ammonia and DIP release from sediments. 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}.

The model predicts that the system shifts to become more strongly N-limited under reduced loads: recovery in denitrification outweighs any increase in P burial. Episodes of high surface ammonia disappear under load reductions of 50 to 70%.

Despite this shift, the model predicts reductions in *Nodularia* bloom intensity by 50 and 70%, more or less in proportion to loads, presumably as a result of reductions in the absolute supply rate of phosphate.

Despite reductions in phytoplankton biomass and suspended sediments, predicted light attenuation remains high, and there is little recovery of benthic plants in the deeper basins. This reflects the large contribution of CDOM to light attenuation in the model. CDOM concentrations in river inputs are assumed not to decline along with suspended sediment loads. This assumption has important implications for benthic plants, and needs to be tested. Certainly water draining from pristine catchments can carry very high concentrations of CDOM, but it is not clear whether changes in land management or water allocation would affect CDOM concentrations in rivers draining the Gippsland Lakes catchments.

Reductions in MID P and N loads by 40% represent only 10 and 3 % reductions respectively in western catchment TN and TP loads. These reductions have impacts in the model more or less commensurate with their magnitude. Reductions in MID loads do represent a reduction in P relative to N, and tend to push the system towards P-limitation, resulting in longer periods of high ammonia in surface waters.

5.3 The response to flows.

In the flow scenarios, river flows are changed while leaving nutrient and sediment loads unaltered. These scenarios therefore simulate the physical effects of changes in freshwater input, circulation and stratification. The dominant effect appears to be the effect on flushing: reduced flows result in reduced flushing and increased eutrophication, while increased flows have the opposite effect.

One might expect increased flows to increase stratification and reduce vertical mixing, but instead they appear to increase ventilation of the bottom layer, presumably through increased estuarine circulation.

As for load scenarios, bottom-water nutrients and sediment denitrification efficiencies are most sensitive to changes in flow. Surface chlorophyll and primary production show smaller responses to 20% changes in flow than to 20% reductions in loads.

While a 20% increase in flows into L. Wellington generally improves water quality, it does increase the predicted delivery of TSS to L. Victoria and results in increases in predicted TSS and small decreases in predicted Secchi depth in L. Victoria and L. King. Given the limitations of the suspended sediment model, this prediction needs further examination. Because CDOM concentrations in freshwater are treated as constant, increased flows also increase light attenuation due to CDOM.

5.4 The response to marine exchanges.

In the model, the 2nd entrance (scenario E1) has a surprisingly large effect on water quality in Lakes Victoria and King, although it has more or less negligible effects on L. Wellington. In the hydrodynamic model, the imposition of a second entrance approximately halved flushing times. In the integrated model, it is approximately equivalent in its effect on water quality in Lakes Victoria and King to a 50% reduction in catchment loads.

This result might be expected in a system where loads are predominantly balanced by exports, but is unexpected in the Gippsland Lakes, where substantial fractions of N and P loads are lost to internal sinks. It may be that the impact of increased flushing is high during extended periods of bottom-water hypoxia, when internal sinks are shut down.

The positive feedback involving bottom-water hypoxia in L. King and L. Victoria undoubtedly amplifies the effect of the 2nd entrance. Bottom-water oxygen values are increased very substantially in this scenario. It is noteworthy that, in the model, the additional marine exchanges do not lead to increased stratification and reduced exchanges, but rather to an overall increase in vertical mixing and ventilation of the bottom layer (in contrast with scenarios E2 and E3 below).

The 2nd entrance also leads to substantial increases in Secchi depth, due primarily to increased salinities and reduced CDOM concentrations. This is insufficient to increase phytobenthos substantially in the deep basins, but would be expected to benefit benthic plants in the margins.

The model also predicts very substantial reductions in *Nodularia* bloom intensities in L. King, due to increased salinities as well as decreased phosphate supply. The 2nd entrance results in increases in salinity by about 3 to 4 in L. King and L. Victoria, and this is sufficient in the model to inhibit *Nodularia* growth in most summer-autumn periods. The salinity-dependence assumed in the model needs to be confirmed by salinity tolerance tests for strains of *Nodularia* isolated from the Gippsland Lakes.

By comparison, modifications to the existing entrance have relatively small effects on water quality. Tidal exchanges contribute relatively little to flushing of the Lakes in the hydrodynamic model. Approximately halving and doubling tidal exchanges reduces and increases exchanges between L. King and the entrance by about 10%. However, in this case the model predicts that increased horizontal exchanges are accompanied by reduced vertical mixing, presumably due to small increases in stratification. The effects of reduced flushing and increased vertical mixing tend to have opposing effects on water quality in L. King and L. Victoria.

These changes to the existing entrance generally have small effects on water quality parameters, in keeping with the small change in flushing rates. However, indicators such as bottom oxygen and nutrient concentrations, which are sensitive to vertical mixing rates, show larger responses, especially to the increased vertical mixing predicted for reduced tidal exchanges (shallower entrance).

5.5 Long-term (30 year) response.

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. As calibrated, the model adapts to nutrient loads with a time scale of about 3 years, and shows effectively no change in model indicators over a 40-year baseline run.

The results obtained at the mid-point (G14) and end (G28) of a 40-year run, with 1% growth rate for nutrient and sediment loads after year 8, are therefore comparable with results for the model run to a steady cycle with approximately a 15% and 30% increase in baseline loads.

The response to these increases in loads more or less mirrors the response to 20% decreases in loads. While median chlorophyll increases more or less in proportion to loads, the positive feedback in bottom waters amplifies the impact on bottom-water ammonium and phosphate concentrations, with median bottom-water ammonium concentrations doubling after 28 years, while median denitrification efficiencies in L. Victoria decline by almost a factor of 4.

Increases in suspended sediment concentrations and reductions in Secchi depth are sufficient to lead to predicted loss of macroalgae even in the shallower western basin in L. Victoria.

5.6 Model Limitations.

Model limitations and uncertainties are discussed at length in the Implementation and Calibration Report, and are referred to at appropriate points throughout the preceding discussion and conclusions. In this section, we summarise these limitations, focusing on their implications for the interpretation of scenario results.

5.6.1 Sediment biogeochemistry

Perhaps the most critical uncertainties are those associated with long-term processes controlling storage of P (and to a lesser extent N) in sediments, and release of N and P from those pools. As was just noted, the model is consequently unable to address questions about the long-term consequences of maintaining existing loads, and limited in its ability to predict the likely rate of recovery if loads are reduced.

The integrated model as formulated and calibrated predicts quite rapid recovery following reductions in loads, as the longest time scales in the model sediment dynamics are around 3 years. 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 Lakes Victoria and King, during this period, and the model is calibrated so as to reproduce this improvement. Of course, it must be noted that the drought period corresponds to a reduction in flow and stratification, as well as a reduction in loads.

There is some evidence from the model calibration that the existing model may not differentiate sufficiently between basins. It appears that predicted denitrification efficiencies may be too high in L. Wellington, so that blooms run down too quickly after flood events, and too low in L. Victoria, where bottom ammonia concentrations are too high. If this is the case, the model may underestimate the sensitivity of L. Wellington, and overestimate the sensitivity of L. Victoria, to changes in loads.

It also appears that DIP burial rates may be too low in L. Wellington relative to L. Victoria and L. King, and that surface DIP is less well buffered in the model than in reality. If so, the balance between N and P-limitation in the model may be too sensitive to changes in loads. On the other hand, observations of high surface ammonia in the long-term EPA data sets suggest that periods of P-limitation may have occurred in the past.

5.6.2 Phytoplankton dynamics

The attempt to represent four phytoplankton functional groups in the model is arguably pushing the limits of our existing understanding and knowledge. In particular, the model is unable to capture the timing of dinoflagellate blooms, and it seems likely that the assumption of a common zooplankton grazer for diatoms and dinoflagellates may be responsible. Consequently, predicted changes in dinoflagellate abundance under different load scenarios should be treated with caution.

The model predictions about responses of *Nodularia* blooms to different load and exchange regimes should also be treated cautiously. The model attempts to capture temperature and salinity tolerances described in the literature, but these need to be confirmed for *Nodularia* strains from the Gippsland Lakes. Thus, the predicted reduction in *Nodularia* bloom densities with a 2nd entrance, partly due to increased salinity, is plausible, given the experience in Peel-Harvey, but should not be treated as conclusive. In the model, *Nodularia* is reliant on phosphate in surface waters, and is unable to access bottom-water phosphate directly through vertical migration (buoyancy control). If this assumption is incorrect, it could substantially change the potential magnitude of blooms. Finally, the model does not attempt to represent the processes controlling akinete germination, and so is unlikely to reproduce interannual variation in bloom magnitude, unless this is simply controlled by growth conditions. The fact that the model tends to predict highest bloom densities in L. Wellington, whereas blooms are commonly observed in L. King, strongly suggests that key processes are missing.

While these interactions among phytoplankton functional groups are poorly understood, and the model almost certainly predicts the wrong phytoplankton composition at times, it does not follow that model predictions of more general water and sediment quality indicators are subject to the same uncertainty or error. Biogeochemically, phytoplankton play the role of converting available nutrients into organic matter where there is sufficient light. Provided the overall phytoplankton assemblage is capable of doing that, and achieving approximately the right biomass and turnover rates, then the cycling of nutrients and organic matter will be captured reasonably well. The phytoplankton and zooplankton assemblage in the model does keep surface nutrients depleted most of the time, as observed, and allows bottom nutrients to accumulate under stratified conditions, also as observed. While it continues to do this, the overall response of the model to changes in loads is controlled more by changes in sediment sinks than by shifts in phytoplankton composition.

The key exception to this is of course N-fixation: to the extent that the model underestimates cyanobacterial blooms and associated N-fixation, it will underestimate N inputs. The model is calibrated to reproduce *Nodularia* blooms of approximately the magnitude observed during the calibration period, and also incorporates levels of N-fixation in keeping with those blooms. The long-term EPA data set records isolated blooms with much higher biomass, and potentially much higher levels of N-fixation. However, it is worth noting that, because *Nodularia* is positively buoyant, surface samples may greatly overestimate its depth-averaged concentration, making it very difficult to quantify its role in the overall nitrogen budget.

The model predicts that there is a relatively fine balance between N and P-limitation in the Gippsland Lakes, although the system is N-limited most of the time. Provided N and P loads are reduced together, it seems likely that the predicted improvements in water quality will be achieved, regardless of *Nodularia* responses. However, if N loads were reduced substantially without reductions in P loads, N-fixation by *Nodularia* might play a more critical role, undermining the predicted improvement.

5.6.3 Physical transport

The estimated physical exchanges in the integrated model are derived from the hydrodynamic model. Physical exchanges could be in error either because the hydrodynamic model predictions are incorrect, or because the coarse horizontal and vertical resolution in the box model does not adequately capture the hydrodynamic model behaviour.

The calibration of the hydrodynamic model is discussed elsewhere. The model reproduces water levels in the Lakes, including tides and low-frequency oscillations, very well. The salinity regime during the calibration period, with intense flood events and long periods of low flows, provides a good basis for calibrating flushing, and the hydrodynamic and box models' reproductions of the salinity regime are sufficiently accurate to provide good confidence in the modelled flushing rates.

Vertical mixing is arguably a more difficult process to both simulate and calibrate. Lakes King and Victoria are stratified almost all the time, even under conditions of prolonged low flow. This implies that vertical mixing rates are very low, and/or that stratification is maintained by small inputs of fresh water at the surface, and salt water at depth, through the entrance. There is potentially some ambiguity in modelled vertical mixing rates in both the hydrodynamic and especially the box model, as vertical mixing rates can be traded off against marine inputs. The hydrodynamic model does not reproduce salinity stratification perfectly, especially during transients following flood events. Box-model exchanges based on direct inversion of salinity observations predict less vertical mixing than exchanges based on hydrodynamic model output.

The model variables that are most sensitive to vertical mixing rates are bottom-water oxygen and nutrient levels. The integrated model has of course been calibrated to reproduce bottom-water oxygen and nutrient conditions under baseline conditions, with the baseline exchanges.

Vertical mixing rates do have significant effects on the model response to some scenarios, particular those involving deepening and shallowing of the existing entrance. While this may add some additional uncertainty, these scenarios generally showed rather small overall impacts on water quality, and this is likely to be a robust conclusion, given the small contribution of tidal exchange to flushing.

5.6.4 Composition of loads

Finally, we lack information about some aspects of the composition of N and P loads. The sensitivity of the model to these assumptions is reduced by the long residence times. However, the response of the model to loads, and changes in loads, could change if there were accompanying changes in the fractions of TN and TP loads that are highly refractory or permanently unavailable.

In scenarios, we have assumed that suspended sediment loads change along with TN and TS. However, we have assumed that CDOM concentrations in freshwater remain fixed. These assumptions have implications for light attenuation in particular. If reductions in loads and TSS were accompanied by proportional reductions in CDOM, light attenuation would be significantly reduced, resulting in improved conditions for plant growth in bottom waters in L. Victoria and L. King.

5.7 Management Response to Uncertainty

Model scenario results of course represent just one of many inputs into management decisions. Managers need to agree on management objectives, and indicators, and to identify practical alternative management actions. Decisions will need to take account of the costs of alternative actions, the predicted benefits, and the uncertainties in those predictions. For the most part, the scenarios run here do not represent specific practical management actions, but are intended to provide background information for formulating management options. At the least, they might be used to frame feasible objectives, to identify the magnitude of the task involved in meeting those objectives, and to rule out actions that are unlikely to be worth pursuing further. There is little point in spending large amounts of time and resources in detailed planning and implementation of management actions that would produce negligible progress towards objectives.

As more detailed management options are identified, there are two approaches that might be adopted to deal with model uncertainty. If management actions involving a large one-off capital expenditure, such as engineering works, were to be seriously contemplated, then it might be worth considerable further investment in research or engineering studies to reduce model uncertainty, prior to committing to the capital works. This might involve investment in more detailed engineering models, to assess physical feasibility and costs. It might involve additional field and laboratory studies, and subsequent model enhancement, to address key uncertainties in the integrated model predictions.

Alternatively, if management actions are likely to involve a series of incremental actions, then there may be less value in upfront investments to reduce uncertainties further. It is arguably more important to identify and agree on a monitoring program for key indicators, which will allow managers to assess progress towards objectives, and modify actions over time in response to system changes. In this kind of *adaptive management*, the model can be used to evaluate alternative monitoring and management strategies up front. In turn, feedback from the monitoring program can be used to assess and revise the model along the way.

It seems likely that such an adaptive management approach will be appropriate if managers choose to reduce catchment loads. Reductions in loads of the order of 50 to 70% seem unlikely to be achieved easily or quickly, and substantial improvements in water quality are still possible with more modest reductions. There is a need to identify alternative actions in the catchment, and assess their likely costs and benefits in terms of load reductions. This may require further catchment studies. An adaptive management program might start by implementing those actions that are most attractive (i.e. "biggest bang for the buck"), and proceed to less attractive options once monitoring makes it clear these are necessary.

6 References

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7 Tables

Table 1. Median Chl a concentrations in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg Chl a m^{-3}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	8.2	13.6	9.1	1.00	1.00	1.00
L1	8.1	12.4	8.7	0.99	0.91	0.96
L2	6.8	10.8	8.9	0.83	0.79	0.97
L3	6.8	10.0	8.7	0.83	0.74	0.96
L4	5.5	9.3	9.1	0.67	0.68	1.00
L5	8.1	12.6	8.9	0.99	0.92	0.98
L6	7.9	12.2	8.9	0.96	0.89	0.98
L7	4.6	5.7	5.2	0.56	0.42	0.57
L8	3.1	4.2	4.0	0.38	0.31	0.44
F1	9.1	14.4	9.4	1.11	1.06	1.03
F2	8.2	14.0	8.9	1.01	1.02	0.98
F3	7.5	12.4	9.0	0.92	0.91	0.99
E1	8.0	7.5	4.7	0.98	0.55	0.52
E2	8.1	12.5	8.8	0.99	0.92	0.97
E3	8.2	12.7	8.9	1.00	0.93	0.98
B14	8.2	13.6	9.1	1.00	1.00	1.00
B28	8.2	13.6	9.1	1.00	1.00	1.00
G14	9.2	15.0	10.1	1.13	1.10	1.11
G28	10.1	19.0	11.2	1.24	1.39	1.23

Table 2. 95%ile Chl a concentrations in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg Chl a m^{-3}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	33.6	34.9	26.8	1.00	1.00	1.00
L1	33.5	33.4	24.3	1.00	0.96	0.91
L2	24.9	24.4	24.5	0.74	0.70	0.91
L3	24.9	24.6	23.3	0.74	0.71	0.87
L4	18.1	22.2	22.7	0.54	0.64	0.85
L5	33.2	33.4	24.7	0.99	0.96	0.92
L6	32.8	33.2	24.0	0.98	0.95	0.89
L7	15.5	12.0	10.7	0.46	0.35	0.40
L8	10.9	8.3	8.3	0.32	0.24	0.31
F1	37.6	37.9	25.7	1.12	1.09	0.96
F2	33.6	35.9	24.0	1.00	1.03	0.89
F3	31.2	31.6	25.2	0.93	0.90	0.94
E1	33.6	22.5	13.1	1.00	0.64	0.49
E2	33.4	32.1	24.5	1.00	0.92	0.91
E3	33.6	33.2	24.9	1.00	0.95	0.93
B14	33.5	34.9	26.9	1.00	1.00	1.00
B28	33.5	34.9	26.9	1.00	1.00	1.00
G14	42.3	45.3	26.1	1.26	1.30	0.97
G28	49.6	54.6	30.3	1.48	1.57	1.13

Table 3. 95%ile Dinoflagellate biomass (Chl a) in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg Chl a m-3			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	0.2	13.0	11.7	1.00	1.00	1.00
L1	0.3	14.2	11.1	1.48	1.09	0.95
L2	0.3	13.5	12.7	1.44	1.04	1.08
L3	0.3	13.5	13.1	1.47	1.03	1.12
L4	0.4	13.5	12.1	1.77	1.03	1.03
L5	0.2	11.1	11.0	0.78	0.85	0.94
L6	0.2	10.4	10.6	0.76	0.80	0.90
L7	0.2	6.8	5.7	1.07	0.52	0.48
L8	0.4	5.0	4.7	1.69	0.38	0.40
F1	0.3	12.6	9.7	1.18	0.97	0.83
F2	0.2	10.8	10.3	0.94	0.83	0.87
F3	0.2	10.7	13.8	0.75	0.82	1.17
E1	0.1	7.1	4.6	0.47	0.54	0.39
E2	0.1	10.4	10.4	0.61	0.80	0.88
E3	0.2	12.4	11.1	1.12	0.95	0.95
B14	0.2	13.1	11.8	1.01	1.00	1.01
B28	0.2	13.0	11.8	1.01	1.00	1.01
G14	0.0	1.3	2.6	0.11	0.10	0.22
G28	0.0	0.0	0.0	0.00	0.00	0.00

Table 4. 95%ile *Nodularia* biomass (Chl a) in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg Chl a m^{-3}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	1.0	4.2	6.7	1.00	1.00	1.00
L1	1.0	4.5	6.2	1.03	1.07	0.93
L2	0.8	3.3	5.9	0.80	0.78	0.88
L3	1.0	3.9	6.3	0.95	0.93	0.95
L4	1.3	4.2	5.7	1.29	1.00	0.85
L5	0.7	3.3	5.4	0.75	0.79	0.81
L6	0.7	3.1	5.2	0.68	0.73	0.78
L7	0.7	2.2	2.5	0.74	0.52	0.37
L8	0.4	1.3	1.9	0.44	0.32	0.28
F1	0.8	5.4	7.0	0.81	1.28	1.04
F2	0.8	3.5	6.8	0.75	0.84	1.02
F3	0.7	3.3	4.9	0.74	0.79	0.73
E1	0.7	1.1	0.7	0.66	0.26	0.11
E2	0.6	2.9	5.4	0.55	0.69	0.80
E3	1.1	4.7	6.6	1.14	1.11	0.98
B14	1.0	4.2	6.7	1.00	1.00	1.01
B28	1.0	4.2	6.7	1.00	1.00	1.01
G14	0.7	3.2	6.6	0.67	0.77	0.98
G28	0.5	3.8	7.6	0.50	0.90	1.13

Table 5. 50%ile ammonia concentration in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg N m^{-3}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	0.9	1.3	1.0	1.00	1.00	1.00
L1	0.9	1.2	0.9	0.98	0.94	0.87
L2	0.7	1.1	0.8	0.83	0.86	0.85
L3	0.7	1.1	0.8	0.82	0.80	0.78
L4	0.6	1.0	0.8	0.66	0.74	0.81
L5	0.9	1.4	1.0	1.00	1.02	0.99
L6	0.9	1.3	0.9	0.96	0.98	0.93
L7	0.5	0.4	0.3	0.55	0.31	0.34
L8	0.4	0.2	0.2	0.39	0.16	0.21
F1	1.0	1.3	1.0	1.16	1.02	0.99
F2	0.9	1.3	0.9	1.02	0.99	0.94
F3	0.8	1.2	0.9	0.89	0.93	0.87
E1	0.8	0.5	0.3	0.90	0.41	0.34
E2	0.9	1.3	0.9	0.99	0.95	0.88
E3	0.9	1.3	0.9	0.99	0.97	0.93
B14	0.9	1.3	1.0	1.00	1.01	1.02
B28	0.9	1.3	1.0	1.00	1.01	1.01
G14	1.0	1.4	1.0	1.15	1.02	0.99
G28	1.2	1.6	1.5	1.30	1.23	1.49

Table 6. 95%ile ammonia concentration in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg N m^{-3}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	6.7	27.0	10.8	1.00	1.00	1.00
L1	8.0	18.2	3.9	1.20	0.67	0.36
L2	3.1	3.6	4.2	0.47	0.13	0.39
L3	3.2	2.7	2.2	0.49	0.10	0.20
L4	1.6	2.1	3.4	0.24	0.08	0.31
L5	34.1	37.4	16.8	5.11	1.38	1.55
L6	32.0	32.3	14.4	4.80	1.19	1.33
L7	1.2	1.1	0.8	0.19	0.04	0.07
L8	0.7	0.5	0.4	0.11	0.02	0.03
F1	8.7	29.3	14.0	1.31	1.08	1.30
F2	6.7	22.2	6.8	1.00	0.82	0.63
F3	12.3	15.8	3.5	1.85	0.58	0.33
E1	11.2	2.0	1.2	1.67	0.08	0.11
E2	6.5	9.7	2.7	0.98	0.36	0.25
E3	7.3	27.7	11.1	1.09	1.02	1.03
B14	6.7	31.4	14.4	1.00	1.16	1.33
B28	6.7	31.4	14.4	1.00	1.16	1.33
G14	12.0	31.8	26.3	1.80	1.18	2.43
G28	27.6	70.2	45.0	4.13	2.60	4.16

Table 7. 50%ile ammonia concentration in bottom waters of Lakes Victoria and King, absolute, and relative to baseline.

	mg N m^{-3}		Relative to baseline	
Scenario	Victoria	King	Victoria	King
B	235.7	126.7	1.00	1.00
L1	167.9	84.5	0.71	0.67
L2	120.8	61.7	0.51	0.49
L3	85.0	37.5	0.36	0.30
L4	37.3	23.2	0.16	0.18
L5	216.6	105.6	0.92	0.83
L6	204.7	98.1	0.87	0.77
L7	4.8	3.7	0.02	0.03
L8	2.4	2.1	0.01	0.02
F1	335.5	175.8	1.42	1.39
F2	273.5	147.6	1.16	1.16
F3	160.6	73.9	0.68	0.58
E1	8.5	3.1	0.04	0.02
E2	219.2	105.7	0.93	0.83
E3	214.9	106.1	0.91	0.84
B14	236.6	127.4	1.00	1.01
B28	236.5	127.5	1.00	1.01
G14	367.0	200.1	1.56	1.58
G28	492.0	274.8	2.09	2.17

Table 8. 95%ile ammonia concentration in bottom waters of Lakes Victoria and King, absolute, and relative to baseline.

	mg N m^{-3}		Relative to baseline	
Scenario	Victoria	King	Victoria	King
B	690.1	329.7	1.00	1.00
L1	648.7	288.4	0.94	0.87
L2	427.3	243.4	0.62	0.74
L3	355.3	182.9	0.51	0.55
L4	301.7	207.0	0.44	0.63
L5	673.6	306.1	0.98	0.93
L6	655.4	297.1	0.95	0.90
L7	81.6	29.5	0.12	0.09
L8	16.8	5.8	0.02	0.02
F1	841.4	421.0	1.22	1.28
F2	721.8	355.5	1.05	1.08
F3	575.9	273.6	0.83	0.83
E1	280.2	105.4	0.41	0.32
E2	618.0	282.3	0.90	0.86
E3	694.9	332.7	1.01	1.01
B14	689.1	330.4	1.00	1.00
B28	688.9	330.4	1.00	1.00
G14	887.8	444.9	1.29	1.35
G28	1054.3	537.9	1.53	1.63

Table 9. 50%ile phosphate concentration in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg P m^{-3}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	27.5	13.0	8.3	1.00	1.00	1.00
L1	27.2	14.0	9.8	0.99	1.07	1.18
L2	26.3	12.3	8.9	0.96	0.94	1.07
L3	25.5	11.9	9.2	0.93	0.91	1.10
L4	23.0	11.5	8.8	0.84	0.88	1.06
L5	21.9	10.5	7.5	0.80	0.81	0.90
L6	22.3	11.3	7.9	0.81	0.87	0.95
L7	21.5	13.8	10.8	0.78	1.06	1.30
L8	17.2	12.4	10.0	0.63	0.95	1.20
F1	29.4	13.5	8.8	1.07	1.04	1.05
F2	27.5	13.3	8.6	1.00	1.02	1.03
F3	25.5	13.6	9.4	0.93	1.04	1.12
E1	26.7	16.8	14.5	0.97	1.29	1.74
E2	27.2	12.4	9.1	0.99	0.96	1.10
E3	27.3	13.5	8.7	1.00	1.04	1.04
B14	27.1	12.9	8.2	0.99	0.99	0.99
B28	27.1	12.9	8.2	0.99	0.99	0.99
G14	26.9	11.7	7.3	0.98	0.90	0.87
G28	25.8	10.3	4.6	0.94	0.79	0.56

Table 10. 95%ile phosphate concentration in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg P m^{-3}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	42.5	35.0	21.7	1.00	1.00	1.00
L1	41.5	36.9	24.6	0.98	1.06	1.14
L2	38.5	41.7	24.6	0.91	1.19	1.14
L3	38.3	37.0	21.3	0.90	1.06	0.99
L4	33.0	27.9	18.5	0.78	0.80	0.85
L5	34.0	30.9	20.0	0.80	0.88	0.92
L6	33.5	32.1	20.2	0.79	0.92	0.93
L7	30.6	22.1	17.7	0.72	0.63	0.82
L8	23.1	18.5	15.4	0.54	0.53	0.71
F1	47.2	39.0	24.6	1.11	1.11	1.14
F2	42.5	38.9	24.5	1.00	1.11	1.13
F3	38.6	33.4	20.8	0.91	0.95	0.96
E1	44.5	29.0	21.2	1.05	0.83	0.98
E2	42.5	38.4	22.6	1.00	1.10	1.04
E3	42.3	32.0	20.9	1.00	0.91	0.96
B14	42.0	35.1	21.6	0.99	1.00	1.00
B28	42.0	35.1	21.6	0.99	1.00	1.00
G14	42.8	36.5	25.0	1.01	1.04	1.16
G28	43.5	41.8	25.0	1.02	1.19	1.15

Table 11. 50%ile phosphate concentration in bottom waters of Lakes Victoria and King, absolute, and relative to baseline.

	mg P m^{-3}		Relative to baseline	
Scenario	Victoria	King	Victoria	King
B	37	29	1.00	1.00
L1	30	26	0.82	0.88
L2	25	24	0.68	0.81
L3	24	20	0.64	0.68
L4	20	16	0.54	0.56
L5	32	26	0.86	0.88
L6	31	25	0.82	0.87
L7	17	14	0.47	0.49
L8	15	12	0.40	0.43
F1	43	34	1.16	1.17
F2	37	31	1.00	1.08
F3	29	24	0.79	0.82
E1	23	18	0.61	0.63
E2	35	29	0.93	0.99
E3	35	27	0.93	0.93
B14	37	29	0.99	0.99
B28	37	29	0.99	0.99
G14	46	32	1.24	1.10
G28	60	35	1.61	1.22

Table 12. 95%ile phosphate concentration in bottom waters of Lakes Victoria and King, absolute, and relative to baseline.

	mg P m^{-3}		Relative to baseline	
Scenario	Victoria	King	Victoria	King
B	141	69	1.00	1.00
L1	122	57	0.87	0.82
L2	99	46	0.70	0.66
L3	83	41	0.59	0.59
L4	64	33	0.46	0.48
L5	131	62	0.93	0.90
L6	128	60	0.91	0.86
L7	25	22	0.17	0.32
L8	20	18	0.14	0.26
F1	173	86	1.23	1.24
F2	148	74	1.05	1.06
F3	114	56	0.81	0.81
E1	50	24	0.36	0.34
E2	135	59	0.96	0.84
E3	135	65	0.96	0.93
B14	141	70	1.00	1.00
B28	141	70	1.00	1.00
G14	185	92	1.31	1.33
G28	218	110	1.55	1.59

Table 13. 50%ile oxygen concentration in bottom waters of Lakes Victoria and King, absolute, and relative to baseline.

	mg O m^{-3}		Relative to baseline	
Scenario	Victoria	King	Victoria	King
B	2913	4753	1.00	1.00
L1	3239	4981	1.11	1.05
L2	3632	5042	1.25	1.06
L3	3953	5339	1.36	1.12
L4	4225	5250	1.45	1.10
L5	3084	4830	1.06	1.02
L6	3159	4842	1.08	1.02
L7	6613	7217	2.27	1.52
L8	7819	7996	2.68	1.68
F1	2604	4332	0.89	0.91
F2	3074	4764	1.06	1.00
F3	3363	4882	1.15	1.03
E1	6375	7468	2.19	1.57
E2	3282	5109	1.13	1.07
E3	2928	4622	1.01	0.97
B14	2914	4756	1.00	1.00
B28	2913	4756	1.00	1.00
G14	2669	4444	0.92	0.93
G28	2111	4031	0.72	0.85

Table 14. 5%ile oxygen concentration in bottom waters of Lakes Victoria and King, absolute, and relative to baseline.

	mg O m^{-3}		Relative to baseline	
Scenario	Victoria	King	Victoria	King
B	584	1910	1.00	1.00
L1	641	2082	1.10	1.09
L2	672	1839	1.15	0.96
L3	723	1936	1.24	1.01
L4	775	1999	1.33	1.05
L5	653	2049	1.12	1.07
L6	696	2090	1.19	1.09
L7	3405	5152	5.83	2.70
L8	5729	6502	9.82	3.40
F1	486	1784	0.83	0.93
F2	637	1999	1.09	1.05
F3	778	2201	1.33	1.15
E1	2380	5044	4.08	2.64
E2	793	2430	1.36	1.27
E3	583	1915	1.00	1.00
B14	581	1907	1.00	1.00
B28	581	1908	1.00	1.00
G14	537	1966	0.92	1.03
G28	384	1699	0.66	0.89

Table 15. 50%ile suspended sediment concentration in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	g TSS m^{-3}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	38.9	10.8	4.8	1.00	1.00	1.00
L1	38.9	10.5	4.5	1.00	0.97	0.93
L2	31.7	9.3	4.3	0.82	0.86	0.89
L3	31.6	8.9	3.9	0.81	0.82	0.82
L4	24.2	7.5	3.7	0.62	0.70	0.77
L5	38.9	10.9	4.8	1.00	1.00	1.00
L6	39.0	10.9	4.8	1.00	1.01	1.00
L7	20.3	5.8	2.6	0.52	0.53	0.53
L8	12.4	3.5	1.6	0.32	0.33	0.33
F1	42.2	9.7	4.2	1.09	0.89	0.87
F2	38.8	11.0	4.8	1.00	1.01	1.00
F3	35.8	11.8	5.3	0.92	1.09	1.11
E1	35.9	10.0	3.4	0.92	0.93	0.71
E2	39.1	11.1	5.3	1.01	1.03	1.11
E3	38.6	10.5	4.4	0.99	0.97	0.92
B14	38.9	10.9	4.8	1.00	1.00	1.00
B28	38.9	10.9	4.8	1.00	1.00	1.00
G14	44.1	12.2	5.4	1.14	1.13	1.13
G28	48.5	13.5	6.0	1.25	1.25	1.24

Table 16. 95%ile suspended sediment concentration in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	g TSS m^{-3}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	56.8	33.3	15.8	1.00	1.00	1.00
L1	56.8	32.3	14.3	1.00	0.97	0.90
L2	46.1	27.9	14.4	0.81	0.84	0.91
L3	46.1	26.9	12.8	0.81	0.81	0.81
L4	35.1	22.1	13.2	0.62	0.66	0.84
L5	56.8	33.3	15.8	1.00	1.00	1.00
L6	56.9	33.4	15.9	1.00	1.00	1.00
L7	29.4	17.2	8.1	0.52	0.52	0.51
L8	17.9	10.5	4.9	0.32	0.32	0.31
F1	62.4	32.0	14.9	1.10	0.96	0.94
F2	56.8	33.4	16.3	1.00	1.00	1.03
F3	51.6	33.4	16.7	0.91	1.00	1.05
E1	55.1	31.5	15.7	0.97	0.95	0.99
E2	56.9	33.5	16.2	1.00	1.01	1.03
E3	56.6	33.0	15.5	1.00	0.99	0.98
B14	56.9	33.3	15.8	1.00	1.00	1.00
B28	56.9	33.3	15.8	1.00	1.00	1.00
G14	65.0	38.1	18.2	1.15	1.15	1.15
G28	71.2	42.0	20.1	1.25	1.26	1.27

Table 17. 50%ile Secchi depth in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	m			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	0.6	1.1	1.5	1.00	1.00	1.00
L1	0.6	1.1	1.5	1.00	1.04	1.03
L2	0.7	1.2	1.6	1.11	1.09	1.06
L3	0.7	1.2	1.6	1.11	1.12	1.08
L4	0.8	1.3	1.6	1.25	1.18	1.10
L5	0.6	1.1	1.5	1.00	1.01	1.01
L6	0.6	1.1	1.5	1.01	1.02	1.02
L7	0.9	1.5	1.8	1.34	1.32	1.24
L8	1.0	1.6	2.0	1.55	1.46	1.33
F1	0.6	1.1	1.6	0.96	1.02	1.07
F2	0.6	1.1	1.5	1.00	0.99	1.02
F3	0.7	1.1	1.4	1.04	0.99	0.98
E1	0.7	1.4	2.2	1.07	1.30	1.47
E2	0.6	1.1	1.5	1.00	1.01	1.01
E3	0.6	1.1	1.5	1.01	1.03	1.03
B14	0.6	1.1	1.5	1.00	1.00	1.00
B28	0.6	1.1	1.5	1.00	1.00	1.00
G14	0.6	1.0	1.4	0.93	0.92	0.97
G28	0.6	0.9	1.4	0.88	0.86	0.93

Table 18. 5%ile Secchi depth in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	m			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	0.5	0.6	0.9	1.00	1.00	1.00
L1	0.5	0.7	0.9	1.00	1.03	1.05
L2	0.5	0.7	0.9	1.15	1.11	1.04
L3	0.5	0.7	1.0	1.15	1.14	1.10
L4	0.6	0.8	0.9	1.34	1.25	1.09
L5	0.5	0.6	0.9	1.00	1.00	1.00
L6	0.5	0.6	0.9	1.00	1.01	1.00
L7	0.7	0.9	1.1	1.46	1.40	1.27
L8	0.8	1.0	1.2	1.77	1.61	1.43
F1	0.4	0.6	0.9	0.93	1.01	1.02
F2	0.5	0.6	0.9	1.00	1.00	1.00
F3	0.5	0.6	0.9	1.06	1.00	0.98
E1	0.5	0.7	1.0	1.01	1.12	1.18
E2	0.5	0.6	0.9	1.00	1.00	0.99
E3	0.5	0.6	0.9	1.00	1.01	1.01
B14	0.5	0.6	0.9	1.00	1.00	1.00
B28	0.5	0.6	0.9	1.00	1.00	1.00
G14	0.4	0.6	0.8	0.90	0.91	0.93
G28	0.4	0.5	0.8	0.84	0.85	0.87

Table 19. 50%ile primary production in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg C m^{-2} d^{-1}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	295	431	340	1.00	1.00	1.00
L1	292	381	294	0.99	0.89	0.86
L2	233	312	303	0.79	0.72	0.89
L3	231	275	251	0.78	0.64	0.74
L4	180	222	252	0.61	0.52	0.74
L5	293	391	321	0.99	0.91	0.94
L6	280	379	320	0.95	0.88	0.94
L7	142	91	95	0.48	0.21	0.28
L8	96	49	54	0.32	0.11	0.16
F1	339	487	364	1.15	1.13	1.07
F2	297	466	340	1.01	1.08	1.00
F3	264	376	308	0.90	0.87	0.90
E1	282	145	106	0.96	0.34	0.31
E2	292	420	341	0.99	0.97	1.00
E3	296	397	313	1.00	0.92	0.92
B14	295	430	341	1.00	1.00	1.00
B28	295	430	341	1.00	1.00	1.00
G14	347	534	385	1.18	1.24	1.13
G28	399	633	459	1.35	1.47	1.35

Table 20. 95%ile primary production in surface waters of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg C m^{-2} d^{-1}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	1073	1711	908	1.00	1.00	1.00
L1	1074	1553	813	1.00	0.91	0.90
L2	829	1249	743	0.77	0.73	0.82
L3	827	1045	665	0.77	0.61	0.73
L4	600	805	610	0.56	0.47	0.67
L5	1072	1656	884	1.00	0.97	0.97
L6	1038	1630	866	0.97	0.95	0.95
L7	473	372	312	0.44	0.22	0.34
L8	261	168	170	0.24	0.10	0.19
F1	1287	1923	959	1.20	1.12	1.06
F2	1071	1804	914	1.00	1.05	1.01
F3	945	1566	791	0.88	0.92	0.87
E1	1076	948	516	1.00	0.55	0.57
E2	1081	1701	919	1.01	0.99	1.01
E3	1070	1631	873	1.00	0.95	0.96
B14	1072	1711	906	1.00	1.00	1.00
B28	1072	1710	906	1.00	1.00	1.00
G14	1318	2123	1120	1.23	1.24	1.23
G28	1473	2452	1307	1.37	1.43	1.44

Table 21. 50%ile sediment respiration rate in Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mmol O_2 m^{-2} d^{-1}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	21.6	12.7	15.0	1.00	1.00	1.00
L1	21.5	11.9	13.6	0.99	0.94	0.91
L2	17.1	11.2	13.6	0.79	0.88	0.91
L3	17.0	10.4	12.3	0.79	0.82	0.82
L4	12.8	9.8	12.8	0.59	0.77	0.85
L5	21.4	12.3	14.5	0.99	0.97	0.97
L6	20.7	12.2	14.3	0.96	0.96	0.95
L7	10.6	6.1	6.3	0.49	0.48	0.42
L8	6.5	3.5	3.8	0.30	0.28	0.26
F1	23.4	13.0	15.3	1.09	1.03	1.02
F2	21.7	13.2	15.1	1.01	1.04	1.01
F3	20.3	12.3	14.1	0.94	0.97	0.94
E1	21.1	10.3	9.0	0.98	0.81	0.60
E2	21.5	12.8	15.2	1.00	1.01	1.01
E3	21.5	12.1	14.2	1.00	0.95	0.94
B14	21.6	12.7	15.0	1.00	1.00	1.00
B28	21.6	12.7	15.0	1.00	1.00	1.00
G14	25.5	13.5	16.5	1.18	1.06	1.10
G28	28.7	14.0	17.9	1.33	1.10	1.19

Table 22. 95%ile sediment respiration rate in Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mmol O_2 m^{-2} d^{-1}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	64.0	23.0	34.3	1.00	1.00	1.00
L1	64.0	22.1	31.1	1.00	0.96	0.91
L2	50.8	21.3	34.4	0.79	0.92	1.00
L3	50.7	20.3	30.1	0.79	0.88	0.88
L4	38.3	19.8	34.4	0.60	0.86	1.00
L5	63.0	22.7	34.4	0.99	0.99	1.00
L6	62.4	22.6	34.4	0.98	0.98	1.00
L7	32.1	14.9	19.3	0.50	0.65	0.56
L8	19.6	9.5	12.0	0.31	0.42	0.35
F1	68.4	22.5	33.4	1.07	0.98	0.97
F2	64.0	23.1	34.0	1.00	1.01	0.99
F3	60.4	22.6	33.1	0.94	0.98	0.96
E1	63.8	24.0	34.5	1.00	1.04	1.00
E2	63.9	23.0	35.1	1.00	1.00	1.02
E3	64.0	22.5	34.2	1.00	0.98	1.00
B14	64.0	23.0	34.3	1.00	1.00	1.00
B28	64.0	23.0	34.3	1.00	1.00	1.00
G14	74.5	23.5	36.3	1.16	1.02	1.06
G28	82.3	23.6	37.1	1.29	1.03	1.08

Table 23. 50%ile denitrification efficiency in sediments of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	%			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	46.5	9.7	25.7	1.00	1.00	1.00
L1	46.5	13.6	26.8	1.00	1.41	1.04
L2	45.8	17.4	25.3	0.99	1.80	0.98
L3	45.8	18.9	26.8	0.99	1.96	1.04
L4	44.9	18.8	25.3	0.97	1.94	0.98
L5	46.5	11.5	26.5	1.00	1.19	1.03
L6	46.5	12.5	26.3	1.00	1.29	1.02
L7	43.4	31.6	35.2	0.93	3.27	1.37
L8	38.3	34.1	34.3	0.82	3.52	1.34
F1	46.3	6.3	21.0	1.00	0.65	0.82
F2	46.4	10.1	24.4	1.00	1.04	0.95
F3	46.4	15.0	27.7	1.00	1.55	1.08
E1	46.0	28.7	32.4	0.99	2.96	1.26
E2	46.5	13.8	28.4	1.00	1.42	1.11
E3	46.5	10.2	24.0	1.00	1.05	0.93
B14	46.5	9.7	25.6	1.00	1.00	1.00
B28	46.5	9.7	25.7	1.00	1.00	1.00
G14	46.3	6.0	21.1	1.00	0.62	0.82
G28	46.2	2.6	16.9	1.00	0.27	0.66

Table 24. 5%ile denitrification efficiency in sediments of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	%			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	32.7	0.2	1.7	1.00	1.00	1.00
L1	32.7	0.2	2.9	1.00	1.09	1.72
L2	33.2	0.2	1.9	1.01	1.19	1.09
L3	33.2	0.2	3.0	1.01	1.32	1.75
L4	33.2	0.3	2.7	1.02	1.57	1.59
L5	33.0	0.2	2.2	1.01	1.26	1.31
L6	33.0	0.2	2.6	1.01	1.44	1.52
L7	32.4	11.0	11.0	0.99	63.81	6.43
L8	24.9	6.7	7.1	0.76	38.89	4.16
F1	32.5	0.2	1.5	0.99	0.90	0.87
F2	32.7	0.2	2.2	1.00	1.24	1.28
F3	33.2	0.3	2.4	1.01	1.73	1.39
E1	32.3	3.8	8.8	0.99	22.00	5.17
E2	32.7	0.3	3.5	1.00	1.50	2.02
E3	32.7	0.2	1.7	1.00	1.01	1.00
B14	32.8	0.2	1.7	1.00	1.00	0.99
B28	32.8	0.2	1.7	1.00	1.00	1.00
G14	32.1	0.2	1.9	0.98	1.05	1.14
G28	31.0	0.1	1.0	0.95	0.77	0.58

Table 25. 50%ile microphytobenthos biomass in sediments of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg Chl a m^{-2}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	0.0	0.1	0.3	1.00	1.00	1.00
L1	0.0	0.1	0.3	1.01	1.04	1.20
L2	0.1	0.2	0.3	1.92	1.17	1.26
L3	0.1	0.2	0.4	1.93	1.23	1.51
L4	0.2	0.2	0.4	5.59	1.44	1.58
L5	0.0	0.1	0.3	1.00	1.01	1.03
L6	0.0	0.1	0.3	1.04	1.02	1.05
L7	0.4	0.4	0.8	9.50	2.99	3.25
L8	0.9	0.8	1.2	24.73	5.92	4.82
F1	0.0	0.1	0.3	0.99	1.08	1.12
F2	0.0	0.1	0.2	1.02	1.02	0.99
F3	0.0	0.1	0.2	1.10	0.97	0.93
E1	0.1	0.6	1.9	2.04	4.81	7.39
E2	0.0	0.1	0.3	0.99	1.01	1.00
E3	0.0	0.1	0.3	1.01	1.03	1.08
B14	0.0	0.1	0.3	0.98	1.00	0.99
B28	0.0	0.1	0.3	0.99	1.00	1.00
G14	0.0	0.1	0.2	0.74	0.92	0.79
G28	0.0	0.1	0.2	0.49	0.82	0.69

Table 26. 95%ile microphytobenthos biomass in sediments of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg Chl a m^{-2}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	0.2	4.7	4.4	1.00	1.00	1.00
L1	0.3	4.7	4.5	1.04	1.00	1.02
L2	0.8	4.9	4.6	3.16	1.05	1.04
L3	0.8	4.9	4.6	3.21	1.05	1.05
L4	1.4	5.1	4.7	5.42	1.09	1.07
L5	0.3	4.7	4.4	1.02	1.00	1.00
L6	0.3	4.7	4.4	1.16	1.01	1.01
L7	1.8	5.1	4.7	7.11	1.10	1.07
L8	2.6	5.2	4.7	10.53	1.11	1.07
F1	0.2	4.8	4.7	0.93	1.04	1.07
F2	0.3	4.7	4.5	1.12	1.01	1.03
F3	0.3	4.5	4.3	1.23	0.98	0.96
E1	1.1	5.5	5.5	4.28	1.17	1.24
E2	0.2	4.7	4.5	0.98	1.00	1.01
E3	0.3	4.7	4.6	1.09	1.00	1.03
B14	0.2	4.7	4.4	0.99	1.00	1.00
B28	0.2	4.7	4.4	0.99	1.00	1.00
G14	0.1	4.5	4.3	0.37	0.96	0.98
G28	0.0	4.2	4.0	0.12	0.90	0.91

Table 27. 50%ile Chl a (including settled phytoplankton) in sediments of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg Chl a m^{-2}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	10.3	9.0	7.1	1.00	1.00	1.00
L1	10.2	8.0	6.4	0.98	0.88	0.91
L2	7.9	7.2	5.8	0.76	0.79	0.82
L3	7.9	6.7	5.0	0.76	0.74	0.71
L4	6.0	6.0	4.6	0.58	0.66	0.65
L5	10.2	8.7	6.9	0.99	0.97	0.98
L6	9.9	8.6	6.7	0.95	0.95	0.96
L7	5.0	3.9	3.1	0.48	0.43	0.44
L8	3.9	3.4	2.6	0.37	0.38	0.37
F1	11.8	9.7	7.6	1.14	1.07	1.07
F2	10.5	9.5	7.0	1.02	1.05	0.99
F3	9.2	7.8	6.3	0.89	0.86	0.89
E1	9.7	6.5	6.1	0.94	0.72	0.87
E2	10.3	8.9	6.8	0.99	0.98	0.96
E3	10.3	8.5	6.9	0.99	0.95	0.98
B14	10.4	9.0	7.1	1.00	1.00	1.00
B28	10.3	9.0	7.1	1.00	1.00	1.00
G14	12.2	11.1	7.5	1.18	1.23	1.06
G28	14.1	12.6	8.8	1.37	1.39	1.24

Table 28. 95%ile Chl a (including settled phytoplankton) in sediments of Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg Chl a m^{-2}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	50.4	22.2	15.7	1.00	1.00	1.00
L1	50.3	21.2	13.2	1.00	0.95	0.85
L2	37.8	20.1	16.0	0.75	0.90	1.02
L3	37.8	19.2	12.9	0.75	0.87	0.82
L4	27.3	17.5	15.1	0.54	0.79	0.96
L5	49.4	21.7	15.9	0.98	0.98	1.02
L6	48.8	21.7	16.1	0.97	0.98	1.03
L7	23.6	14.4	6.6	0.47	0.65	0.42
L8	16.1	12.3	5.5	0.32	0.55	0.35
F1	55.4	24.4	19.0	1.10	1.10	1.21
F2	50.8	23.1	18.2	1.01	1.04	1.16
F3	46.8	22.2	12.3	0.93	1.00	0.78
E1	50.4	24.8	13.9	1.00	1.12	0.89
E2	50.2	21.4	16.2	1.00	0.96	1.03
E3	50.5	22.0	15.7	1.00	0.99	1.00
B14	50.3	22.2	15.6	1.00	1.00	1.00
B28	50.3	22.2	15.6	1.00	1.00	1.00
G14	61.4	26.0	19.4	1.22	1.17	1.23
G28	70.2	30.5	22.0	1.39	1.37	1.40

Table 29. 95%ile macroalgal biomass in Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg N m^{-2}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	0.0	33.7	0.0		1.00	
L1	0.0	37.1	0.0		1.10	
L2	0.0	59.5	0.0		1.76	
L3	0.0	57.7	0.0		1.71	
L4	0.0	81.1	0.0		2.40	
L5	0.0	33.6	0.0		1.00	
L6	0.0	34.7	0.0		1.03	
L7	0.0	49.3	0.8		1.46	
L8	0.0	33.9	3.8		1.01	
F1	0.0	43.3	0.0		1.28	
F2	0.0	32.2	0.0		0.96	
F3	0.0	27.3	0.0		0.81	
E1	0.0	44.1	5.5		1.31	
E2	0.0	24.9	0.0		0.74	
E3	0.0	30.5	0.0		0.91	
B14	0.0	25.2	0.0		0.75	
B28	0.0	24.5	0.0		0.73	
G14	0.0	4.0	0.0		0.12	
G28	0.0	0.0	0.0		0.00	

Table 30. 95%ile benthic primary production in Lakes Wellington, Victoria and King, absolute, and relative to baseline.

	mg Cm^{-2} d^{-1}			Relative to Baseline		
Scenario	Wellington	Victoria	King	Wellington	Victoria	King
B	0.1	40.6	37.6	1.00	1.00	1.00
L1	0.1	42.2	39.5	1.08	1.04	1.05
L2	1.1	52.1	41.4	9.03	1.29	1.10
L3	1.2	52.8	43.2	9.25	1.30	1.15
L4	4.2	60.3	44.9	33.50	1.49	1.19
L5	0.1	41.3	37.8	1.03	1.02	1.01
L6	0.2	41.9	38.4	1.27	1.03	1.02
L7	7.4	59.8	48.4	58.77	1.47	1.29
L8	17.4	62.1	51.5	137.41	1.53	1.37
F1	0.1	43.7	44.5	0.85	1.08	1.18
F2	0.2	40.6	39.6	1.19	1.00	1.05
F3	0.2	39.4	34.7	1.46	0.97	0.92
E1	2.2	65.6	71.7	17.70	1.62	1.91
E2	0.1	39.5	39.0	0.96	0.97	1.04
E3	0.1	40.3	41.1	1.16	0.99	1.09
B14	0.1	39.6	37.6	0.98	0.98	1.00
B28	0.1	39.6	37.6	0.99	0.98	1.00
G14	0.0	31.7	34.0	0.20	0.78	0.90
G28	0.0	26.8	28.5	0.03	0.66	0.76

8 Figures

Figures 1-51 show indicator statistics for model scenarios. In all these figures, the vertical line joins minimum and maximum, the thin box joins 5%ile and 95%ile, the thick box 20%ile and 80%ile, and the horizontal bar marks the median.

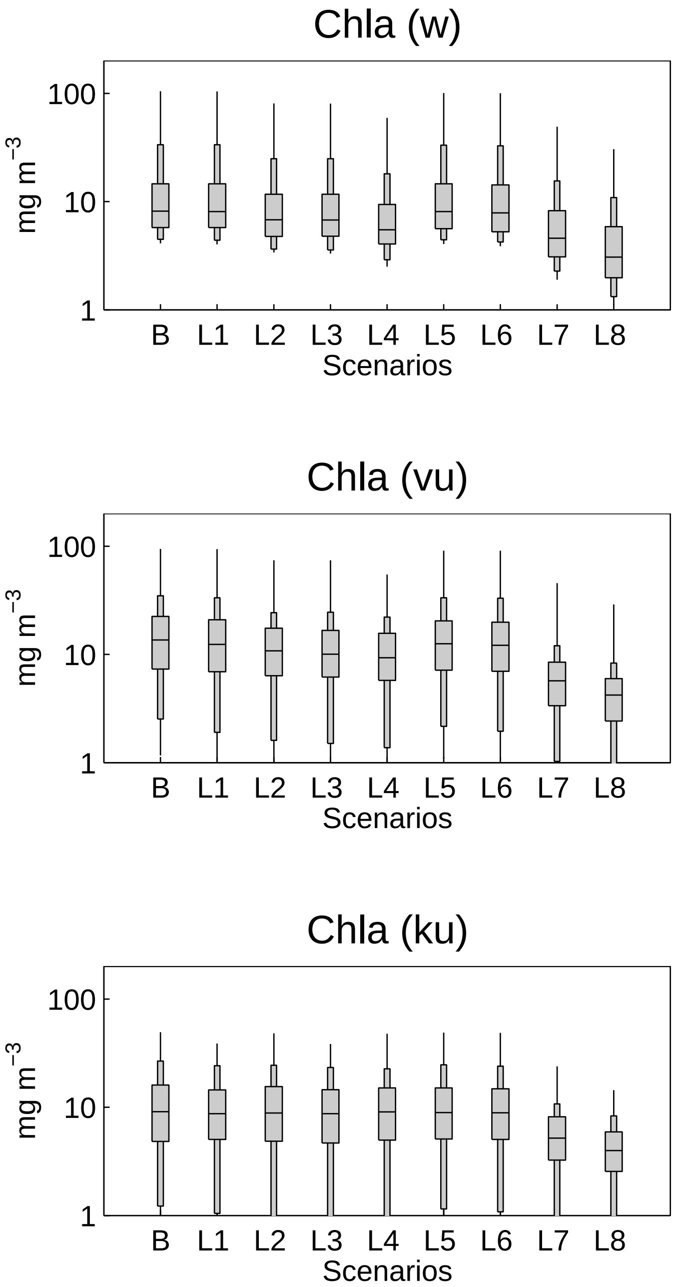


Fig. 1. Phytoplankton biomass (Chl a) vs load scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

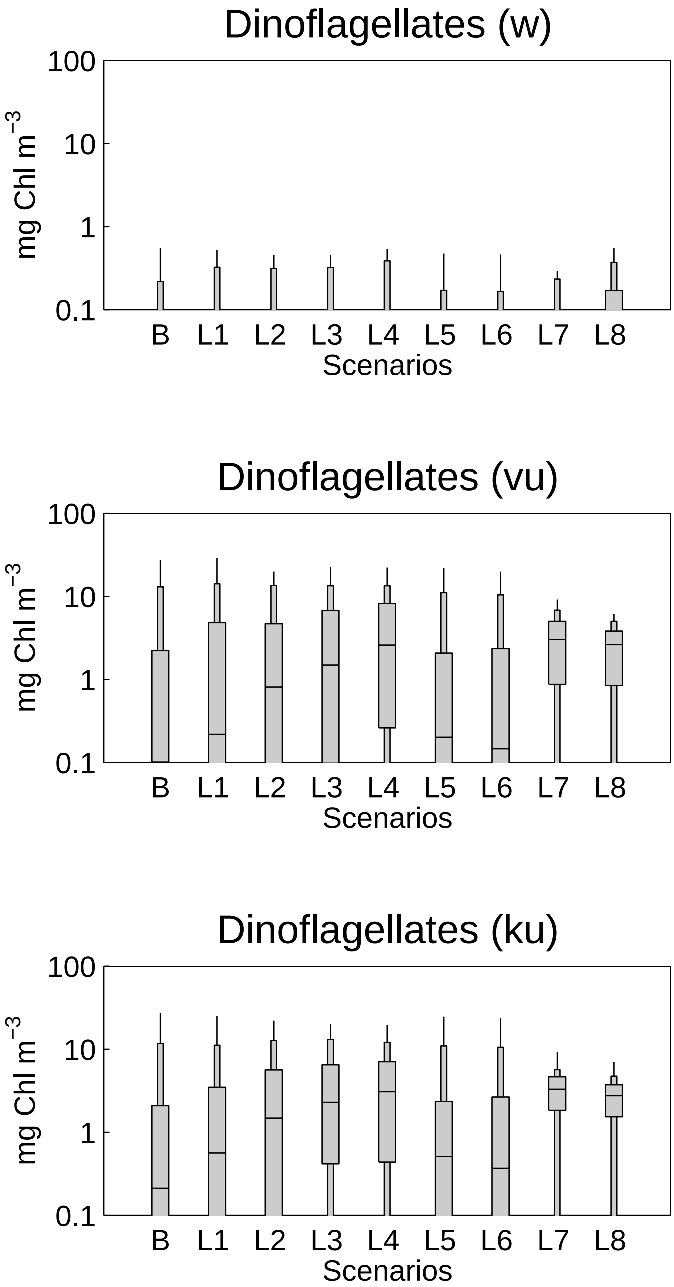


Fig. 2. Dinoflagellate biomass (Chl a) vs load scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

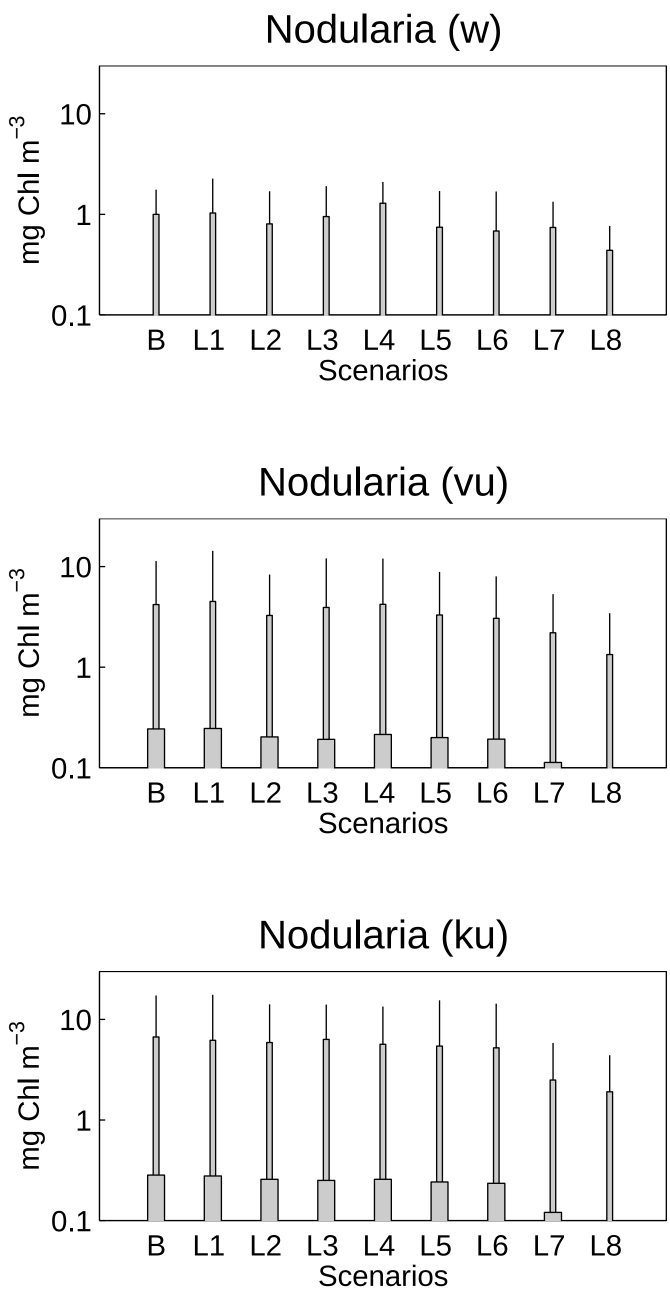


Fig. 3. Nodularia biomass (Chl a) vs load scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

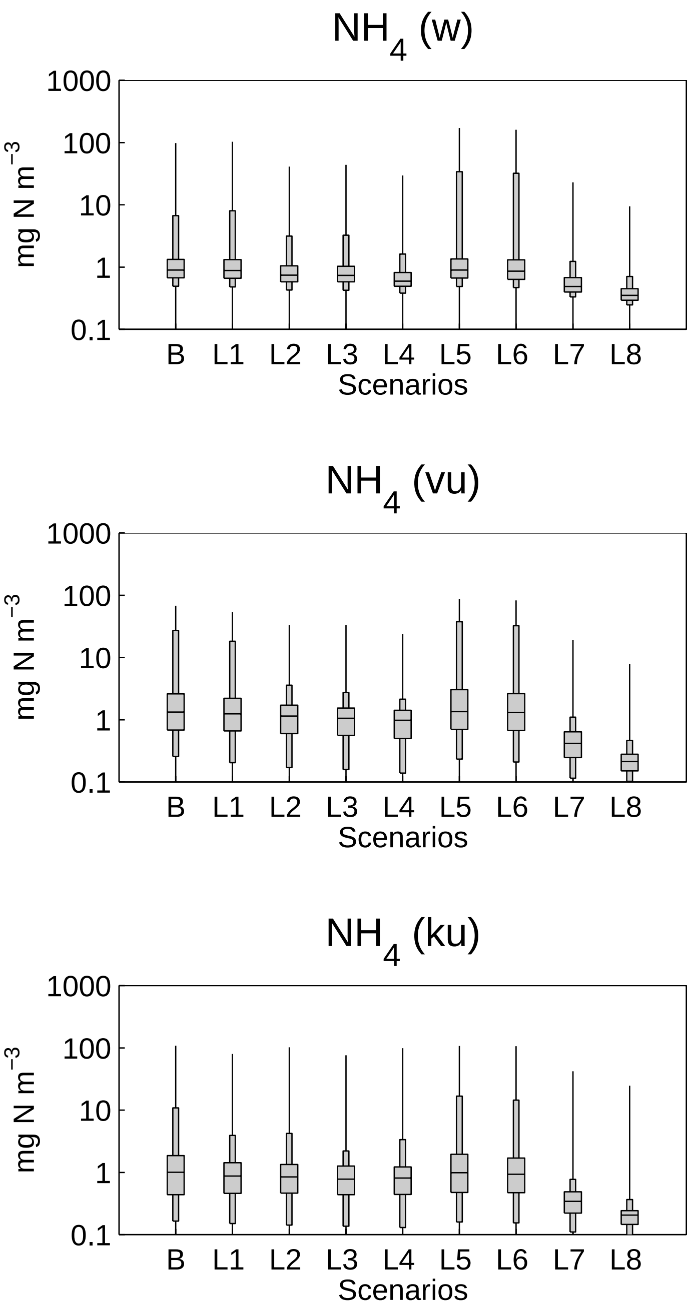


Fig. 4. Ammonia concentration vs load scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

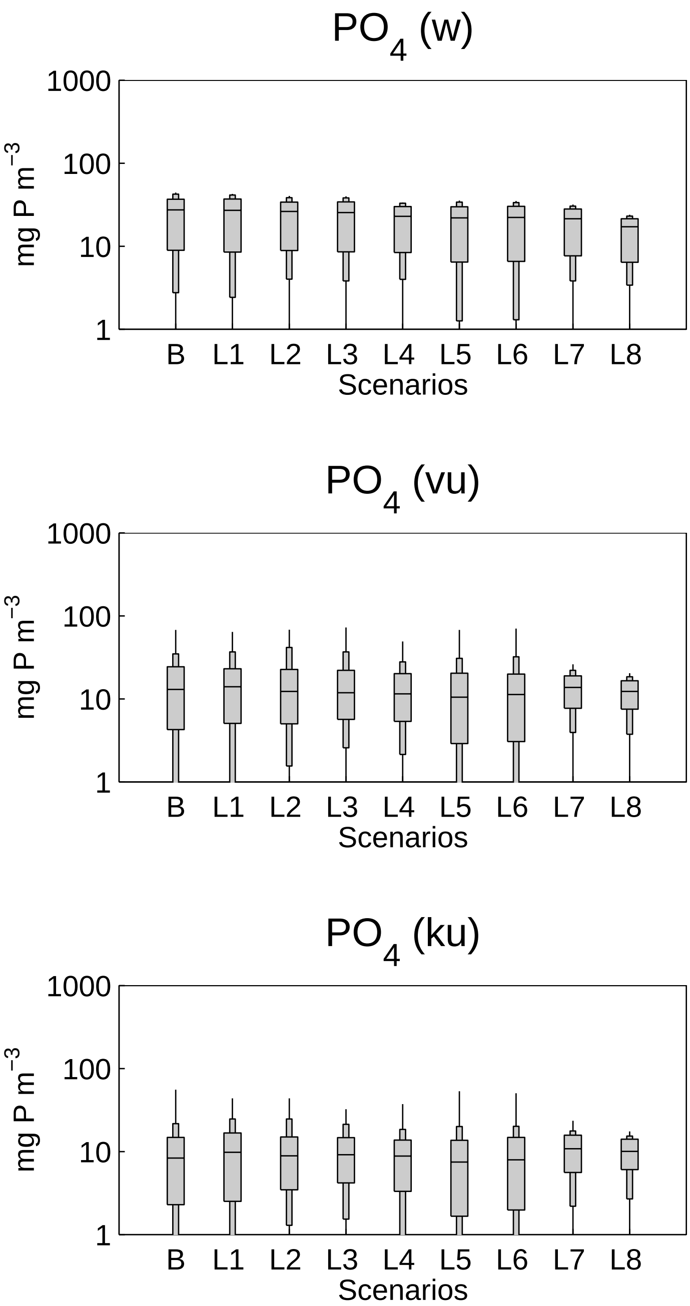


Fig. 5. Phosphate (DIP) concentration vs load scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

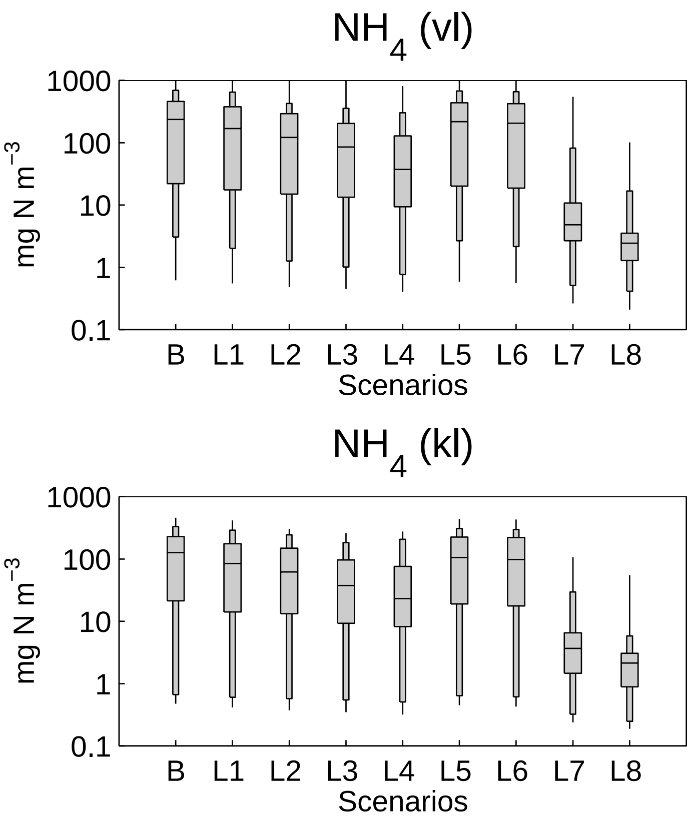


Fig. 6. Ammonia concentration vs load scenarios for L Victoria bottom water (vl) and L. King bottom water (kl).

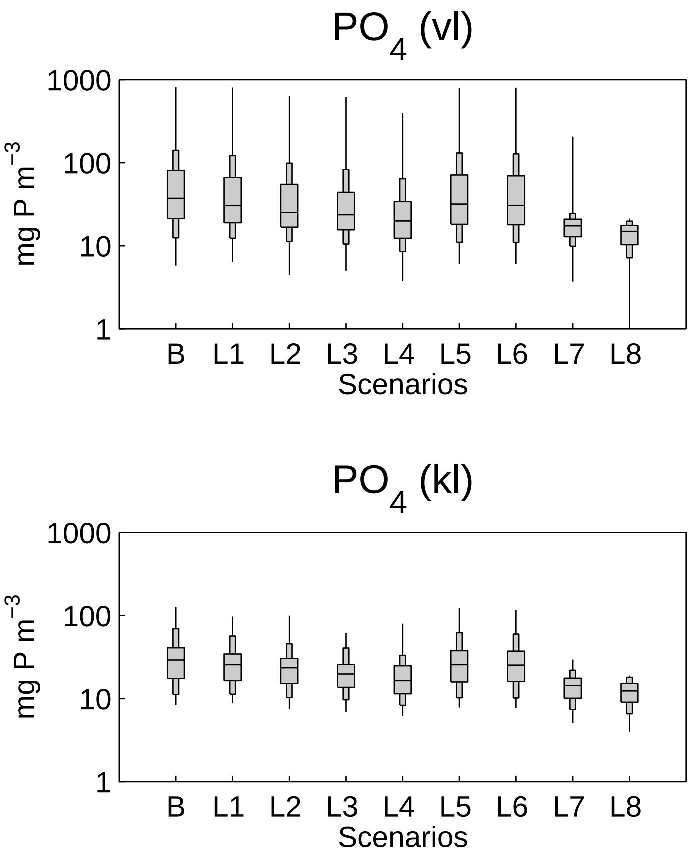


Fig. 7. Phosphate (DIP) concentration vs load scenarios for L Victoria bottom water (vl) and L. King bottom water (kl).

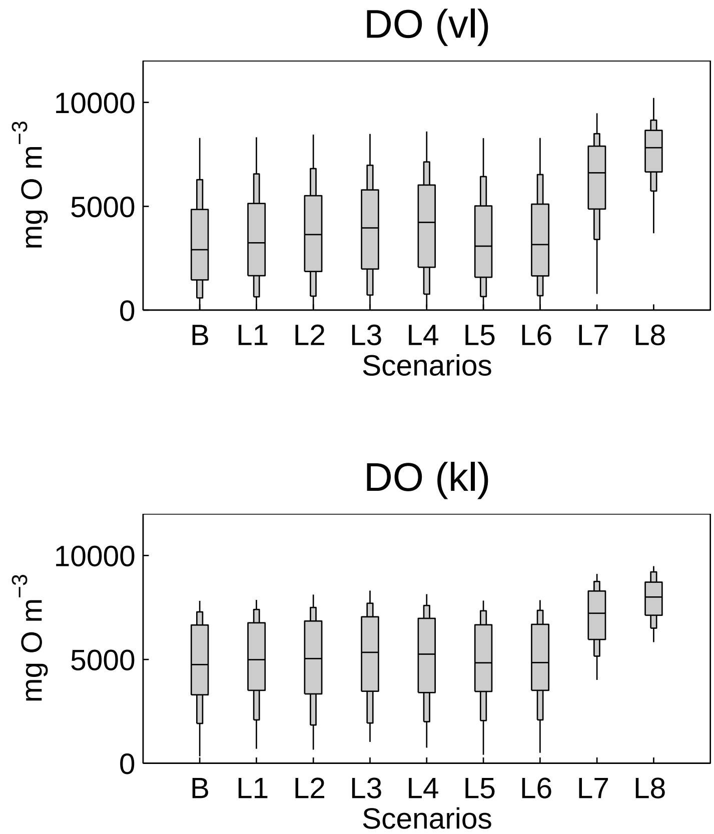


Fig. 8. Dissolved oxygen concentration vs load scenarios for L Victoria bottom water (vl) and L. King bottom water (kl).

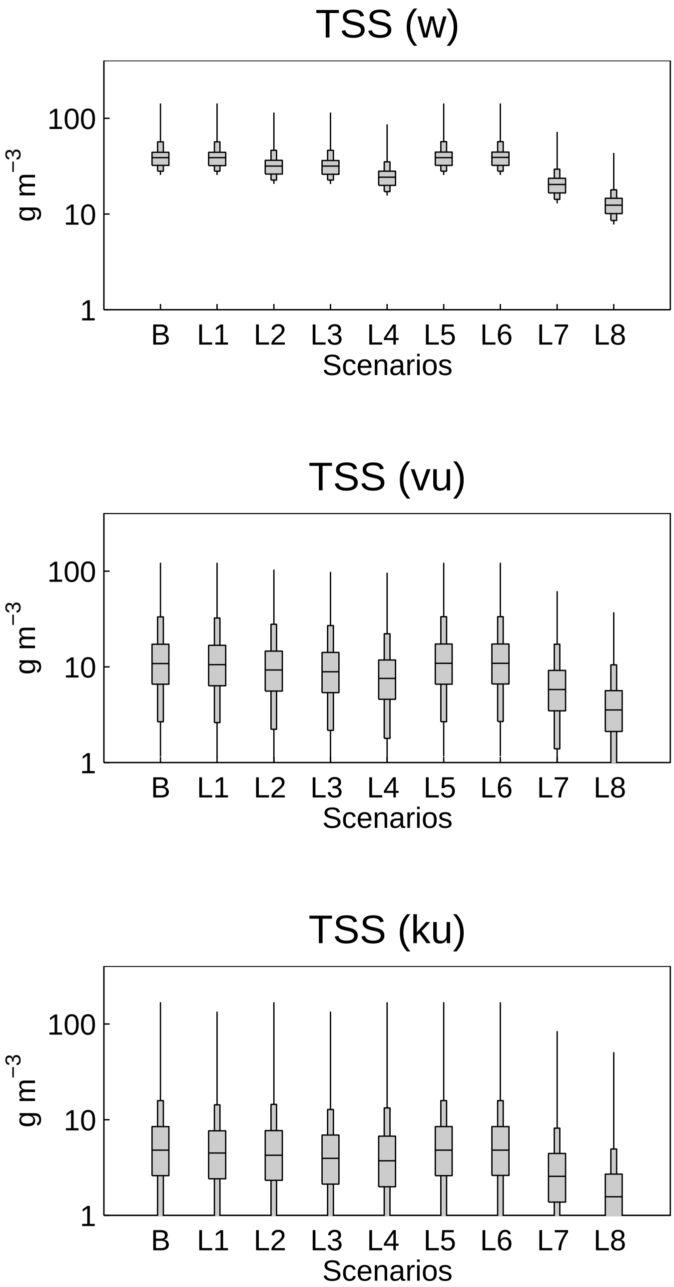


Fig. 9. Suspended sediment (TSS) concentration vs load scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

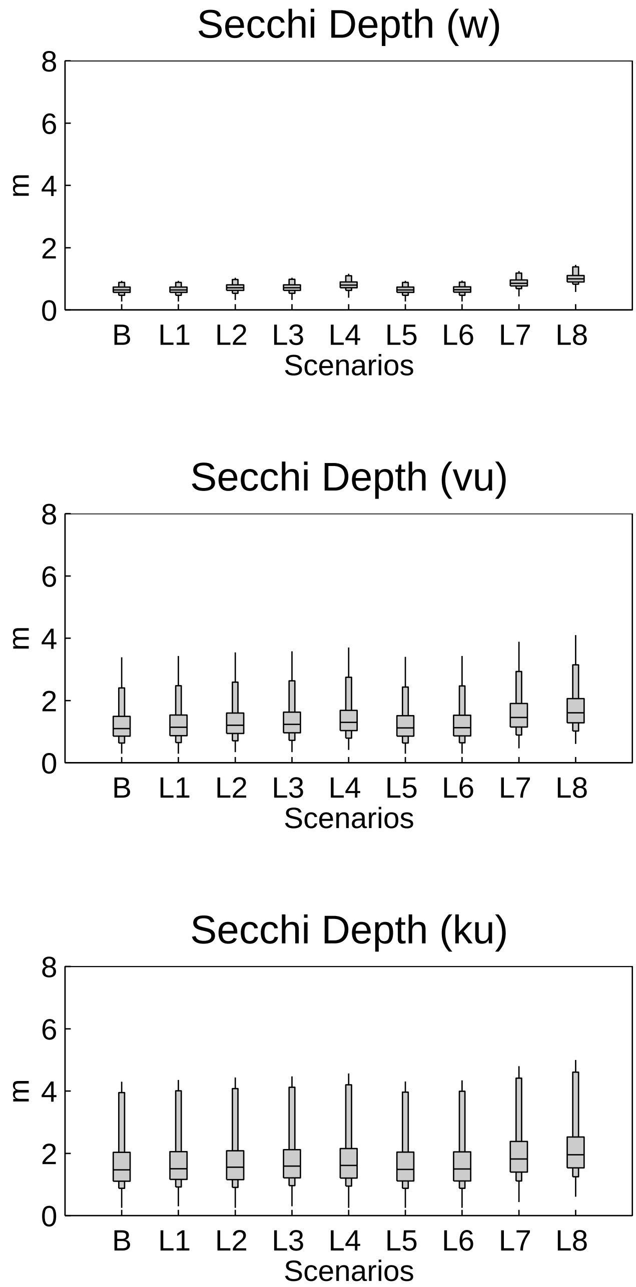


Fig. 10. Secchi depth vs load scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

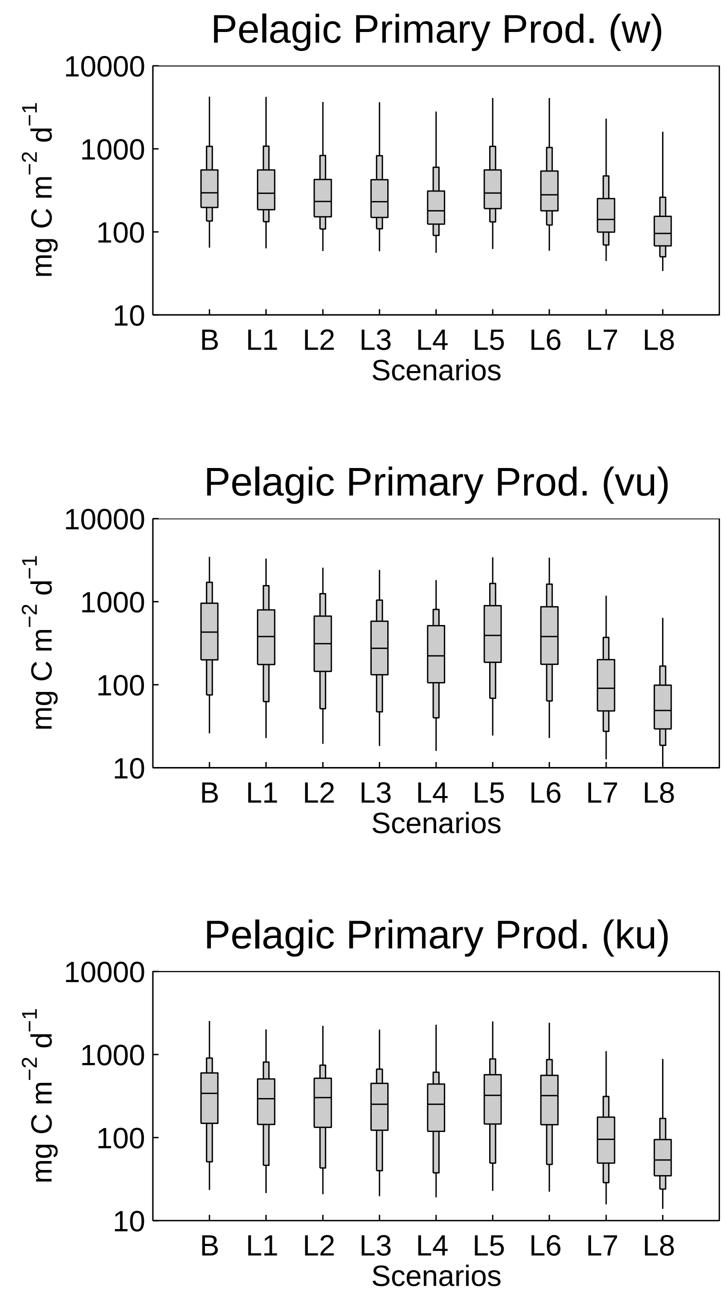


Fig. 11. Pelagic (phytoplankton) primary production vs load scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

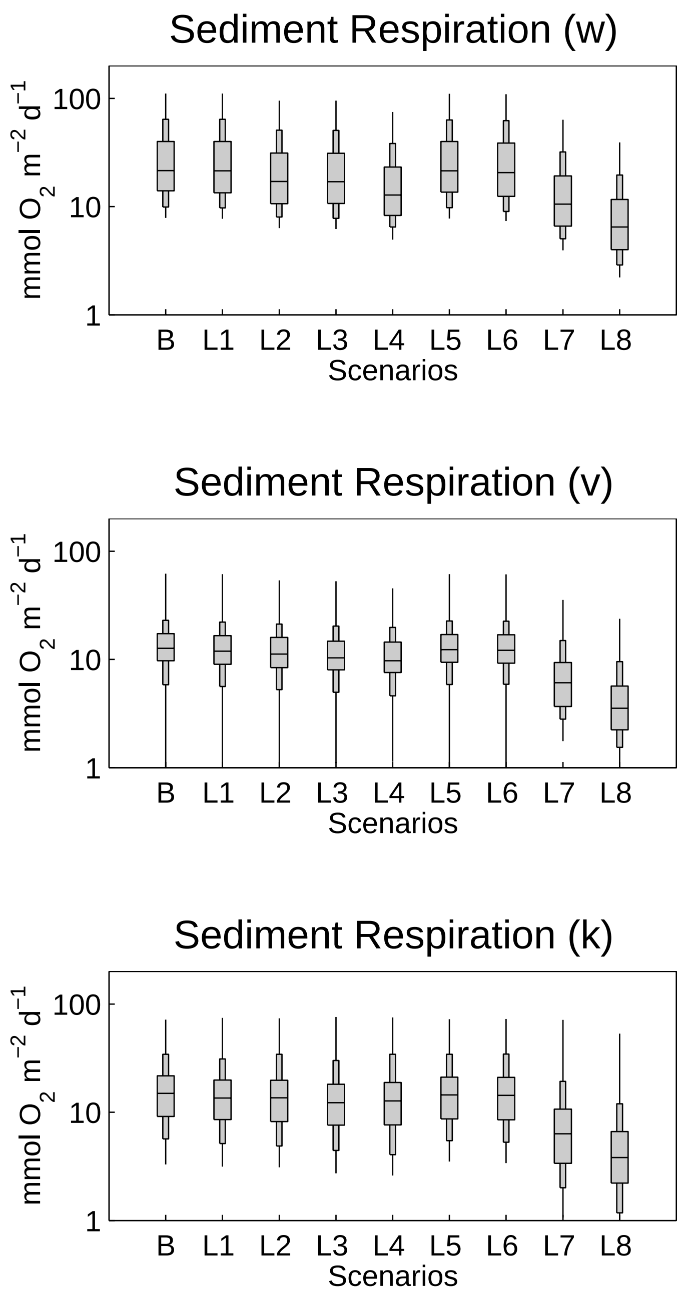


Fig. 12. Sediment respiration rate vs load scenarios for L. Wellington (w), L Victoria (v), L. King (k).

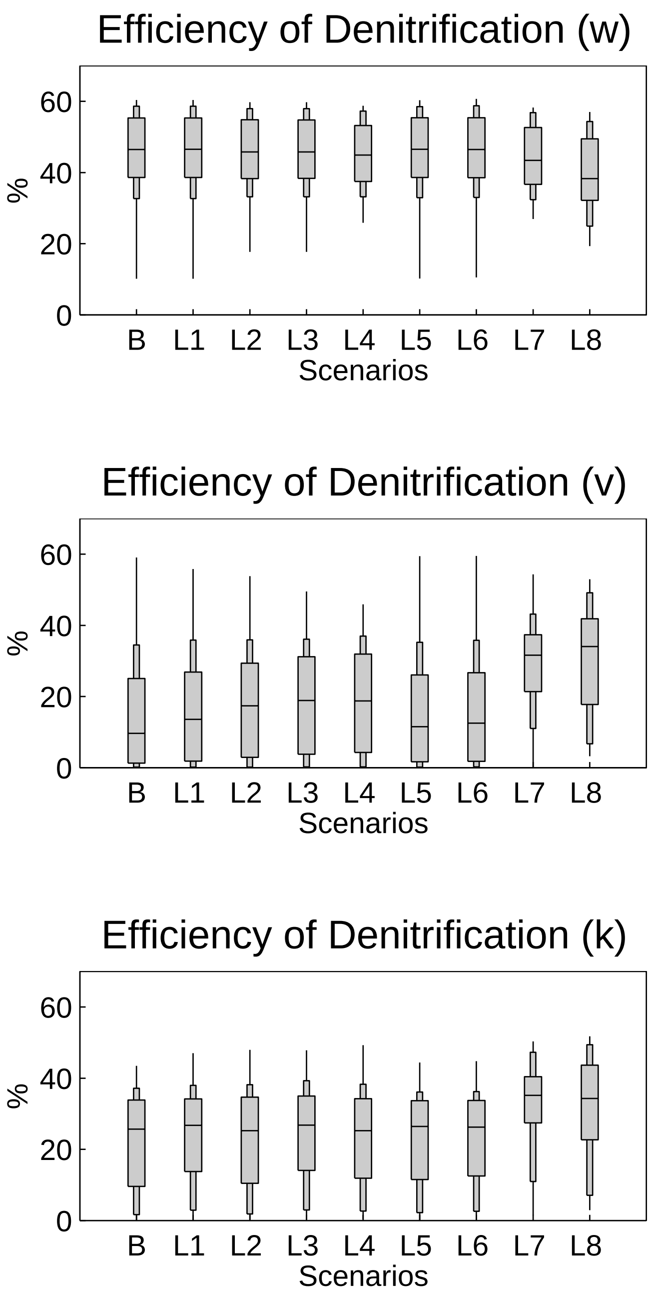


Fig. 13. Sediment denitrification efficiency vs load scenarios for L. Wellington (w), L Victoria (v), L. King (k).

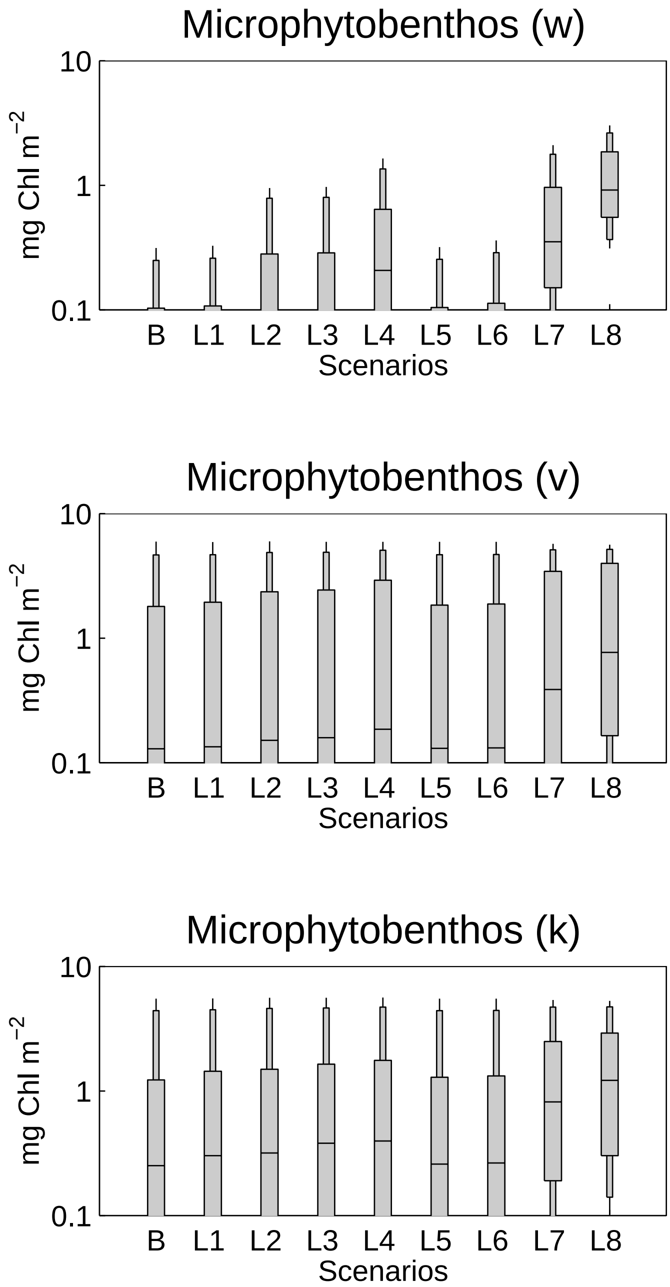


Fig. 14. Sediment microphytobenthos biomass (Chl a per unit area) vs load scenarios for L. Wellington (w), L Victoria (v), L. King (k).

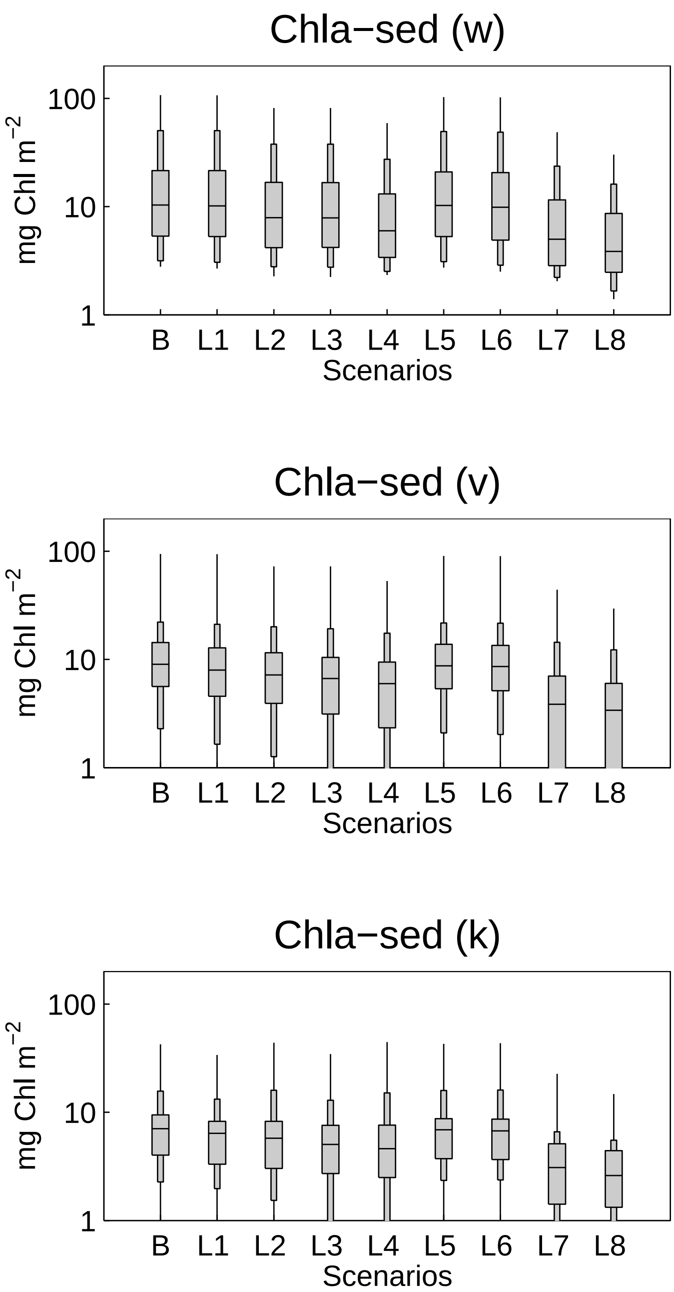


Fig. 15. Total sediment Chl a (per unit area) vs load scenarios for L. Wellington (w), L Victoria (v), L. King (k). (Includes settled phytoplankton).

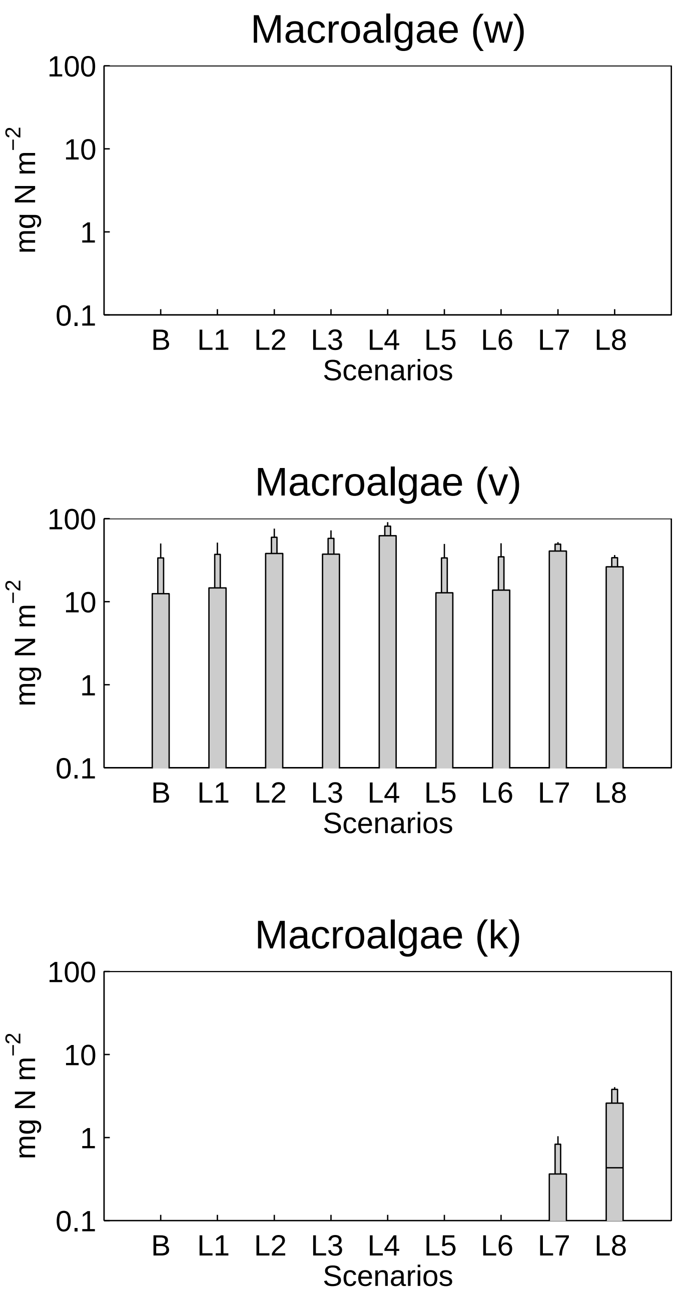


Fig. 16. Macroalgal biomass vs load scenarios for L. Wellington (w), L Victoria (v), L. King (k).

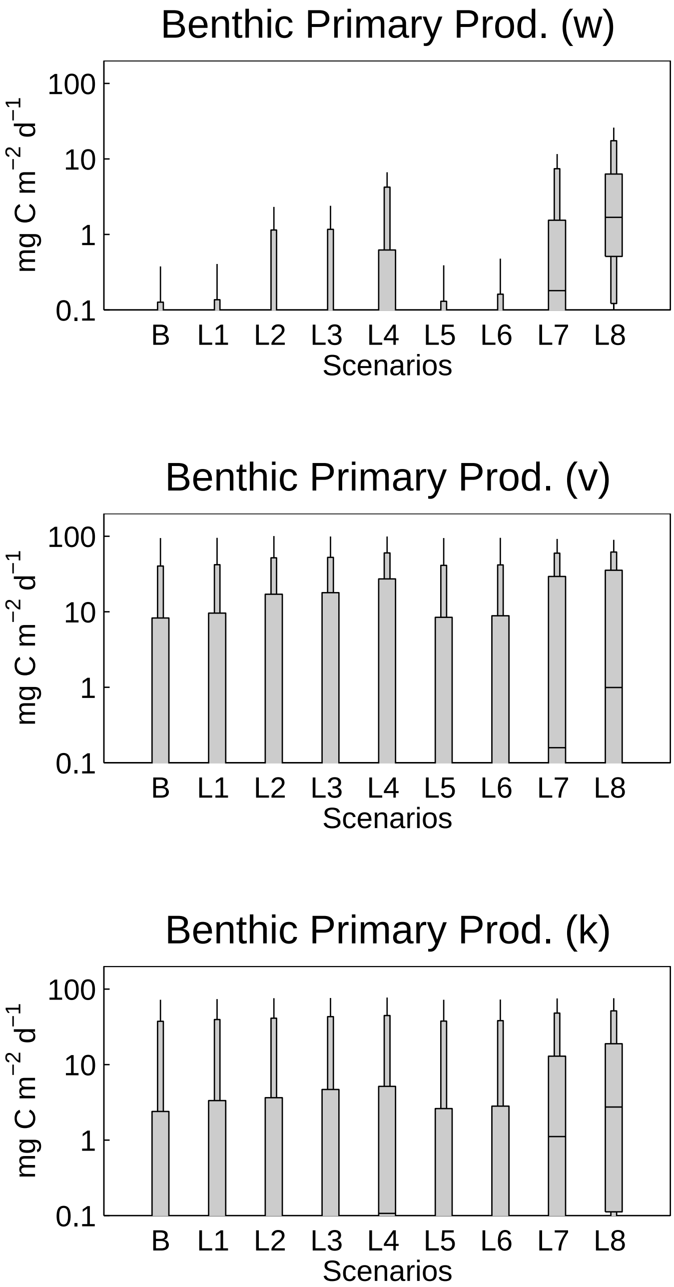


Fig. 17. Benthic primary production (macroalgae + microphytobenthos) vs load scenarios for L. Wellington (w), L Victoria (v), L. King (k).

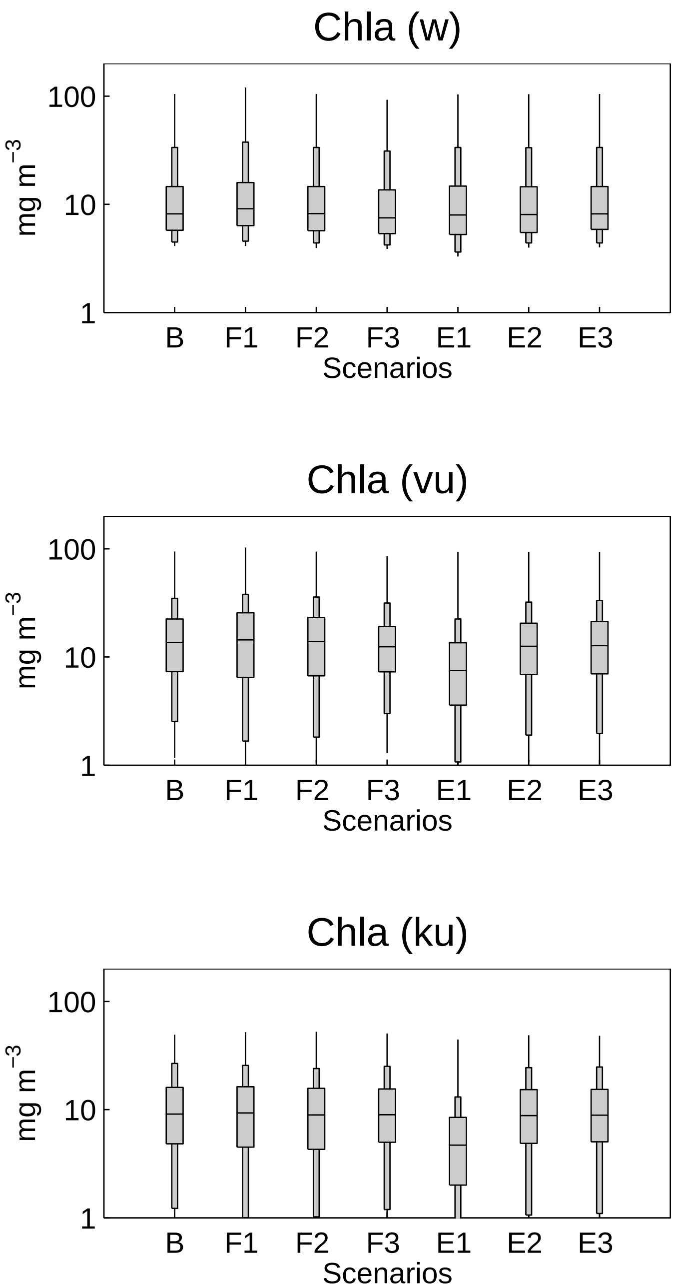


Fig. 18. Phytoplankton biomass (Chl a) vs flow, exchange scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

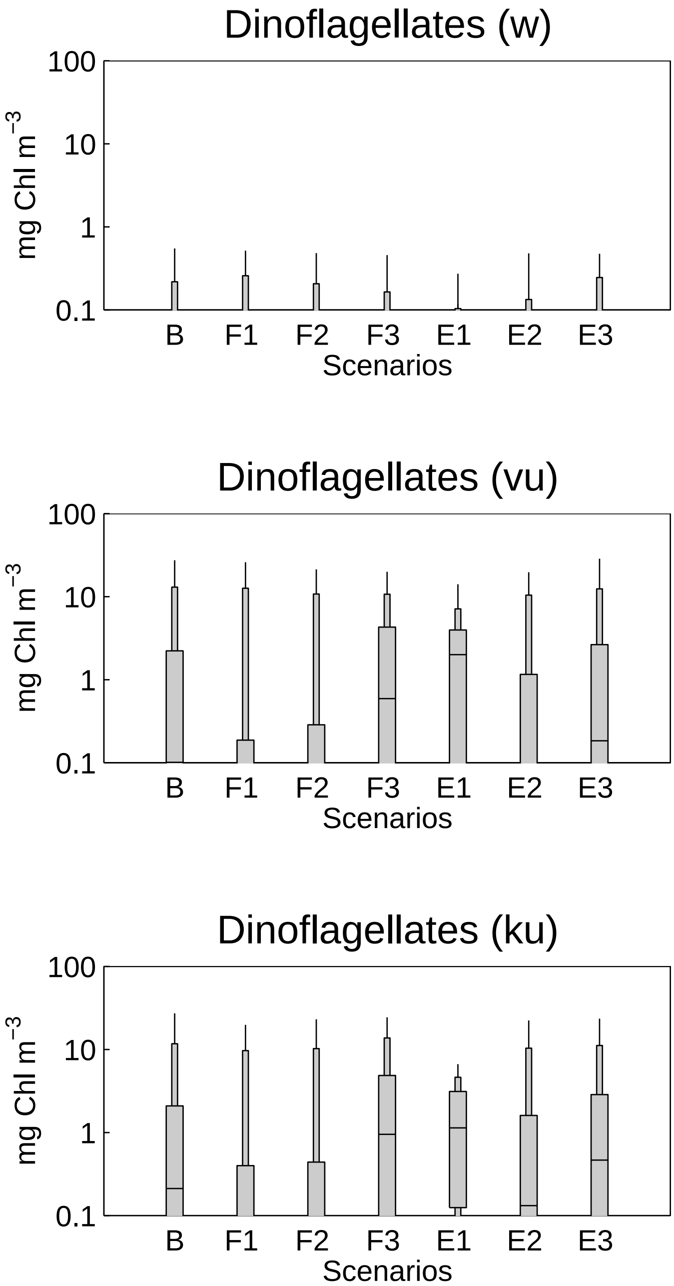


Fig. 19. Dinoflagellate biomass (Chl a) vs flow, exchange scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

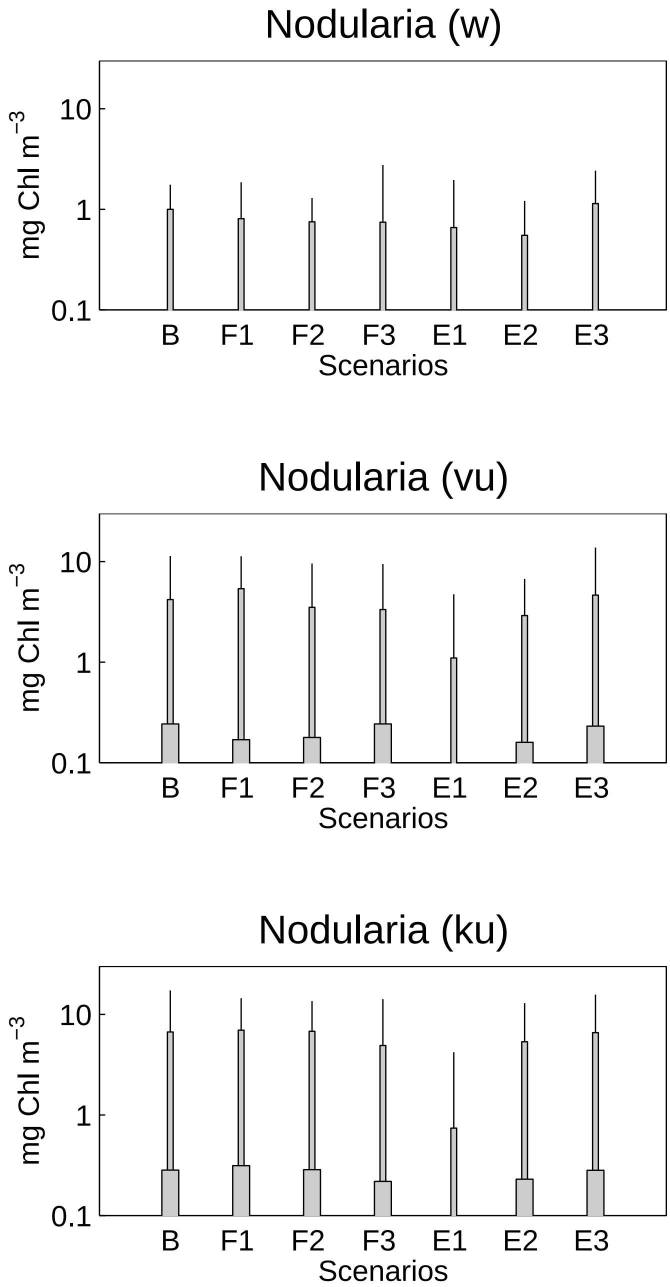
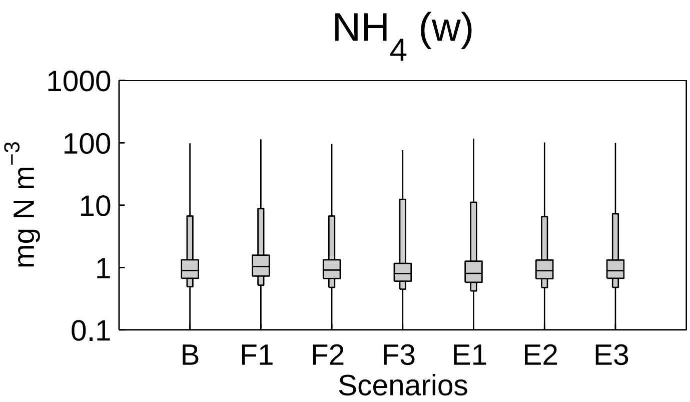
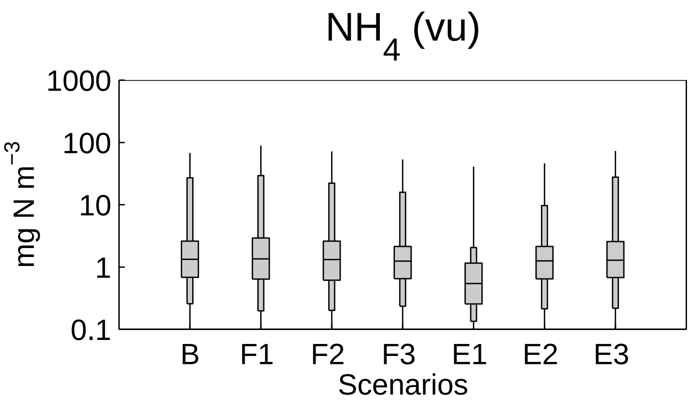


Fig. 20. *Nodularia* biomass (Chl a) vs flow, exchange scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).





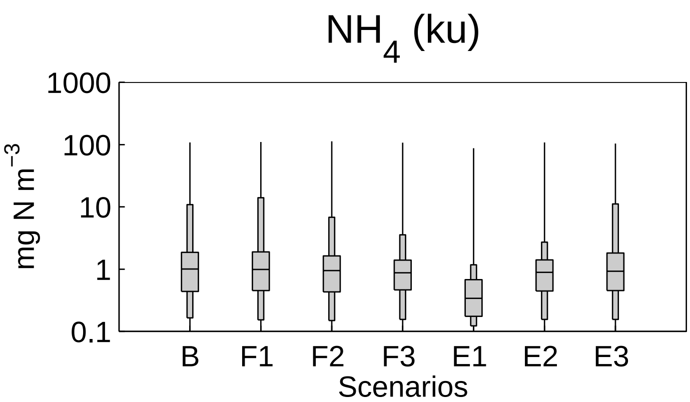


Fig. 21. Ammonia concentration vs flow, exchange scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

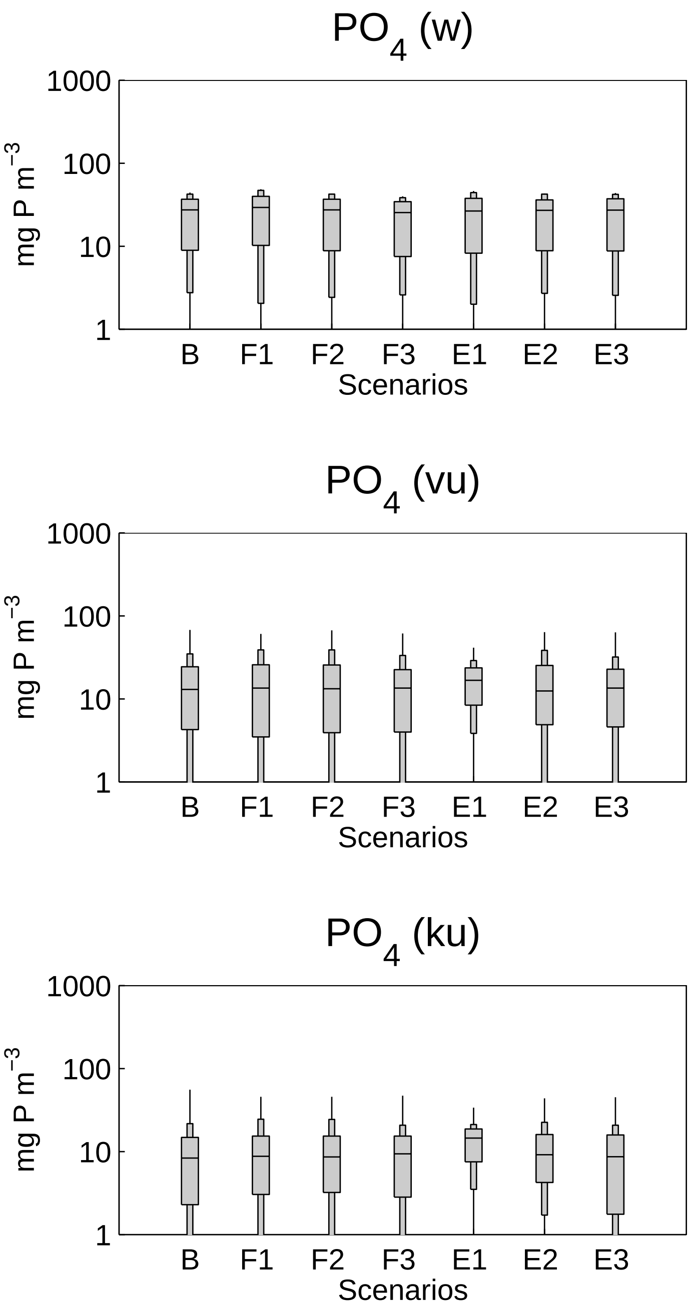


Fig. 22. Phosphate (DIP) concentration vs flow, exchange scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

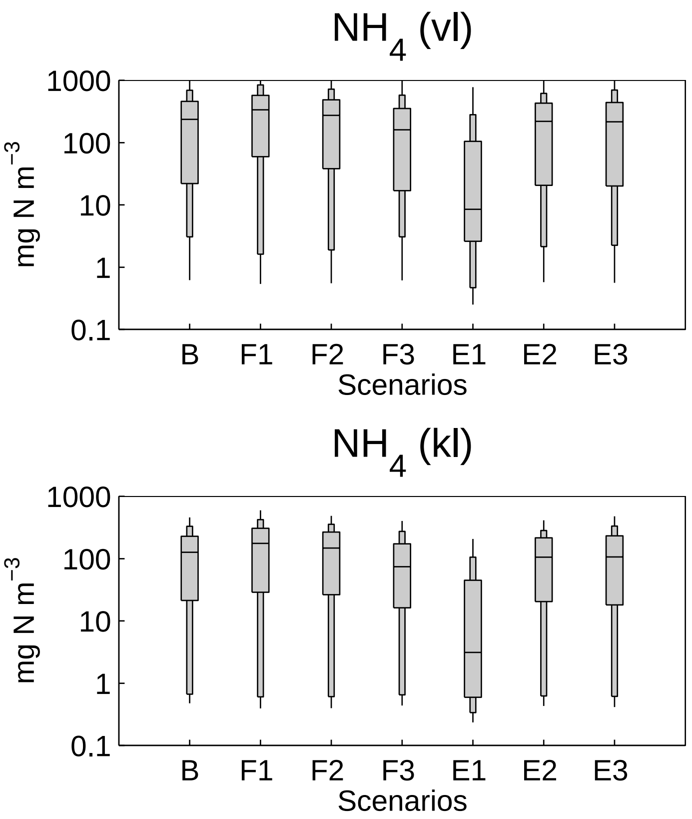


Fig. 23. Ammonia concentration vs flow, exchange scenarios for L Victoria bottom waters (vl) and L. King bottom waters (kl).

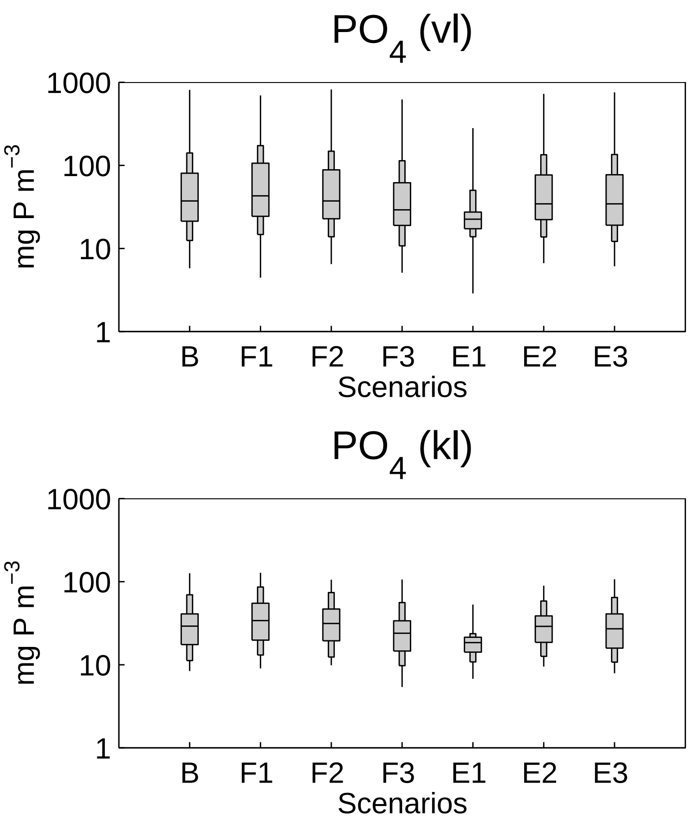


Fig. 24. Phosphate (DIP) concentration vs flow, exchange scenarios for L Victoria bottom waters (vl) and L. King bottom waters (kl).

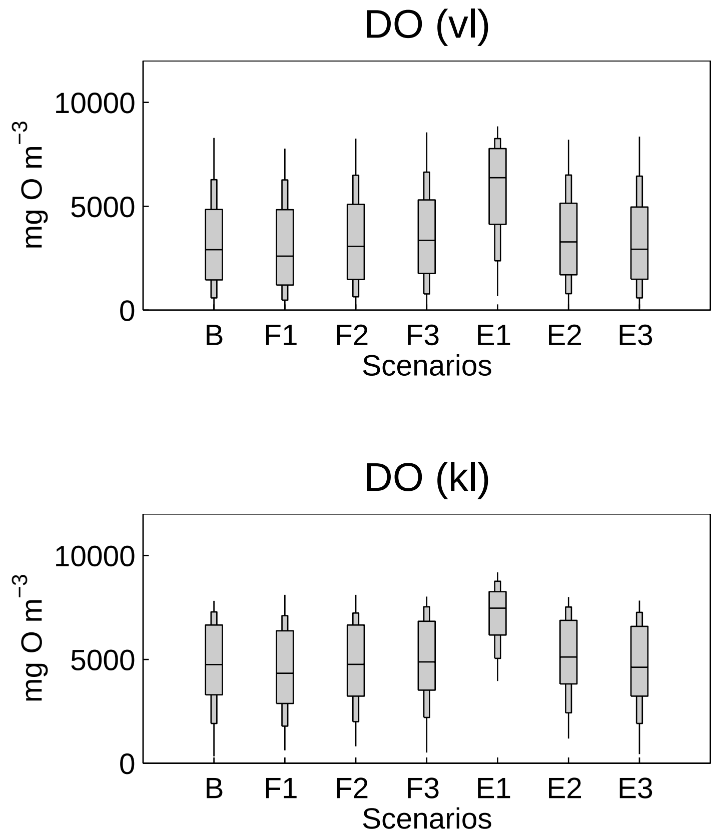


Fig. 25. Dissolved oxygen concentration vs flow, exchange scenarios for L Victoria bottom waters (vl) and L. King bottom waters (kl).

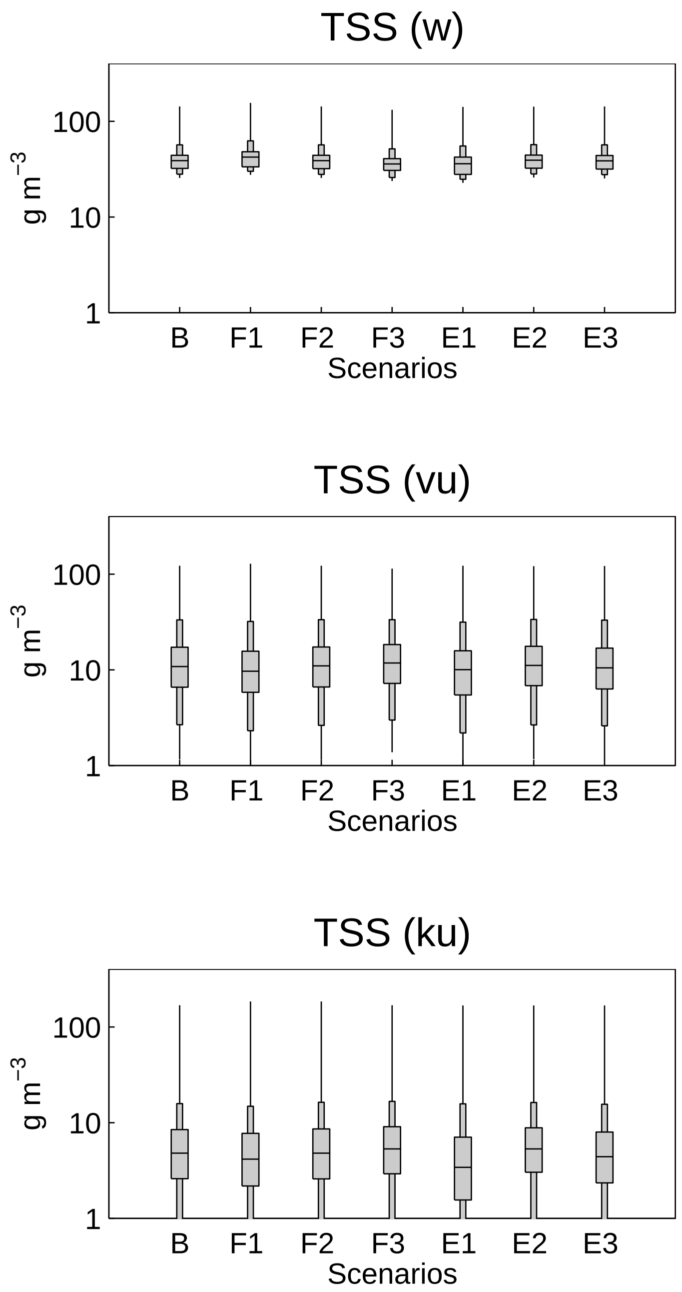


Fig. 26. Suspended sediment concentration vs flow, exchange scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

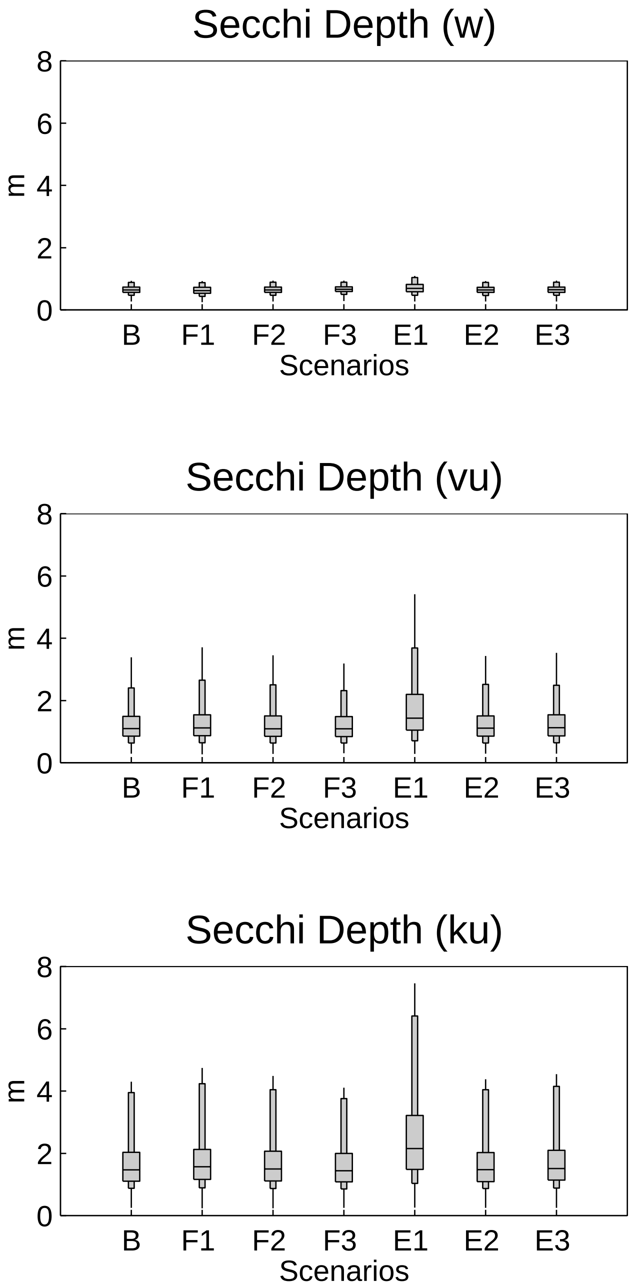


Fig. 27. Secchi depth vs flow, exchange scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

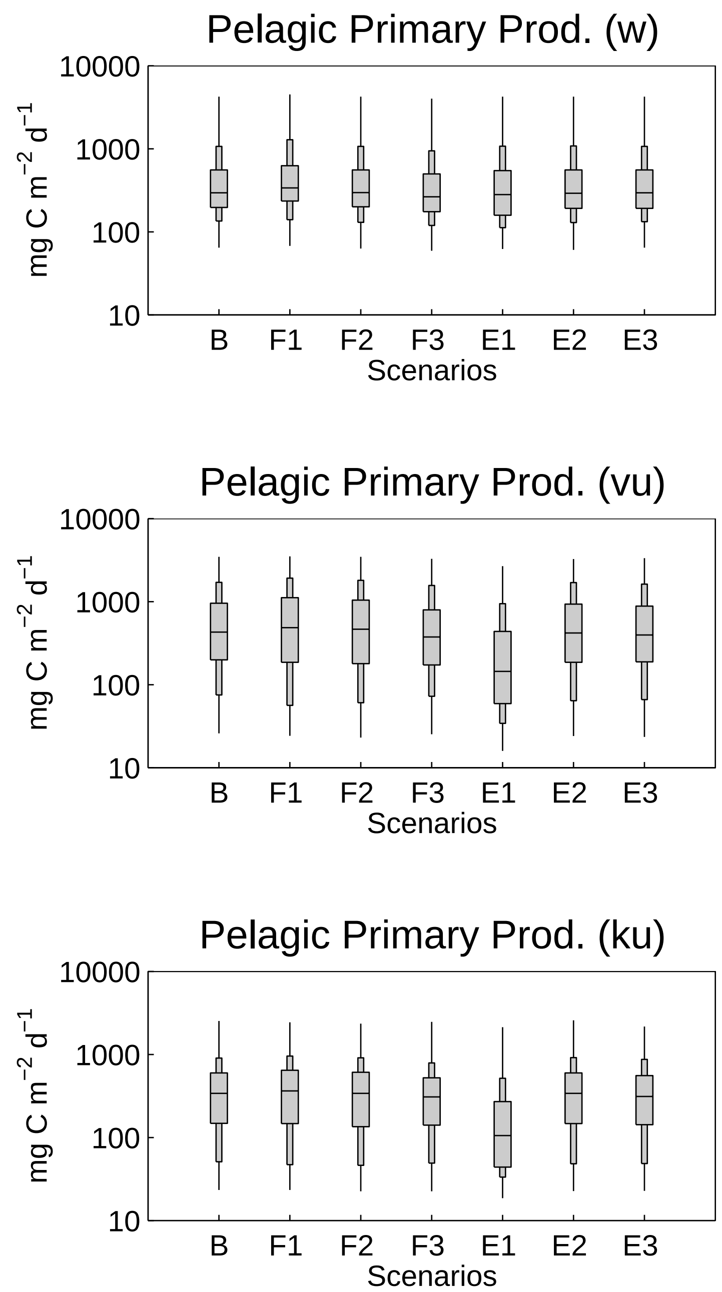


Fig. 28. Pelagic (phytoplankton) primary production vs flow, exchange scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

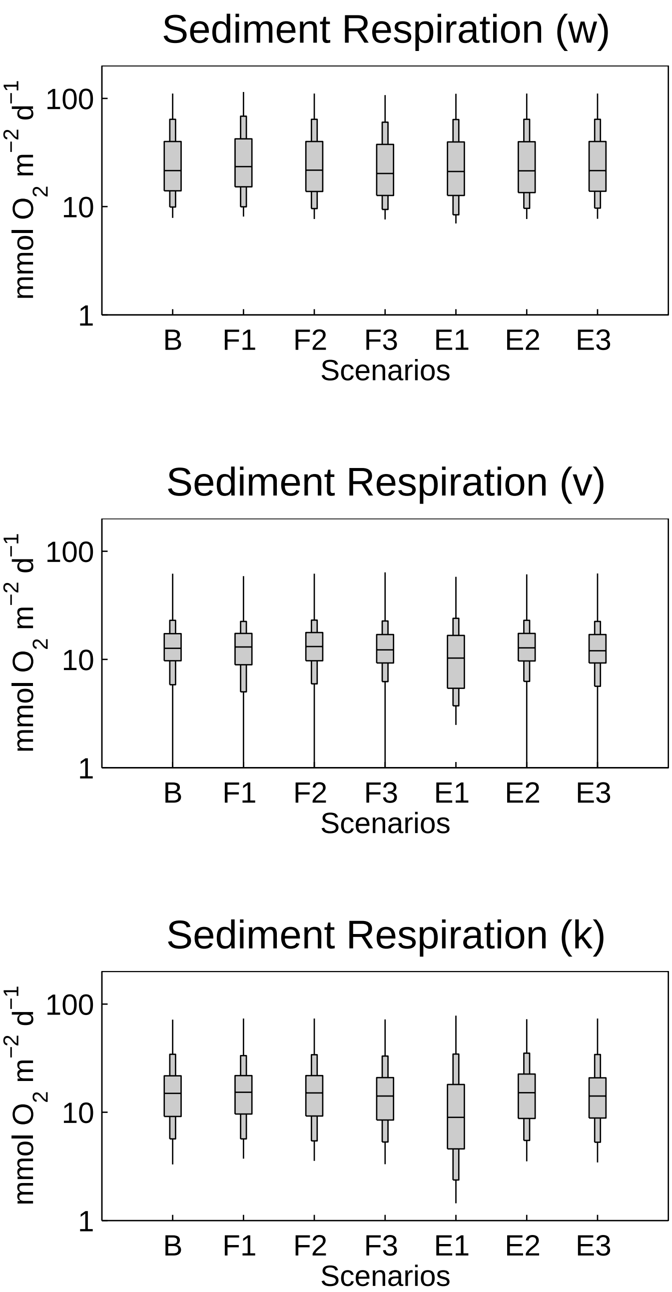


Fig. 29. Sediment respiration rate vs flow, exchange scenarios for L. Wellington (w), L Victoria (v), L. King (k).

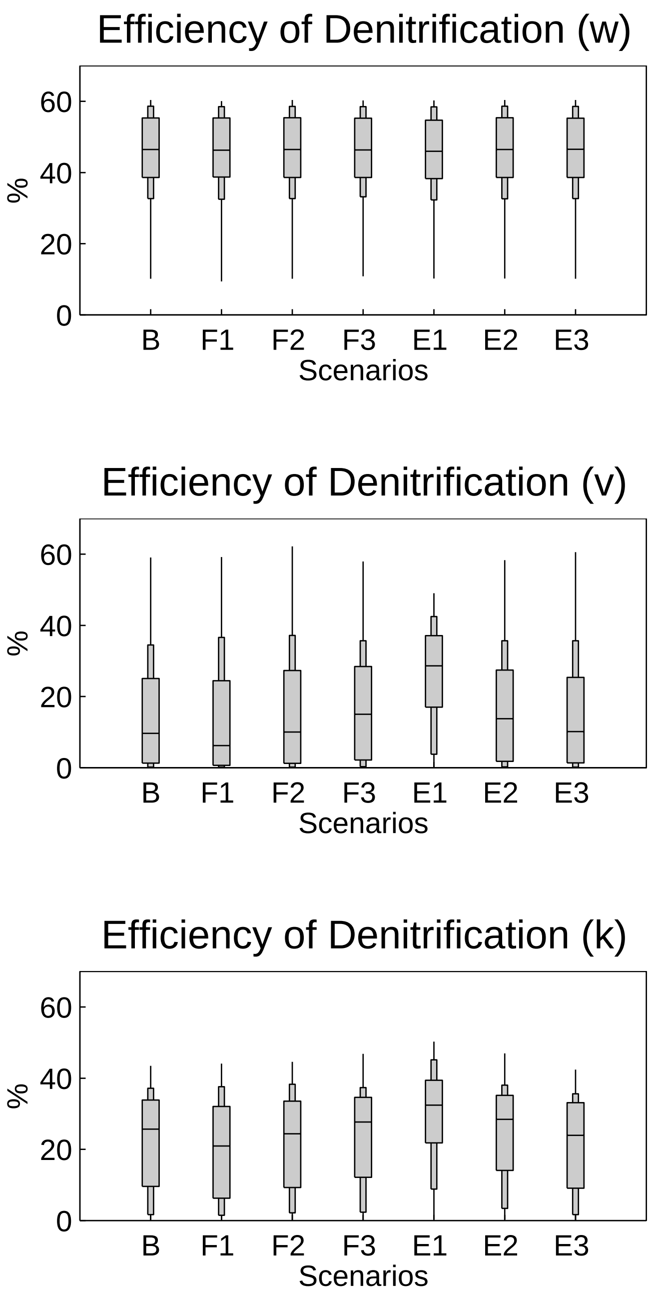


Fig. 30. Sediment denitrification efficiency vs flow, exchange scenarios for L. Wellington (w), L Victoria (v), L. King (k).

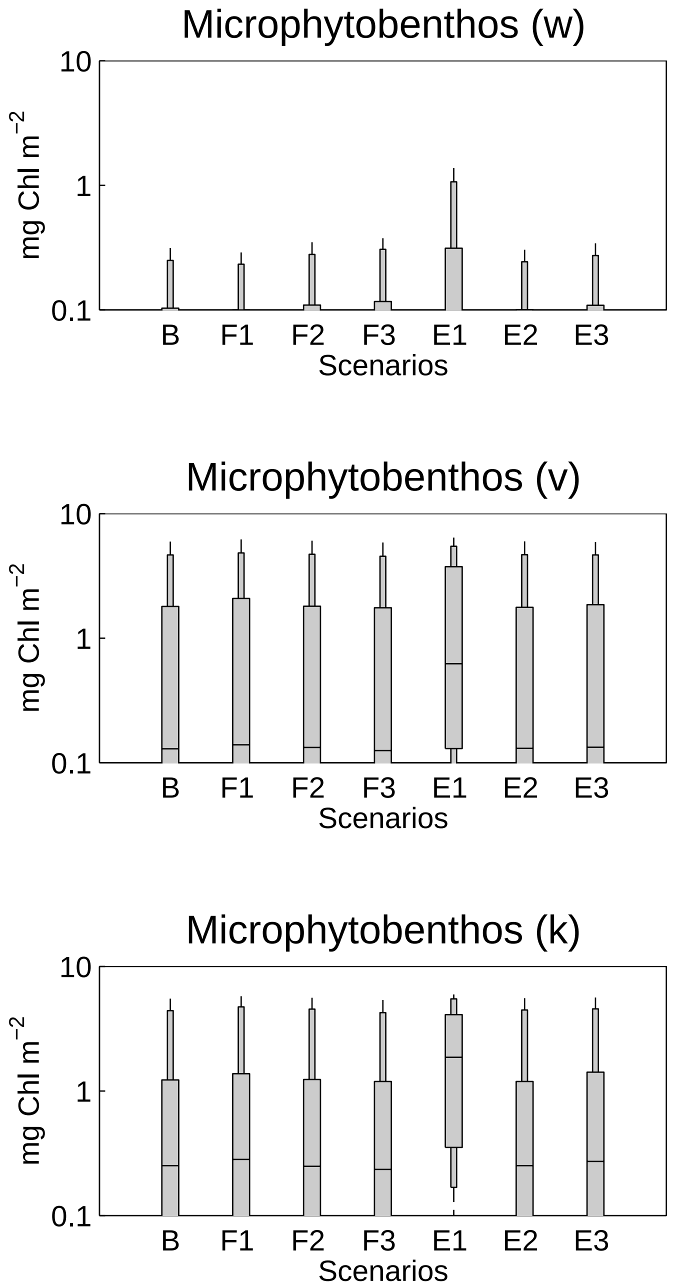


Fig. 31. Sediment microphytobenthos biomass (Chl a per unit area) vs flow, exchange scenarios for L. Wellington (w), L Victoria (v), L. King (k).

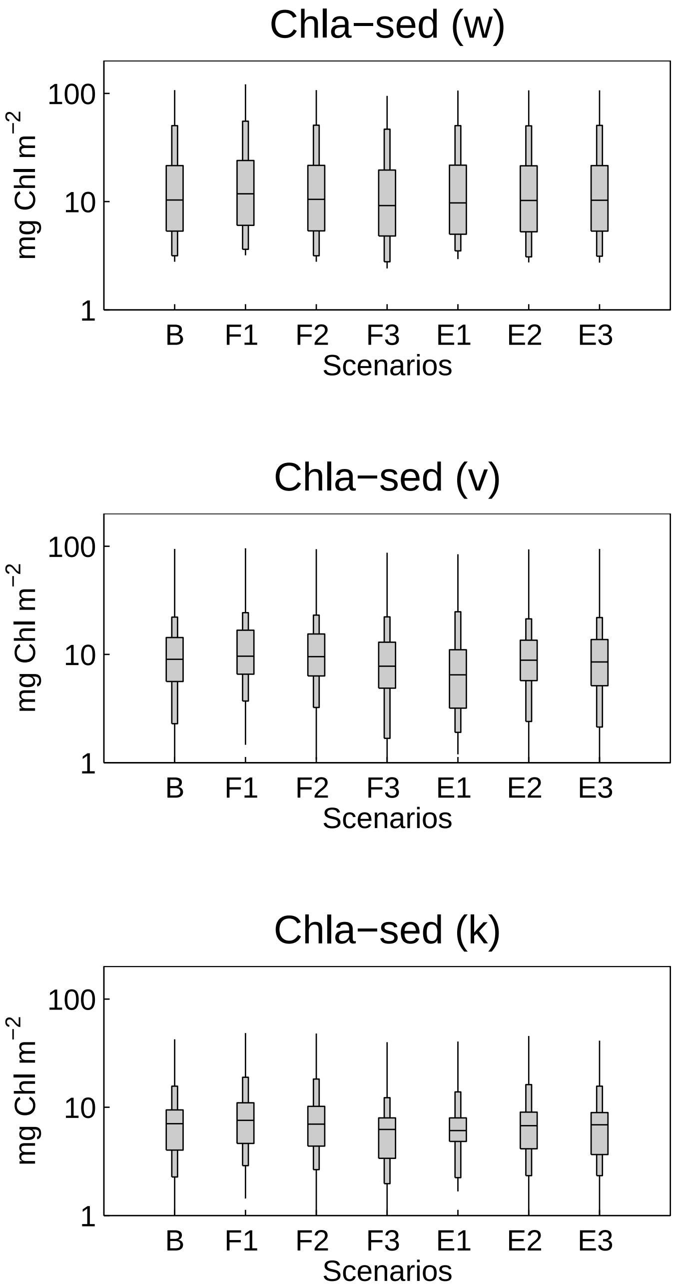


Fig. 32. Total sediment Chl a (per unit area) vs flow, exchange scenarios for L. Wellington (w), L Victoria (v), L. King (k). (Includes settled phytoplankton).

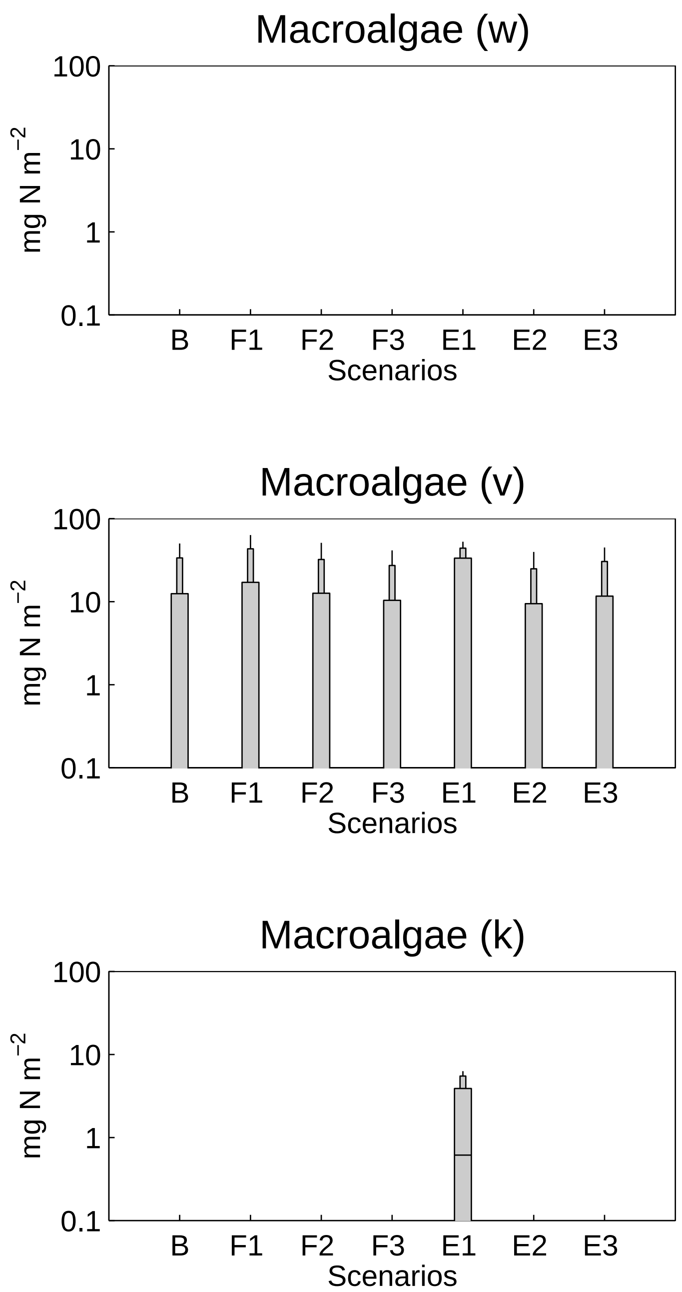


Fig. 33. Macroalgal biomass vs flow, exchange scenarios for L. Wellington (w), L Victoria (v), L. King (k).

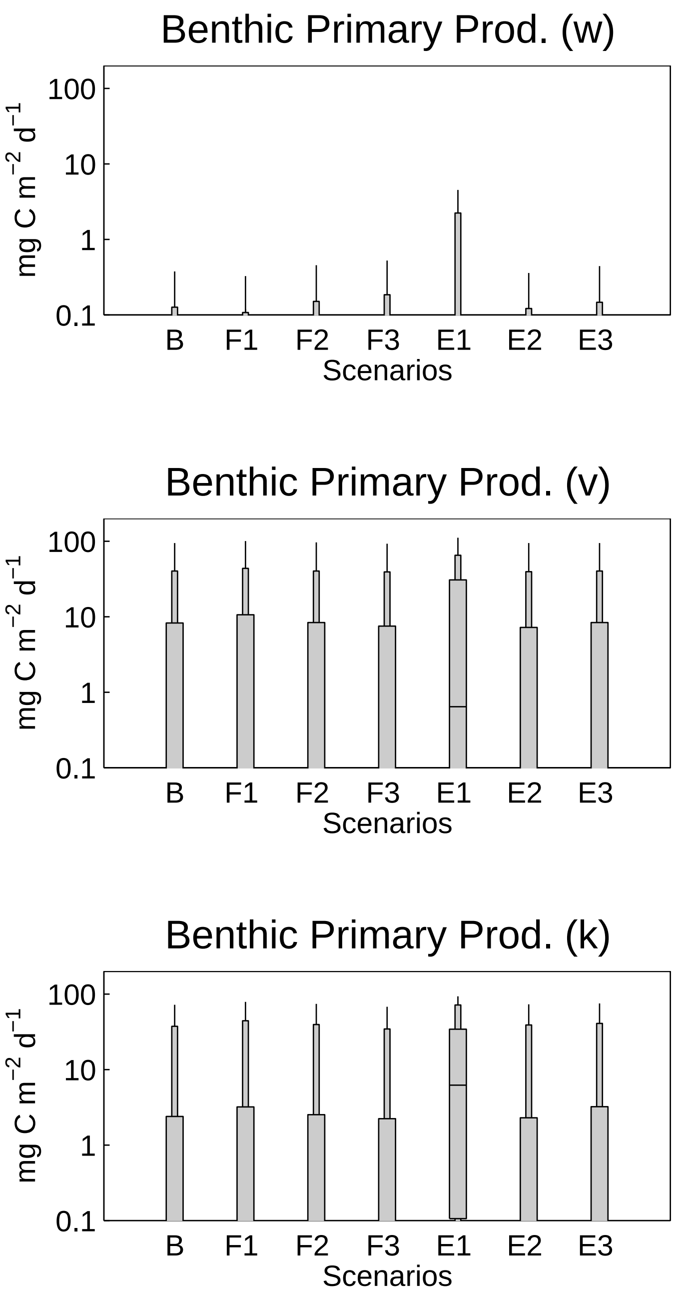


Fig. 34. Benthic primary production (macroalgae + microphytobenthos) vs flow, exchange scenarios for L. Wellington (w), L Victoria (v), L. King (k).

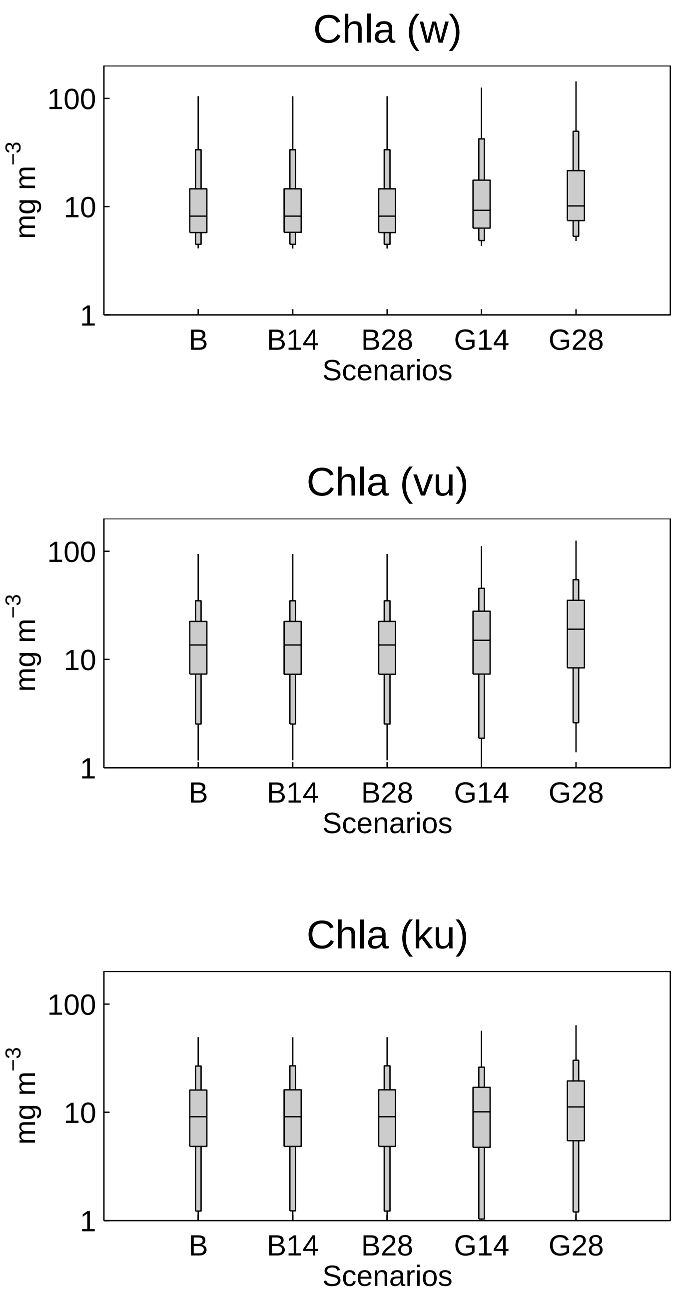


Fig. 35. Phytoplankton biomass (Chl a) for long-term scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

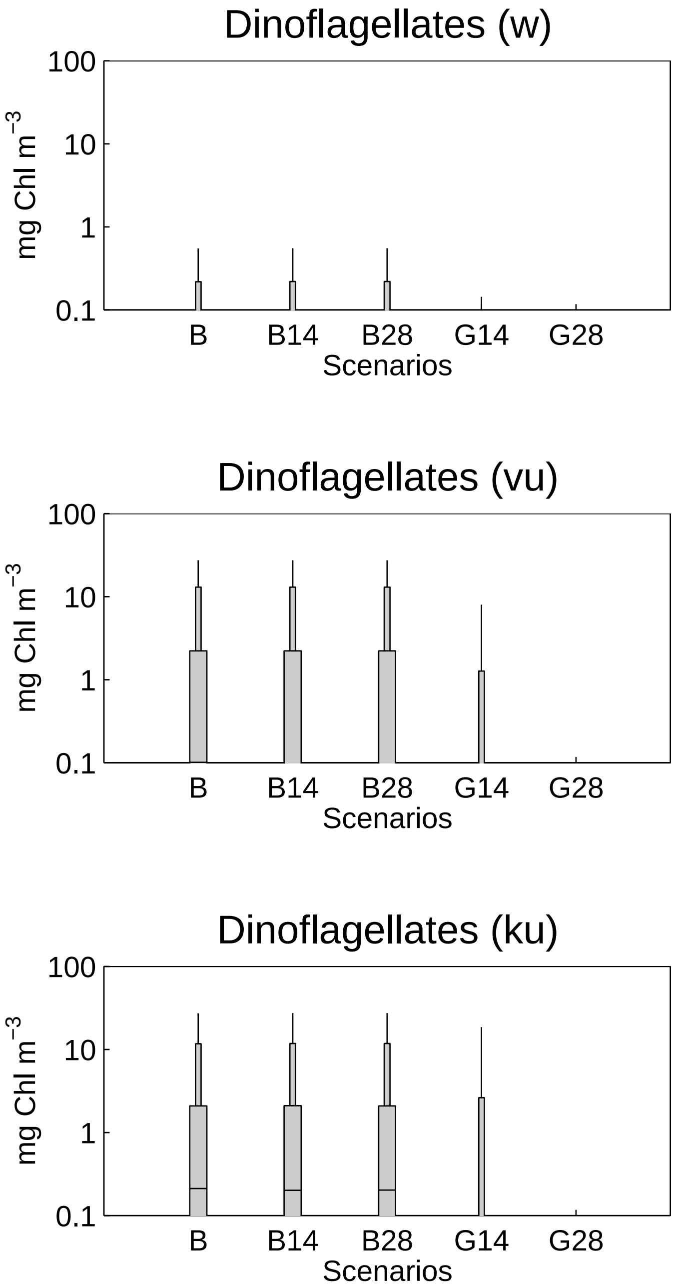


Fig. 36. Dinoflagellate biomass (Chl a) for long-term scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

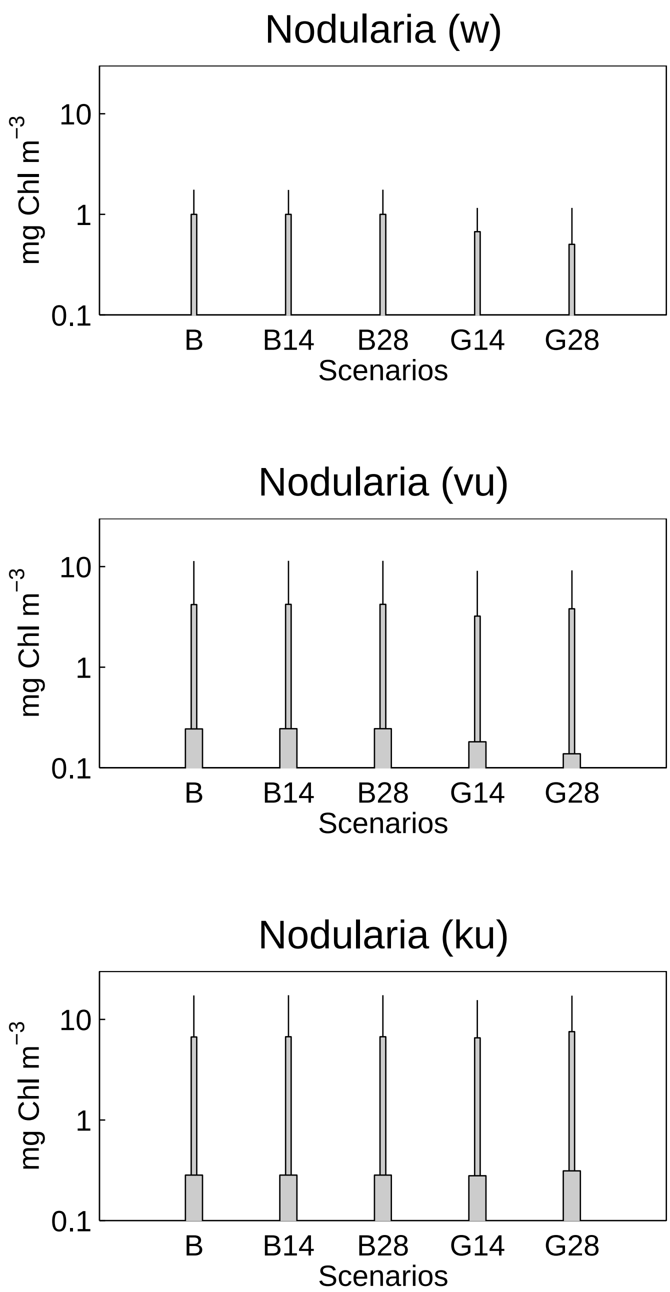


Fig. 37. *Nodularia* biomass (Chl a) for long-term scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

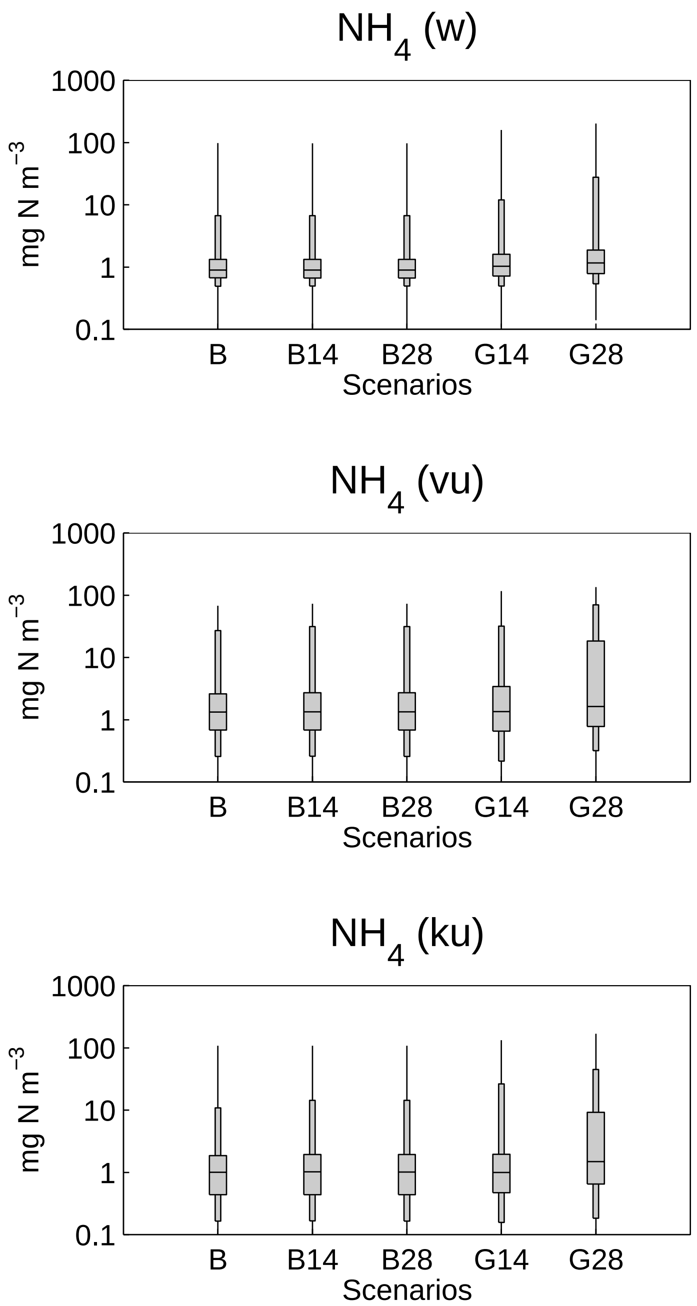


Fig. 38. Ammonia concentration for long-term scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

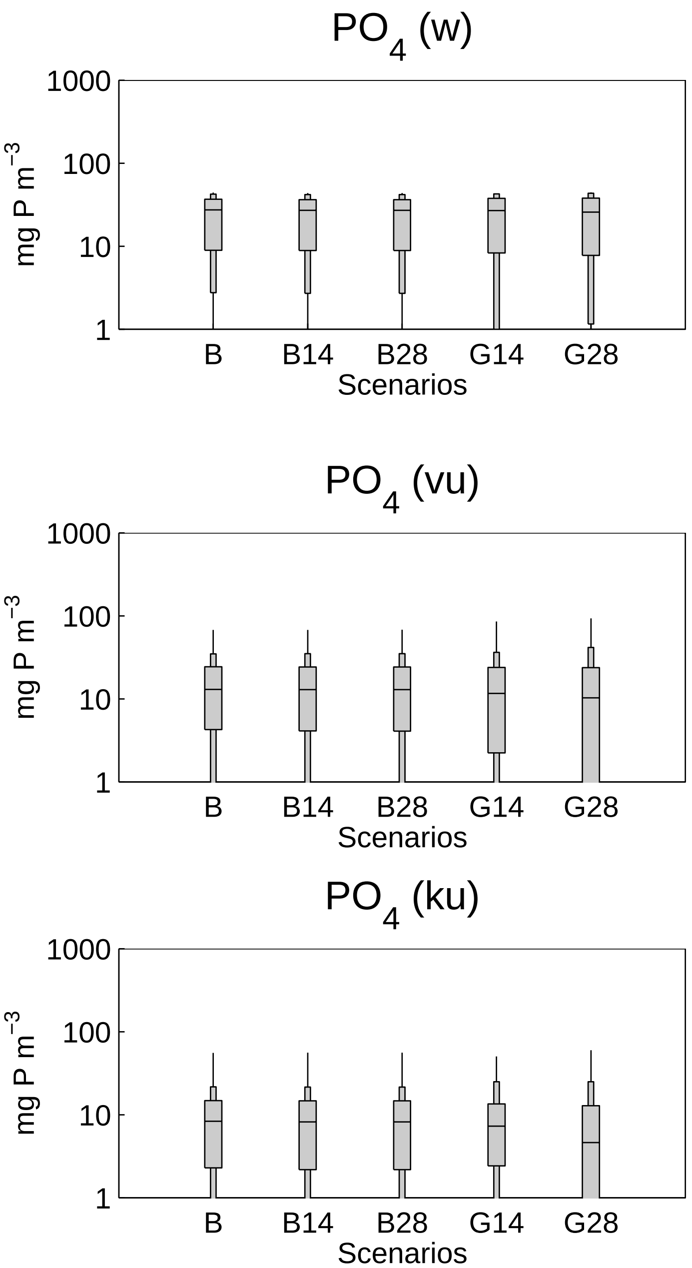


Fig. 39. Phosphate (DIP) concentration for long-term scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

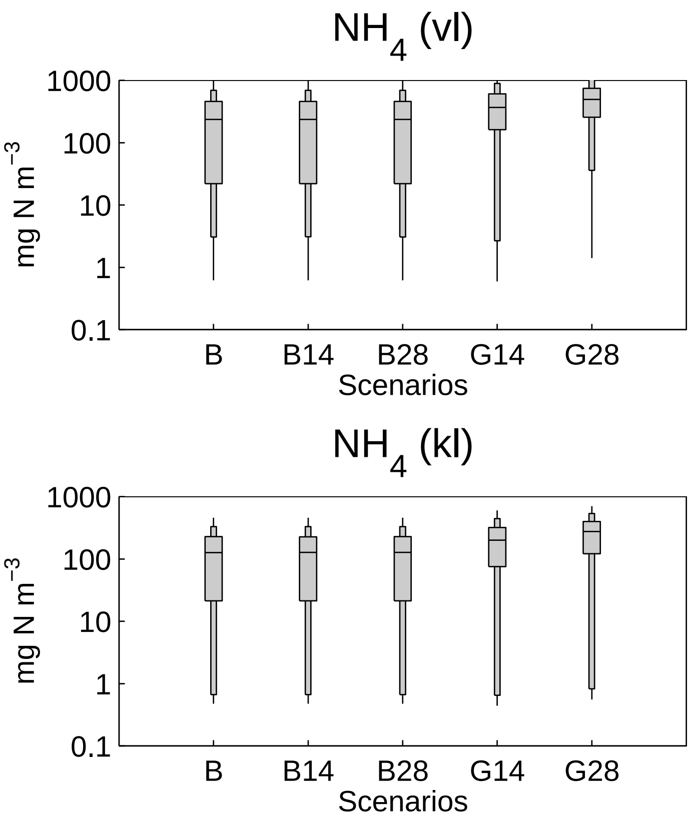


Fig. 40. Ammonia concentration for long-term scenarios for L Victoria bottom water (vl) and L. King bottom water (kl).

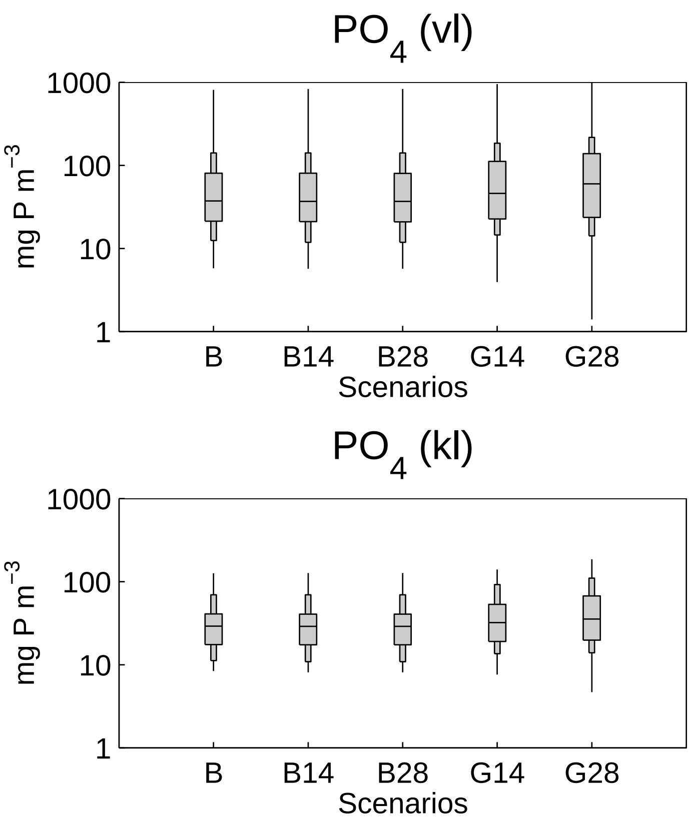


Fig. 41. Phosphate concentration for long-term scenarios for L Victoria bottom water (vl) and L. King bottom water (kl).

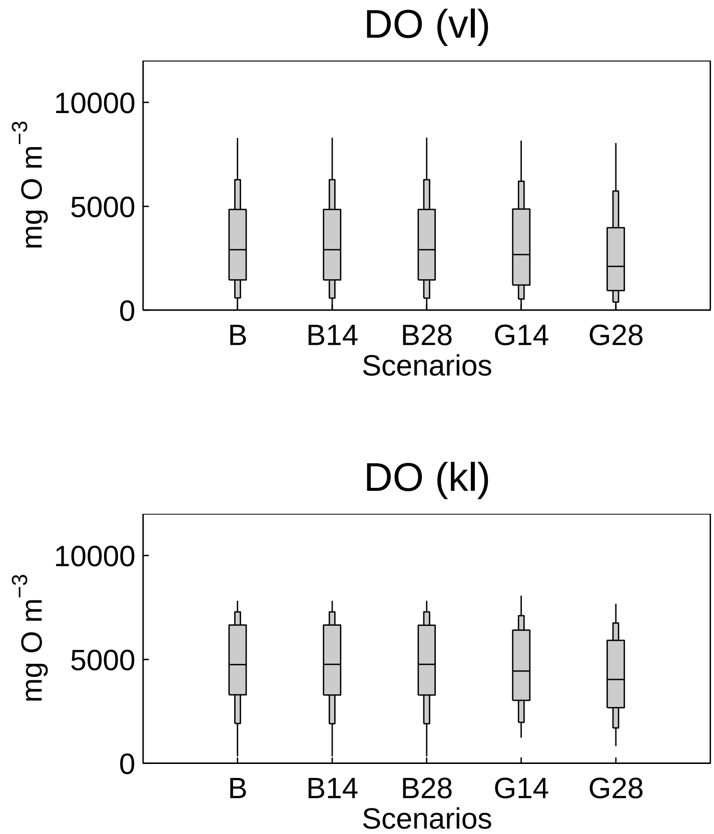


Fig. 42. Dissolved oxygen concentration for long-term scenarios for L Victoria bottom water (vl) and L. King bottom water (kl).

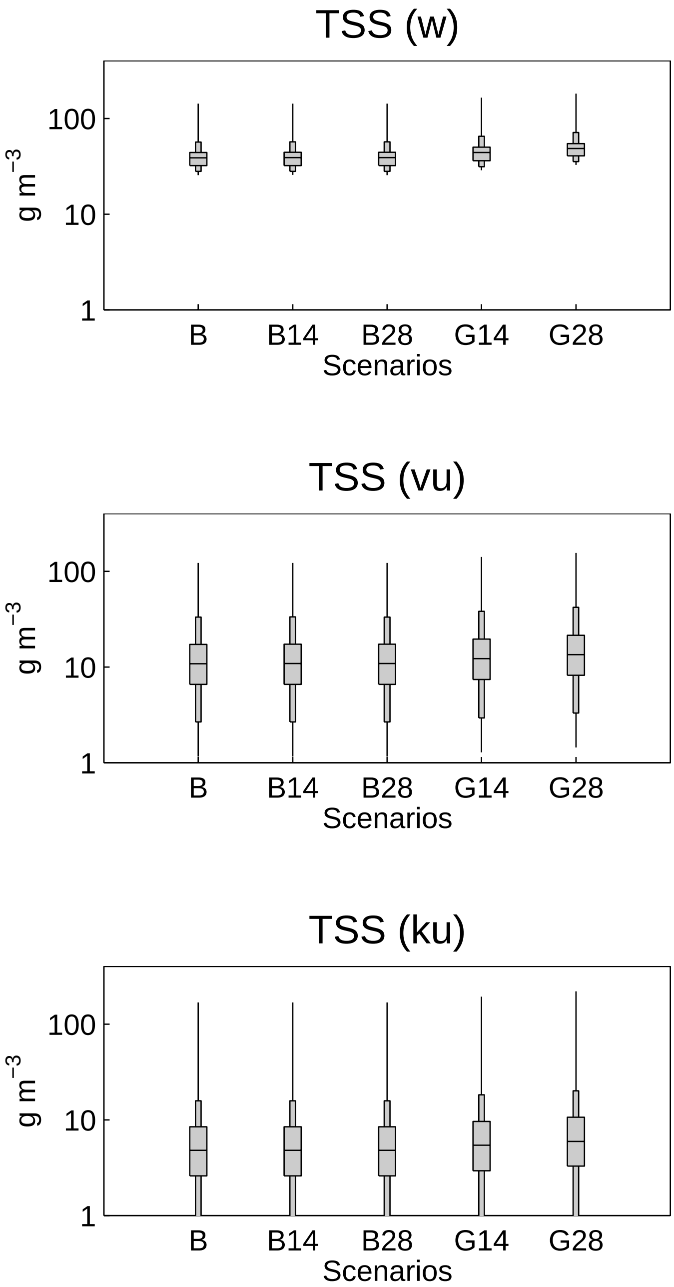


Fig. 43. Suspended sediment concentration for long-term scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

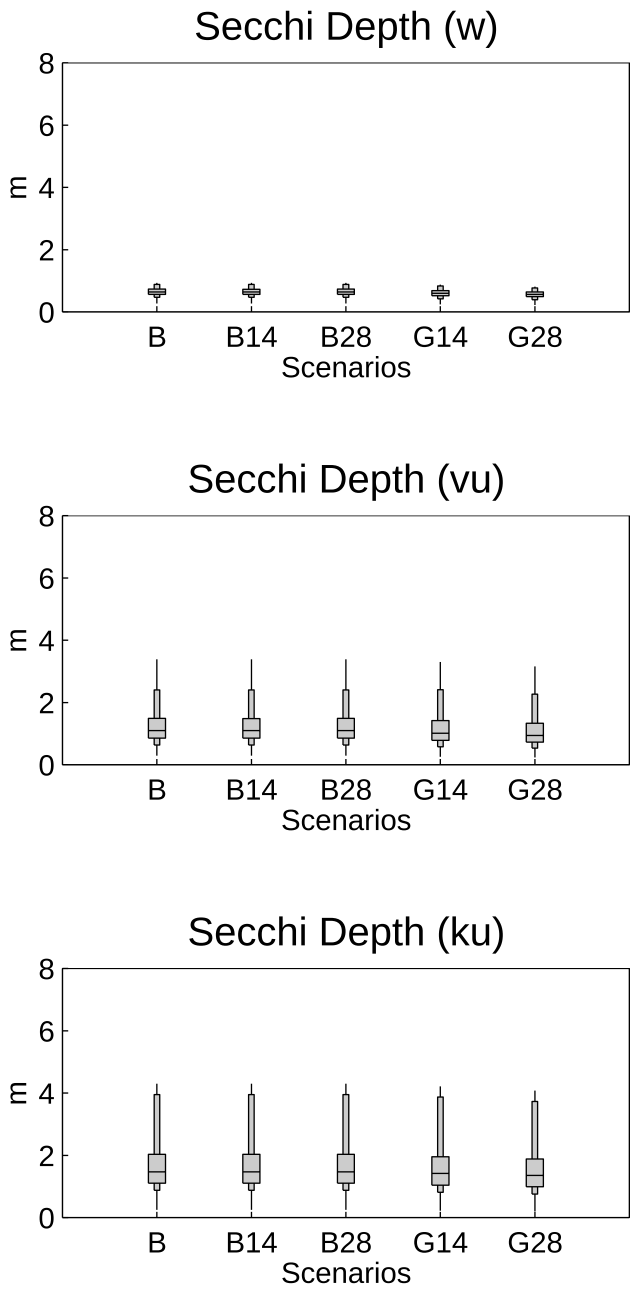


Fig. 44. Secchi depth for long-term scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

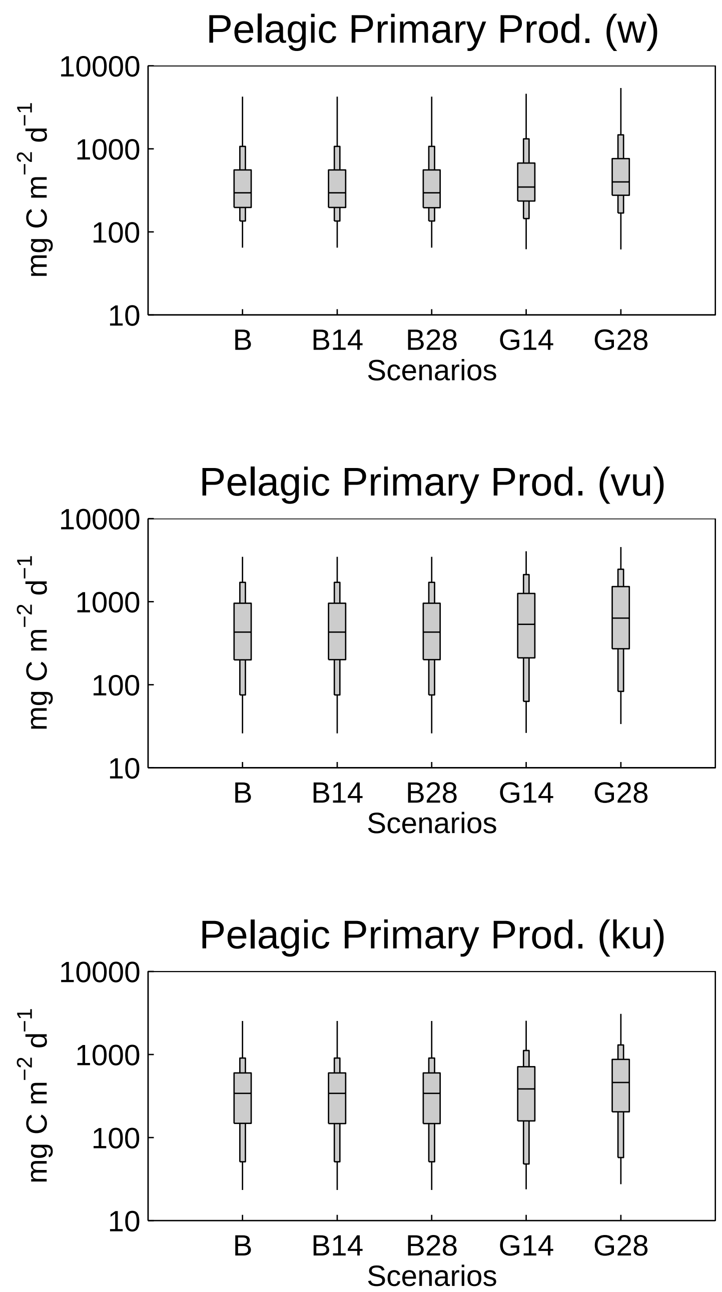


Fig. 45. Pelagic (phytoplankton) primary production for long-term scenarios for L. Wellington (w), L Victoria surface (vu), L. King surface (ku).

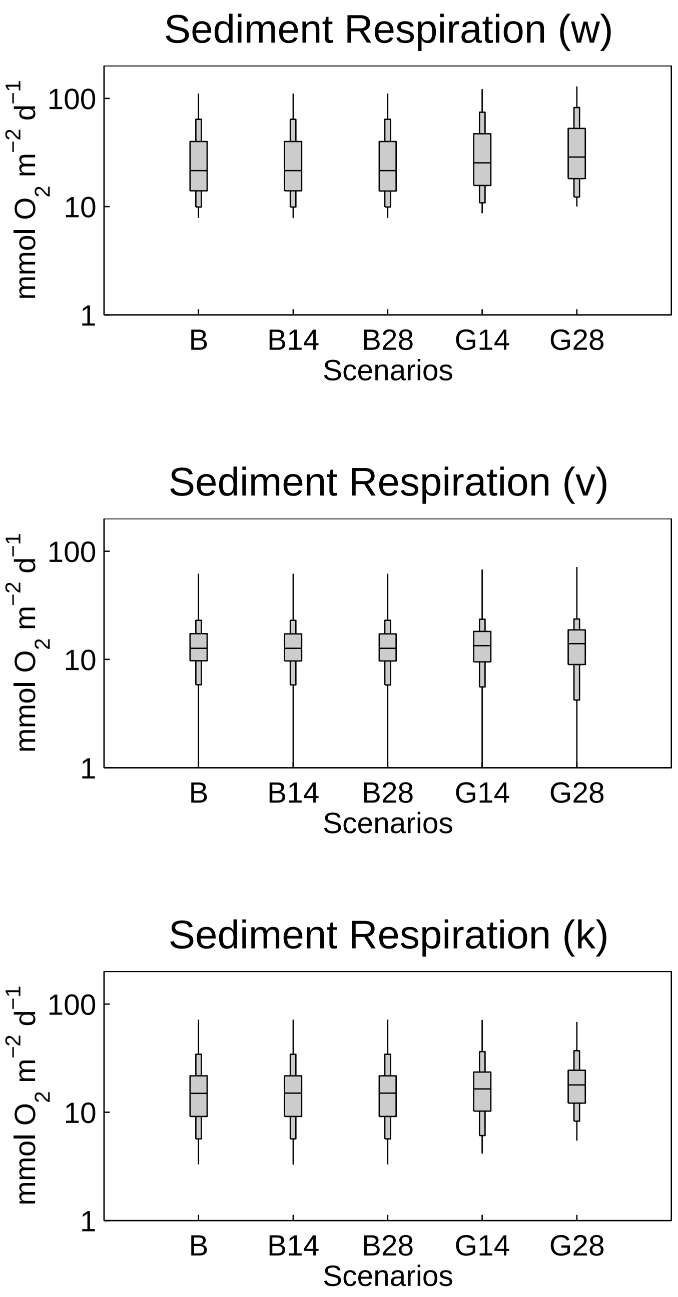
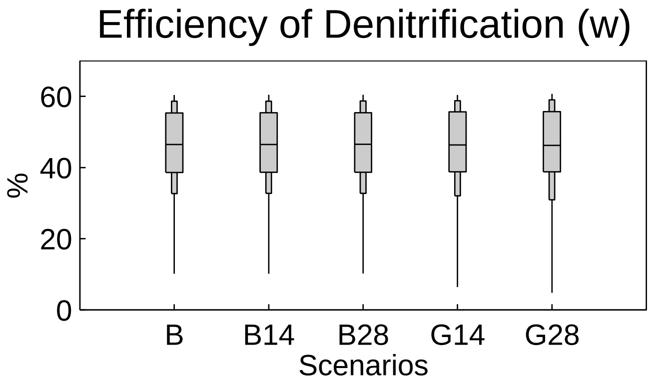
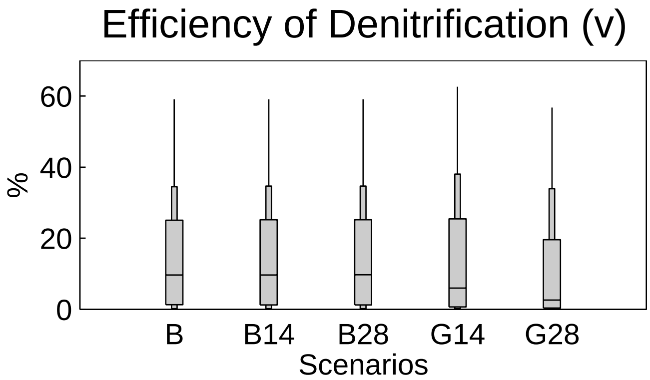


Fig. 46. Sediment respiration rate for long-term scenarios for L. Wellington (w), L Victoria (v), L. King (k).





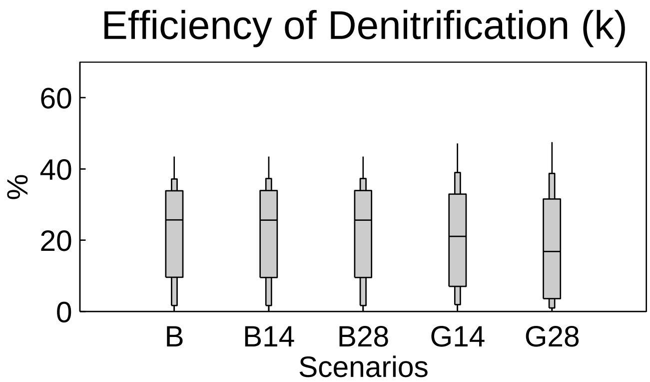


Fig. 47. Sediment denitrification efficiency for long-term scenarios for L. Wellington (w), L Victoria (v), L. King (k).

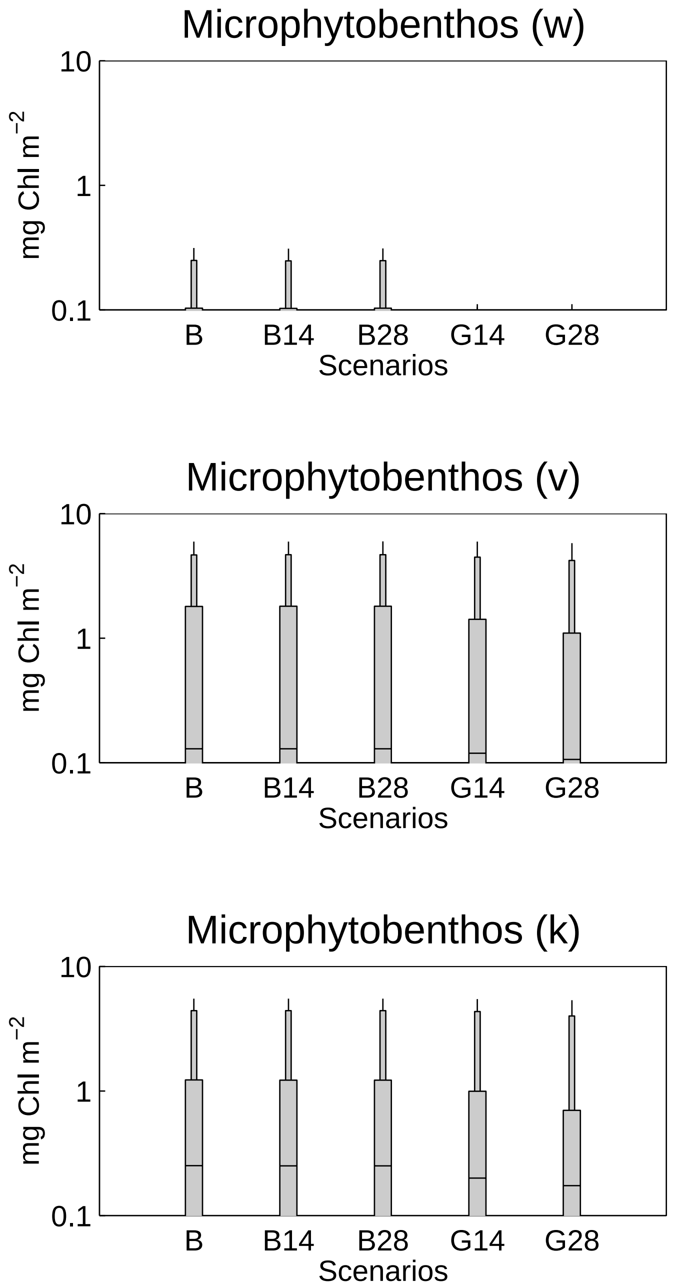


Fig. 48. Sediment microphytobenthos biomass (Chl a per unit area) for long-term scenarios for L. Wellington (w), L Victoria (v), L. King (k).

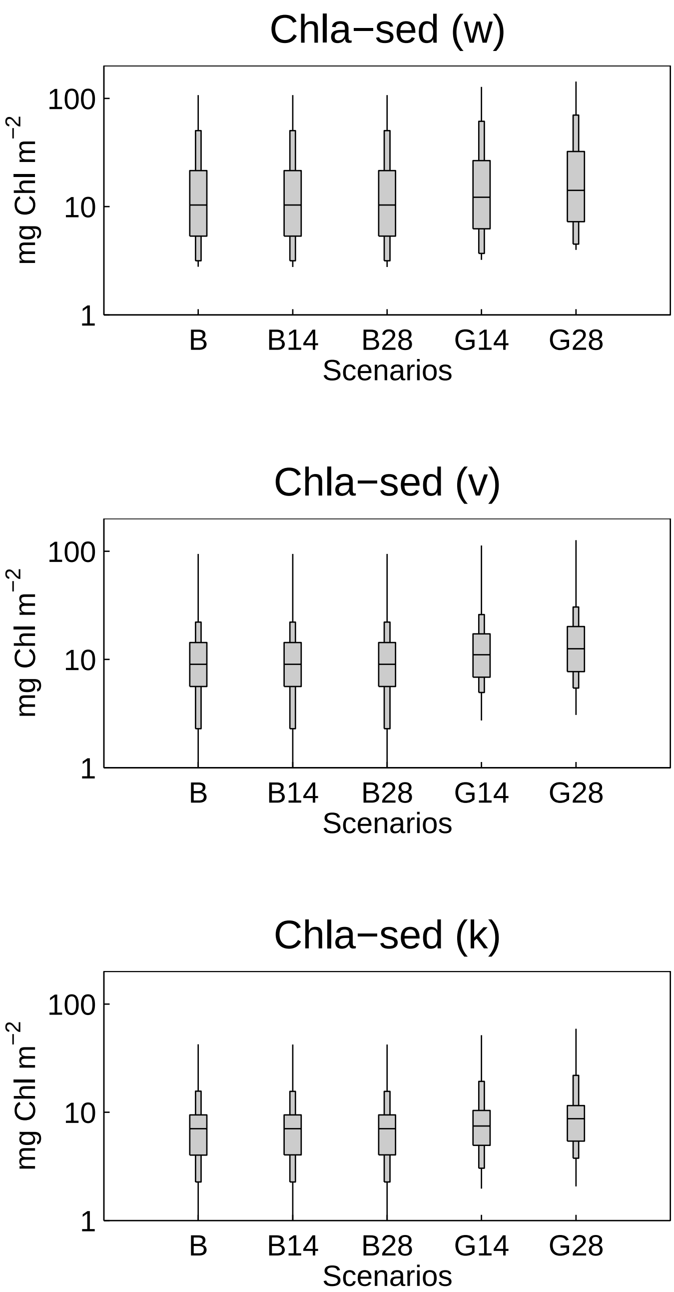


Fig. 49. Total sediment Chl a (per unit area) for long-term scenarios for L. Wellington (w), L Victoria (v), L. King (k). (Includes settled phytoplankton.)

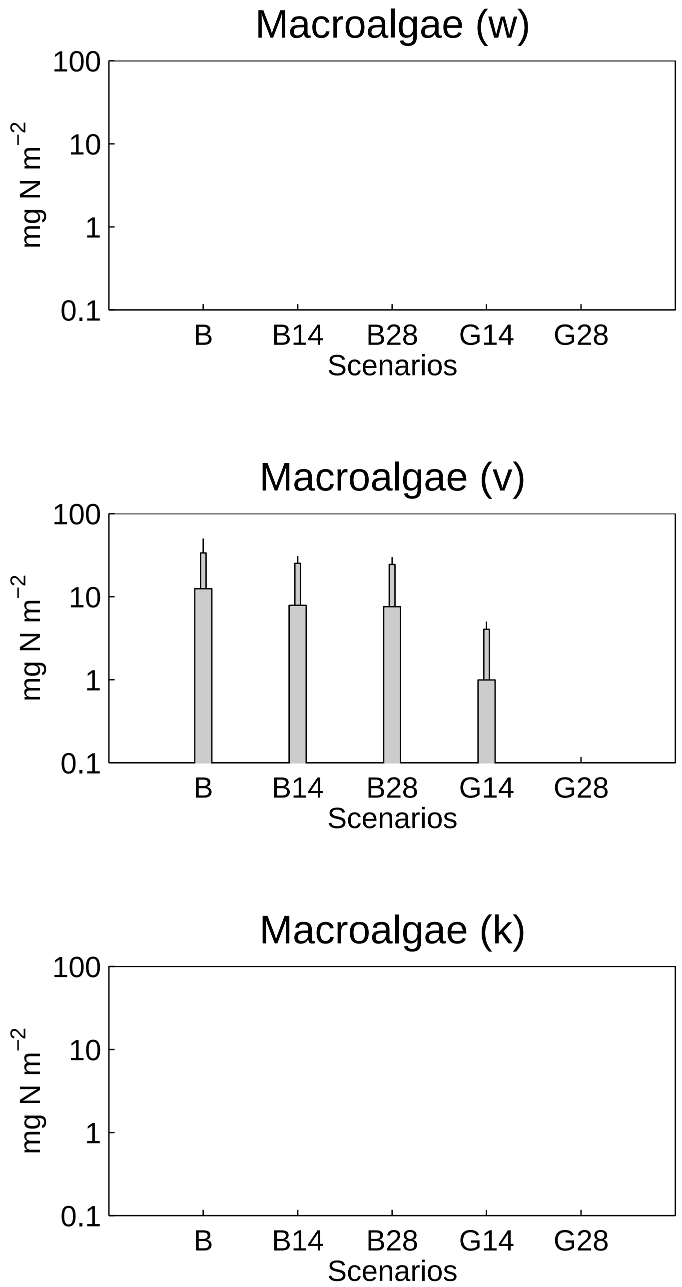


Fig. 50. Macroalgal biomass for long-term scenarios for L. Wellington (w), L Victoria (v), L. King (k).

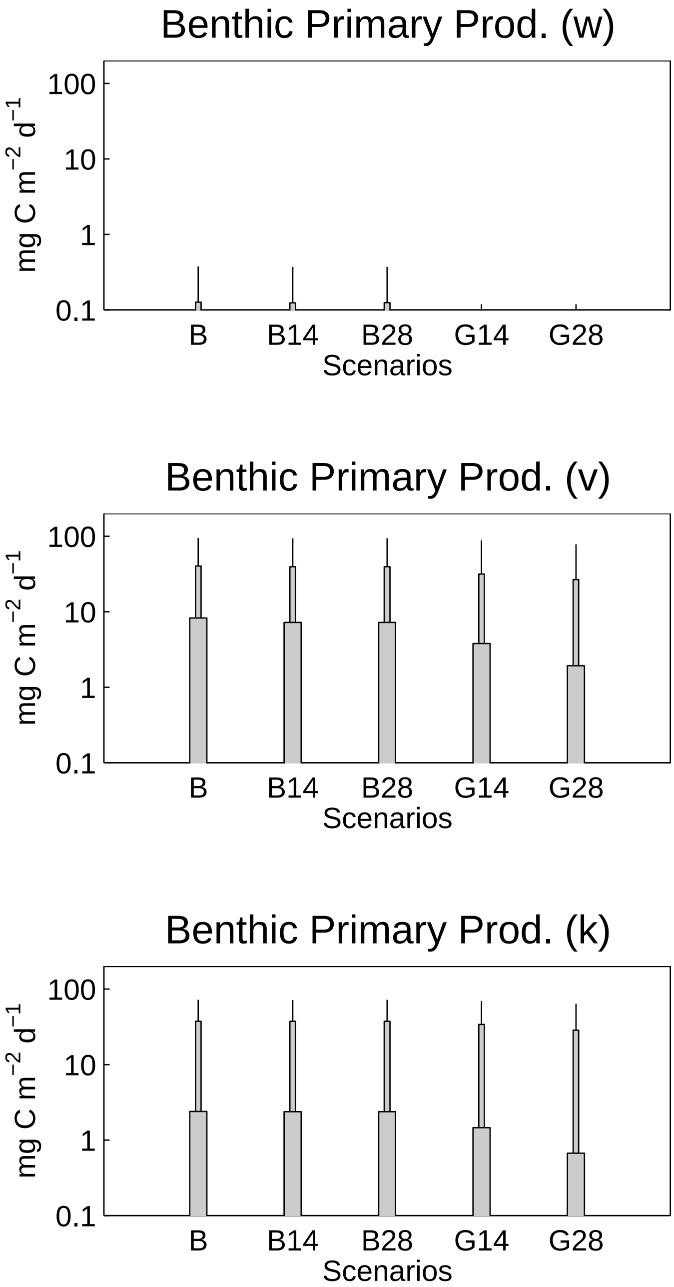


Fig. 51. Benthic primary production (macroalgae + microphytobenthos) for long-term scenarios for L. Wellington (w), L Victoria (v), L. King (k).