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Waituna Lagoon Modelling:
Developing quantitative assessments to assist with lagoon management

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Executive Summary

A decision support tool has been developed to assist with managing *Ruppia* beds and maintaining the biological diversity of Waituna Lagoon. The results from this tool are presented as outputs from model simulations of physical, chemical and biological variables within the lagoon. Input data for the model simulations were adjusted to simulate sensitivity of the model to (i) input data quality and quantity, (ii) complexity and formulations used in the model itself, (iii) parameters used to adjust physical and biogeochemical environmental responses in the model, and (iv) potential management options. Parts i-iii were necessary to assure a level of confidence in the model so that part iv, the management options, could be used to provide a basis for what might be required to sustain persistent, productive *Ruppia* beds in the lagoon.

Waituna Lagoon, and the wider Awarua complex, was designated a Ramsar site in 1976, signifying a wetland of international importance. The lagoon has high biological diversity and a number of rare and endangered species. New Zealand's obligations to the Ramsar Convention (1971) require that Waituna Lagoon be managed by statutory authorities to preserve habitats that are important to sustain its endemic species. A species that has been identified as critical for sustaining biodiversity in Waituna Lagoon is the macrophyte *Ruppia*, considered a "keystone" species. Two species of *Ruppia* occur in Waituna Lagoon; *Ruppia polycarpa* and *Ruppia megacarpa*, and they are often found within the same ecosystem but are reproductively isolated.

The present report was initiated because of concern that *Ruppia* beds may be threatened by increasing nutrient loads from the catchment, and recent observations of *Ruppia* population decline. Conversions of relatively low-intensity sheep and beef farms to dairy farms over the past 10-15 years, as well as establishment of pasture in areas of peat which would otherwise be inundated during high lagoon water levels will increase nutrient concentrations in tributary and groundwater inputs to the lagoon. A possible sequence of events that would lead to loss of *Ruppia* might be nutrient-stimulated increases in epiphytic algae and macroalgae that would shade and stress *Ruppia* populations and, with further increases in nutrient supply, a subsequent increase in phytoplankton biomass as macroalgae were in turn shaded out by increasing phytoplankton biomass. This sequence (sometimes referred to as a "regime-shift") may be reinforced by loss of stability of the shallower bed sediments as submerged macrophytes disappear, compounding reductions in water column light availability and restricting any regeneration of *Ruppia*.

The modelling approach adopted in this study used a deterministic numerical model (DYRESM-CAEDYM) which required specification of sufficient variables so as to adequately capture the dynamics of the ecosystem. DYRESM-CAEDYM is a coupled hydrodynamic-ecological model that is one-dimensional in spatial representation, i.e., it represents the lagoon as a series of horizontally homogeneous layers that are stacked on top of each other. The state variables selected for the ecological component of the model, CAEDYM, included three groups of phytoplankton (broadly representing cyanophytes, cryptophytes and diatoms), a single group of filamentous macroalgae and a single group that represented the two *Ruppia* species. The inclusion in CAEDYM of the state variables macroalgae and *Ruppia* required development of specific algorithms to represent processes such as shading (e.g., macroalgae that tend to cover and shade *Ruppia*), effects of salinity, and dynamic feedback between *Ruppia* biomass and resuspension of sediment, organic matter and phytoplankton.

An analysis was undertaken to quantify model sensitivity with a view to defining how the simulation was affected by a range of potential errors associated with input data (e.g., meteorology, inflows), selective removal of key state variables and processes (e.g., *Ruppia*, sediment resuspension) and parameters that adjusted the kinetics of biogeochemical processes. The results were not unexpected; *Ruppia* was a key state variable, in agreement with its keystone species status. Macroalgae were also important in terms of their system-wide effects, and sediment resuspension strongly affected all of the primary producers both directly (e.g., via direct transfers of biomass from the bottom sediments to the water column) and indirectly (e.g., via attenuation of light by resuspended sediments). Growth and loss rates for *Ruppia* and macroalgae were identified as sensitive parameters in terms of their effects not only on the biomass of the relevant state variable but also as a result of flow-on effects to many other components of the Waituna Lagoon ecosystem.

The “base” simulation for Waituna Lagoon consisted of obtaining a best fit of simulated state variables to measured (or derived) state variables which included *Ruppia* biomass derived from areal coverage surveys, nutrients (total phosphorus, total nitrogen, nitrate, ammonium and dissolved reactive phosphorus), suspended sediment, chlorophyll *a* (as a proxy for total phytoplankton biomass) and macroalgae biomass (based on visual estimates). The model parameters were adjusted for a calibration period of Oct 2001-Oct 2007 and remained fixed for a validation period of Oct 2007-Oct 2011. A range of statistical measures was used to assess the model fit for the calibration and validation periods, and these measures indicated that the model fit to the observed data was amongst the best of any applications documented in the literature, despite some limitations imposed by detection limits for nutrients and chlorophyll *a*, and with a caveat that some measures of primary producers were based only on visual assessments of percentage cover (*Ruppia* and macroalgae).

A series of model simulations were run to examine how the model responded to an imposed set of environmental conditions and management regimes different from those of the calibration and validation period. These scenarios were based on scientific inputs provided through the Waituna Lagoon Technical Group, Environment Southland and iwi, and categorised broadly into changes in (i) hydrology and climate, (ii) catchment-derived nutrient and sediments, (iii) both hydrology and nutrients. The scenarios were used to assess whether various management scenarios could meet goals to prevent a “regime shift” and sustain a “healthy” *Ruppia* population in the lagoon. A “healthy” *Ruppia* population was defined as being both abundant and stable, with average *Ruppia* biomass similar to that observed in 2007, a year described by Stevens and Robertson (2007) as having high *Ruppia* coverage, and with minimum biomass not less than that observed in 2009, the beginning of a period of *Ruppia* decline in the lagoon (Robertson and Stevens, 2009; Robertson and Funnell, 2012). A “regime-shift” was defined as an ecological change in which macroalgae and/or phytoplankton dominate primary producer biomass and *Ruppia* biomass declines to very low levels.

The hydrological scenarios involved changes to the lagoon opening regime and demonstrated that the timing and duration of opening of the lagoon had a significant impact on lagoon water quality and ecology. Scenarios that did not include actively managed openings, or raised the opening trigger level so that openings were very infrequent over the simulation period, resulted in a collapse in *Ruppia* biomass as macroalgae and subsequently, phytoplankton biomass increased, meeting the criteria of a “regime shift”. The absence of regular tidal flushing of nutrients transported to the lagoon via surface and groundwater inflows, promoted growth of phytoplankton and macroalgae. Hydrological scenarios involving openings ranging from 3-6 months resulted in biomass of *Ruppia*, macroalgae and phytoplankton similar to the “base” case. These scenarios indicate that it is not possible to maintain a “healthy” *Ruppia* population in the lagoon with changes to the opening regime alone, i.e., nutrient load reductions are required simultaneously. Regardless, opening the lagoon for long periods would represent a shift away from the more natural state of the lagoon as a coastal lake, and towards a periodically estuarine ecosystem.

Current catchment nutrient loads are highly influenced by anthropogenic, or “non-natural”, inputs. The nutrient and sediment reduction scenarios that were relatively low in magnitude (i.e., 10-25% reduction of current loads, when current loads may represent c. 100-fold increase in nitrogen and 10-fold increase in phosphorus relative to “natural” loads), resulted in only a small change in modelled variables. Nutrient reduction scenarios that included a reduction in nitrogen loading of 50% or more resulted in increased *Ruppia* biomass, decreased macroalgae biomass and chlorophyll *a* concentrations. There was an approximately proportional reduction in modelled chlorophyll *a* with reductions in nutrient concentrations. A scenario involving a 50% reduction in nitrogen and 25% reduction in phosphorus loading resulted in an abundant and stable *Ruppia* biomass on an inter-annual basis, we well as reduced macroalgae and chlorophyll *a* concentrations.

If the lagoon is not opened at all (i.e. natural opening regime) then very substantial nutrient load reductions (70-90% reduction in nitrogen and phosphorus) are required to obtain a “healthy” *Ruppia* population, and to reduce macroalgae and phytoplankton biomass to low levels. However, 3-month winter openings combined with a 50% reduction in nitrogen loading and 25% reduction in phosphorus loading resulted in a “healthy” *Ruppia* population and reduced macroalgae and phytoplankton biomass, consistent with research on nutrient loading thresholds for sustaining macrophytes in lagoon ecosystems by Scanes (2012), Schallenberg and Schallenberg (2012) and Wriggle (2012).

Simulation of lagoon hydrodynamics with the three-dimensional model (ELCOM) indicated that when the lagoon is open salinity distributions may be horizontally and vertically variable, particularly in the eastern arm of the lagoon. This is an area that has not historically been the focus of water quality monitoring, despite the proximity to *Ruppia* beds in this part of the lagoon. The three-dimensional model simulations also showed substantial short-term variations of the water level (by up to 0.5 m) at the eastern end of the lagoon in association with the strength of the prevailing westerly winds. Anecdotal observations of scouring of accumulated fine sediments during lagoon opening were reinforced in simulations which gave outputs of very high water velocities for grid cells near the lagoon opening during the initial opening phase. A remote sensing scoping exercise was also undertaken in this study which explored the possibility of using Landsat images to measure chlorophyll *a* in the lagoon. However, remote sensing may have limited potential as a water quality monitoring tool for Waituna Lagoon, due to the shallowness and bottom reflectance present.

The model applications to Waituna Lagoon have highlighted complex interactions and processes that have a significant effect on the lagoon ecology. Results from this study are consistent with previous research that suggests that increasing eutrophication in this type of system tends to result in, firstly, dominance of macroalgae over *Ruppia* beds, with subsequent dominance of phytoplankton over macroalgae. Results indicate that it is not possible to maintain an abundant and stable *Ruppia* population in the lagoon with changes to the opening regime alone. If the lagoon is not opened at all, substantial (70-90%) nutrient load reductions are required; alternatively winter openings in combination with nutrient load reductions of 50% nitrogen and 25% phosphorus are likely to maintain a healthy lagoon ecosystem, consistent with other research on nutrient loading thresholds for macrophyte health in coastal ecosystems.

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Ecosystem modelling glossary

This glossary is intended to aid the non-modeller to understand and evaluate a modelling study (and is particularly applicable to this study). It is not an exhaustive list of ecosystem modelling terminology. A scientific model aims to represent variables and processes in a logical and objective way, but is necessarily a simplified view of a complex reality. Models are generally used when it is impossible or impractical to create experimental conditions in which outcomes can be directly measured. For example, for simulating large-scale experiments, or processes over a long period of time. Models can be used to simulate the complex interactions amongst physical and biogeochemical processes that may not readily included in empirical or statistical relationships amongst variables.

Ecosystem modelling

An ecosystem model is a simplified representation (usually mathematical) of an ecosystem. The model is generally utilised to gain understanding of, and make predictions about, the dynamics of the system. The complexity of such models can vary significantly depending on the research needs and questions, and the modelling objectives.

Conceptual models

Conceptual models are often used to help understand or visualise an ecosystem, and as a starting point before the interactions between model components are quantified mathematically (such as in process-based or statistical models). Conceptual models may often be refined as a result of new insights or understanding gained after undertaking mathematical or simulation modelling, or conducting experimental studies. Thus, a conceptual model for an ecosystem may be refined through time to reflect evolving scientific knowledge.

Process-based models

Process-based models attempt to describe the relevant processes of interest (e.g., biogeochemical cycles, biological interactions etc.) in an ecosystem using a series of differential equations, and then to reproduce data collected in the field. These types of models may also be described as deterministic or simulation models. Process-based models require calibration and validation (preferably on independent datasets) before the model is able to be used in a predictive capacity to undertake scenario testing.

Statistical models

Statistical models attempt to fit a model (e.g., an equation) to describe and predict relationships between data, rather than explicitly representing the underlying processes in an ecosystem. Because of the lack of process representation and the empirical nature of statistical models, extrapolating results beyond the data range with which the model was constructed is generally invalid.

Parameters

Model parameters in process-based models are generally entered as constant inputs, but parameter values may be adjusted within the model simulation depending on other state variables that vary through the duration of a simulation. Complex process-based models may have one-hundred or more parameters relating to physical, chemical, or biological processes, such as maximum nutrient release rates from the sediment, or maximum growth rates for phytoplankton, etc. Model parameters are defined using data specific to the system being modelled, or based on available literature for similar ecosystems. The full set of model parameters become specific to each application as during the process of model calibration parameters are adjusted within literature ranges to improve the agreement between modelled variables and field data.

Forcing data (also called boundary conditions)

Forcing data are the input data used to "drive" the model. As with model parameters, the forcing data required for each model application will depend on the exact model configuration. Process-based coupled hydrodynamic-ecological models (e.g. DYRESM-CAEDYM, which is used in this study) require time series forcing data for the hydrodynamic model (e.g. meteorological data, inflow and outflow volumes, water temperature and salinity) and the ecological model (e.g. inflow nutrient and sediment concentrations) resolved at daily or sub-daily time steps.

Calibration and model performance statistics

Model calibration is the process of the estimation and adjustment of model parameters (within literature ranges) to improve the agreement between model output and field data (Rykiel, 1996). Both visual inspection of model output and field data, and quantification of model performance using statistical measures are important and necessary to assess the accuracy of the model output. A range of model performance statistics are typically used to assess the "goodness-of-fit" of model output. Pearson's product-moment correlation coefficient (R) is often used as a measure of model performance, because it is familiar and intuitive to both modellers and non-modellers, particularly as R^2 is commonly understood to be the proportion of the "variance explained". However, it has long been argued that the magnitude of R is not consistently related to the accuracy of model prediction (Willmott, 1982). For example, the correlation between model output and field observations can approach 1 whether or not there is a consistent offset between the two. Also, it should be remembered that there is inherent variability in both the model output and field observations, thus the use of R^2 as a measure of the "variance explained" becomes confounded. Rather than focusing on one measure it is appropriate to report on several indices that quantify different aspects of model performance. Average difference can be described by the root mean square error (RMSE) and/or the mean absolute error (MAE), with MAE being less sensitive to extreme values. Consistent under- or over-prediction can be assessed using the mean signed difference (MSD), the mean of the predicted values, and the mean of the observations. Thus, during the process of model calibration the modeller has to aim to minimise RMSE, MAE, and MSD, but also to maximise R.

Validation

Rykiel (1996) defines validation as "a demonstration that a model within its domain of applicability possesses a satisfactory range of accuracy consistent with the intended application of the model". Model validation assesses model performance and its predictive power by comparing model output with field data not used in model development and calibration (i.e. the model calibration and validation use separate forcing and field observation datasets). As with calibration, statistical measures of model performance are used to assess the model accuracy and compare performance between the calibration and validation periods.

Sensitivity and uncertainty analyses

Sensitivity analyses explore and quantify the effect of potential errors in input data and/or the effect of alternative assumptions on modelled variables, with results illustrated graphically or numerically (Loucks *et al.*, 2005). Uncertainty analyses employ probabilistic descriptions of model inputs/parameters to derive probability distributions for, and/or quantity the effect on, model output. For complex process-based models, sensitivity analyses may be relatively simple to perform, but uncertainty analyses that derive probability distributions for model output often require many thousands of simulations, which may not be practical given the computational requirements of complex process-based models. More simple first order uncertainty analyses using, for example, one-factor-at-a-time methods, can be used to quantify the effect of variation in model parameters on model output. These types of analyses, coupled with sensitivity analyses, can help to quantify the effect of uncertainty in input data and model parameters on uncertainty in output variables (Loucks *et al.*, 2005).

Scenario testing

Scenario testing involves using the model to make predictions about the effect of various perturbations or management options on modelled variables. Scenarios must first be conceptualised and then model forcing data adjusted as appropriate, e.g. meteorological forcing data can be adjusted for climate change scenarios, or nutrient concentrations in inflows may be adjusted for nutrient load reduction scenarios. Scenario testing must not be used as an assessment of model performance and can be applied only when the model has been calibrated and validated, i.e., model parameters are generally fixed and must not be adjusted during scenario testing. However, certain conceptual scenarios may require the alteration of one or more model parameters (e.g. a scenario involving sediment management may require the reduction of sediment nutrient release rates to conceptualise the effects of application of a sediment capping agent).

Output from scenario simulations can be directly compared with the calibration/validation period (also referred to as the “base case” or “base scenario”).

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1 Introduction

1.1 Background

The University of Waikato was contracted by Environment Southland to provide a decision support tool comprising of a suite of models. The decision support tool was designed to help assessment of whether various catchment and lagoon management scenarios will meet goals for sustaining the natural values of the Waituna Lagoon ecosystem. One immediate goal was to provide guidance on management options to prevent the decline and potential collapse of *Ruppia* beds, which could have significant adverse impacts on the viability of other organisms in the lagoon and may be hard to reverse once the beds disappear. Therefore the goal of the modelling was to assess suitable catchment and lagoon management techniques to sustain an abundant and stable *Ruppia* population that would support higher levels of the lagoon's food web. The questions asked of the decision support system were diverse, necessitating application of models with inter-connected physical, chemical and biological processes, and including a range of temporal and spatial scales of interest (e.g. from tidal to seasonal and inter-annual).

The different temporal and spatial requirements for the decision support tool, as well as the requirement to include substantial detail of the planktonic and benthic primary producers, necessitated both changes to an existing ecological model (DYRESM-CAEDYM) which is one-dimensional in spatial extent (vertically resolved/horizontally averaged) as well as application of an existing three-dimensional hydrodynamic model (ELCOM). The latter model was used to address specific issues about salinity intrusion into Waituna Lagoon in response to two different opening locations used when water levels begin to result in inundation of farmland surrounding the lagoon. One other model (PCLake) was investigated for the purpose of examining possible alternation (a "regime shift") of dominance by *Ruppia* with planktonic or benthic filamentous algae (i.e., an "alternate state model"). Ultimately it was deemed that this model was unsuitable for application to Waituna Lagoon (see Section 2.2) and modifications of the DYRESM-CAEDYM model were undertaken to provide the regime shift simulation capacity specific for Waituna Lagoon (see Section 2.3.1).

The modelling approach described in this report is specific to the lagoon and not to the catchment. In other words, we set about using the model to assist with prescribing practices to support stable *Ruppia* populations based on phenology of the opening (i.e., timing, duration and location) and nutrient and sediment loads to the lagoon, rather than using the model to specify practices in the catchment that could meet specific loading criteria for the lagoon (see Scanes, 2012, for interim loading criteria for catchment nitrogen and phosphorus inputs from the catchment). A remaining part of the contract involves the use of a model (the "Surface Water Assessment Tool", "SWAT") to develop a more detailed understanding of catchment hydrology, nutrient losses and land use practices. Two other, minor components of the contract were undertaken by the University of Waikato (UW). One was an assessment of the relevance and potential applicability of remote sensing as a tool to assist with monitoring of the lagoon. The assessment is described in this report. The other was to provide advice to Environment Southland on setting up an *in situ* remote monitoring station in Waituna Lagoon to provide continuous readings of relevant variables such as salinity, temperature and water levels, and so that the station could potentially be used as early warning system for rapid changes in variables that might be consistent with a regime shift.

1.2 ICOLLs

Coastal lakes and lagoons occupy approximately 13% of the world's coast. They are similarly prevalent around Australasia. Of 130 coastal water bodies in New South Wales, 90 are considered to be Intermittently Closing and Opening Lakes and Lagoons (ICOLLs). In New Zealand, Schallenberg *et al.* (2003) noted a high density in New Zealand of coastal lakes, lagoons and wetlands (7.4 per 100 km coastline), and indicated that many would be threatened by sea level rise as they are within 1 m elevation of mean seawater levels. Kjerfve (1986) makes a number of interesting observations about why ICOLLs appear to be less studied and their dynamics generally poorly understood compared with their estuarine counterparts which have a permanent connection to the ocean. He suggests that the interconnected factors of shallow depth, lack of shipping channels and low population densities play a significant role in the lack of attention focused on processes occurring in ICOLLs. However, shallow depths and reduced flushing make ICOLLs potentially highly susceptible to eutrophication arising from development activities in their catchments that result in increased nutrient loads (Haines, Tomlinson & Thom 2006; Scanes *et al.* 2007).

ICOLLs in their natural state fill with freshwater until they breach a coastal barrier that then results in rapid loss of freshwater from the lagoon. Prior to the breach seawater ingress to the lagoon is strongly limited but can occur with overtopping on spring tides or infiltration via the barrier bar that separates the lagoon and the coastal ocean. By contrast, upon opening there may be high rates of tidal exchange and substantial seawater ingress into the lagoon. Many ICOLLs around the world where there has been substantial catchment development, including for urban or agricultural purposes, are opened mechanically by people. In New Zealand inundation of farmland is a primary driver for opening ICOLLs and lagoon water levels are generally closely monitored once there is a threat of inundation of surrounding land. In New South Wales (NSW) perceived poor water quality at high water levels has also been a driver for artificial openings of ICOLLs. The NSW Department of Primary Industries has responded to public pressure for opening ICOLLs through education to inform local communities of freshwater biodiversity that may be threatened by prolonged openings, the costs and potential dangers of opening works, and the need to have a management plan to reduce nutrient loads from the catchment (Department of Primary Industries 2012).

Schallenberg *et al.* (2010) compare the effects of openings for two New Zealand ICOLLs, Te Waihora (Lake Ellesmere, Canterbury) and Waituna Lagoon. These ICOLLs present contrasting responses to opening. Te Waihora has limited tidal exchange relative to its lake volume and becomes only moderately brackish (generally considerably less than 50% of seawater salinity in the central part of the lake) whereas Waituna Lagoon is highly flushed and shows strongly contrasting variations in chlorophyll and nutrients immediately before, and soon after, opening. Schallenberg *et al.* (2010) suggest that there may be "unintended consequences" for the water quality and ecology of ICOLLs when the phytoplankton and nutrient dynamics associated with the open and closed states are not well understood. There are obviously complex interactions amongst the opening regime, climate, catchment nutrient loads, and lagoon water quality and ecology. For example, for Waituna Lagoon Robertson and Funnell (2012) point out not only the potential loss of resilience associated with decline of *Ruppia* beds and increases in phytoplankton biomass since 2009, but also indicate the "trade-off" between the salinity and desiccation pressures on macrophytes from opening events versus the potential for these events to flush nutrient-laden freshwater and organically-enriched sediments from the lagoon. The phenology of these opening and closing events may play a critical role in maintaining the resilience of the ecosystem contemporaneously with increasing nutrient loads to the lagoon, as there are indications that the extent of catchment land use in

Waituna Lagoon is at a level that is likely to predispose the lagoon to elevated nutrient levels (Abell *et al.* 2010) and a regime shift to a turbid, phytoplankton-dominated state that may not easily be reversed (Schallenberg & Sorrell 2009; Scanes 2012).

Figure 1: Waituna Lagoon (on the right), separated from the ocean by the barrier at the eastern end

1.3 Model applications

Environmental decision making requires an in-depth understanding of how systems are directly or indirectly affected by anthropogenic activities. In this regard empirical approaches provide an opportunity to develop statistically significant relationships between dependent variables (e.g., phytoplankton biomass) and independent variables (e.g., nutrients) but they can have limitations as relationships are not necessarily causal (i.e., only statistical), and it can be difficult to include interactions amongst the variables. By contrast ecological models are based on attempts to include functional relationships amongst different variables (Schmolke *et al.* 2010). In aquatic sciences, ecological models have long been used to help understand and analyse physical, chemical and/or trophic dynamics (Norberg & DeAngelis 1997) and also for water quality management and forecasting purposes (Friedman *et al.* 1984; Arhonditsis & Brett 2005). Some of the disadvantages of ecological models are that they are deterministic (i.e., each model run provides only one possible solution unless a statistical approach such as Monte Carlo is invoked) and that their complexity can make it difficult to derive cause-effect relationships that managers often seek to assist with implementation of simple environmental management criteria. Bayesian Belief Networks and Artificial Neural Networks are other modelling systems that may invoke expert knowledge, decision support trees and may attempt to weight the relative importance of independent variables towards the model outcome. We deal only with process-based deterministic models in the Waituna Lagoon application described in this report, primarily because they can be used to test the plausibility of conceptual models of lagoon function, to shed light on the importance of various processes contributing to observed system behaviour, and to extrapolate to scenarios that fall outside the range of historical observations.

We chose a one-dimensional, coupled hydrodynamic-ecological model, DYRESM-CAEDYM, developed at the Centre for Water Research, University of Western Australia, as our primary tool for assessing effects of altered hydrology, nutrient and sediment loads, and climate on physical and biogeochemical processes in Waituna Lagoon. The model has previously been applied in New Zealand and overseas for the purpose of assisting with lake management decisions (e.g. Burger, Hamilton & Pilditch 2008; Trolle, Skovgaard & Jeppesen 2008; Gal *et al.* 2009; Özkundakci, Hamilton & Trolle 2011). DYRESM-CAEDYM is the most widely used and cited aquatic ecosystem model documented in the scientific literature (Trolle *et al.* 2011a).

Figure 2: *Ruppia* and macroalgae in shallow water at the eastern end of the lagoon

1.4 Remote sensing

Remote sensing encompasses all methods which use satellite-based imagery to determine particular characteristics (e.g. temperature, chlorophyll *a* concentration) of water bodies. The aim of remote sensing of lakes is to provide truly synoptic monitoring of water quality (Kloiber, Brezonik & Bauer 2002). While remote sensing only allows monitoring of optically active water quality parameters, it has the advantage of greatly increasing the spatial resolution of monitoring (Dekker, Vos & Peters 2002) and, depending on the satellite platform, the temporal coverage of monitoring (Binding *et al.* 2007). Remote sensing of lakes with satellites has the potential to produce a truly synoptic tool with which to monitor water quality variables such as chlorophyll *a*, total suspended sediment, suspended minerals and coloured dissolved organic matter (Mayo, Gitelson & Ben-Avraham 1995; Zhang *et al.* 2002). By contrast, routine monitoring based on analysis of water samples can be laborious and has limited capacity to capture the scales and extent of horizontal heterogeneity in lake water quality (Dekker *et al.* 2002).

Use of Landsat images for inland water assessments has evolved from reconnaissance analysis of water colour as a general indication of trophic state and algal biomass (Wrigley & Horne 1974; Brown, Warwick & Skaggs 1977) to generating reliable predictions of trophic state (Lillesand *et al.* 1983), Secchi depth transparency (Kloiber *et al.* 2002) and chlorophyll *a* at landscape scales (Koponen 2006). The aim of the remote sensing in this project was to provide a preliminary assessment of the potential of Landsat to monitor water quality in Waituna Lagoon.

1.5 Study objectives

The main objective of the present study was to construct and apply process based models in order to assess whether various management scenarios could meet goals for sustaining the natural values of the ecosystem. Specifically, an immediate goal of lagoon management was to prevent the decline and potential collapse of *Ruppia* beds, and subsequently to provide a pathway of management to sustain an abundant and stable *Ruppia* population that supports higher levels of the lagoon's food web.

2 Methods

2.1 Study site – Waituna Lagoon

Waituna Lagoon, and the broader wetland ecosystem within this area (collectively, the Awarua complex), is designated as a wetland ecosystem of international significance, according to criteria of the Ramsar Convention (1971). The designation of Ramsar status for Waituna Lagoon in 1976 was based on the diversity of endemic species, including several threatened species, supported by the swamp, peatland and open water areas of the lagoon. Waituna Lagoon is an outstanding example of a largely unmodified temperate shallow coastal lagoon. Lagoons such as Waituna are referred to as Intermittently Closed and Opened Lake Lagoons (ICOLLs). Many ICOLLs in New Zealand have been adversely affected by nutrient and sediment export from intensive agriculture which characterises many lowland coastal areas of New Zealand. Specific issues for ICOLLs are related to relatively high loadings of nutrients, sediments and microbial contaminant inputs from their catchments, modification to the riparian area often associated with drainage of wetlands and pasture encroachment, and management of the opening-closing regime where this is altered artificially. Fishing, shooting and conservation are often strongly valued in ICOLL environments.

The Waituna Lagoon Ramsar site was extended to include the wider Awarua wetland system in 2008. This extension involved addition of c. 15,400 ha of land, including three nearby estuaries (Toi Toi, Awarua Bay and New River) and Tiwai Peninsula. At a local scale Waituna Lagoon is a taonga for Te Rūnanga o Ngāi Tahu and is highly valued for its diversity of mahinga kai resources. The lagoon holds important historic cultural values for this tribe, including the existence of wahi taonga and wahi tapu sites along its shores. Te Rūnanga o Ngāi Tahu has statutory acknowledgment for Waituna Lagoon under a Deed of Settlement with the Crown (Ngai Tahu Claims Settlement Act, 1998). The Southland Regional Council ("Environment Southland"; ES) has statutory responsibilities for the lagoon water resources under the Resource Management Act (1991). The Department of Conservation manages the wetland on behalf of the Crown in concordance with the 1983 status of the lagoon as a scientific reserve. Southland Fish and Game Council is responsible for fishing and hunting management and season licenses related to these pursuits. The wider community, including farmers, anglers hunters and trampers, value the lagoon for aesthetic reasons and for the wide array of recreational opportunities it provides, and there is also a Waituna Landcare Group whose aim is preservation, restoration and guardianship of the lagoon.

Given the value of the lagoon internationally, to New Zealanders generally, and to iwi and the local community, it is critical that activities within the Waituna Lagoon as a whole are consistent with preserving these values. The lagoon has an area of approximately 1350 ha, with a further 2,200 ha designated as Department of Conservation (DOC) land and managed primarily to sustain the peatland vegetation, which is dominated by manuka (*Leptospermum scoparium*), wire rush (*Empodisma minus*), tangle fern (*Gleichenia dicarpa*), and inaka (*Dracophyllum longifolium*), and provides habitat for endemic fauna such as the nationally-threatened Australasian bittern (*Botaurus poiciloptilus*) and the South Island fernbird (*Bowdleria punctata*). The current peat wetland and the peat soils that underlie much of the rest of the Waituna Lagoon catchment, which were once part of a massive (c. 20,000 ha) peat bog, have an important influence on the composition of water entering the lagoon. Inflows are slightly acidic (i.e., $pH < 7$), and have high concentrations of dissolved organic carbon, some of which includes chromophoric material such as tannins, which strongly absorb light. Thus water in the lagoon generally appears dark (Figure 3) when the lagoon is dominated by freshwater rather than relatively clear seawater.

Waituna Lagoon supports a large number of wading bird species and has good populations of inanga (*Galaxias maculatus*), long finned and short finned eels (*Anguilla dieffenbachii* and *A. australis*). Giant kokopu (*Galaxias argenteus*) are found in the lagoon and many of the tributary watercourses, and introduced sea-run brown trout (*Salmo trutta*) are widespread through the lagoon and its tributaries and constitute a valued fishery (Thompson & Ryder 2003).

Figure 3: Characteristic tannin stained water of Waituna Lagoon

Aside from the lagoon area and the peatland which is managed by DOC, the remaining catchment area of Waituna Lagoon of approximately 19,000 ha consists of indigenous and exotic forest, sheep and beef farming and intensive dairy farming. The proportion of land devoted to different farming activities has changed rapidly in the Waituna Lagoon catchment in recent years, mostly due to conversion of sheep and beef to dairy. Intensive dairy farming has higher rates of nutrient loss than sheep and beef, leading to concerns by management authorities, stakeholders and the community that these changes could compromise *Ruppia* beds in the lagoon, ultimately leading to their collapse and the existence of an undesirable turbid (low-clarity) state characterised by reduced biodiversity. Recent reports have indicated a decline in *Ruppia* cover in the lagoon, commencing around 2007, and presence of nuisance growths of benthic filamentous algae ("macroalgae") that may potentially further shade *Ruppia* beds and are considered to be a precursor for further reductions in *Ruppia* biomass (Stevens & Robertson 2007; Robertson & Stevens 2009; Stevens & Robertson 2010; Sutherland & Taumoepeau 2011). Based on NZ-wide data (Schallenberg & Sorrell 2009; Abell *et al.* 2011) it appears that the percentage of agricultural land in the catchment has for some time exceeded a percentage of the catchment area that could threaten the stability of the clear-water macrophyte (*Ruppia*)-dominated state, let alone intensification of this land. The Waituna Lagoon Technical Group (LTG) proposed interim guidelines in May 2011 that recommended a 75% reduction in current nitrogen loadings and a 50% reduction in current phosphorus loadings to return the lagoon to a state that supports "healthy" *Ruppia* populations (Robertson *et al.*, 2011). The authors recognised that it would likely take some time to reduce nutrient loads to target levels, and so also recommended, as a short term measure, that the lagoon may need to be opened at defined ecological triggers to flush nutrients and phytoplankton from the system. Furthermore, the LTG suggested that the timing of lagoon opening events should be carefully considered as prolonged opening events are likely to adversely affect *Ruppia* populations. These concerns prompted the present project, whose objective was to contribute a rigorous, science-based approach to support objective policy decisions required to manage *Ruppia* beds and support biodiversity in the lagoon.

2.2 PCLake

A PCLake simulation was undertaken for Waituna Lagoon for the period of nine years between 1 January 2002-31 December 2010, the results of which are included in Appendix 1. PCLake was extensively calibrated with parameter values adjusted within ranges found in the literature. Comparison of the calibrated PCLake output with output from DYRESM-CAEDYM indicated that the latter performed better at simulating variables associated with phytoplankton, and dissolved and organic nutrients. Due to the shortcomings of PCLake not being able to reproduce the observed field data adequately, it became apparent that it was unsuitable for modelling the biogeochemical processes in Waituna Lagoon. A more detailed discussion of the issues we identified with this particular model application is included in Appendix 1. Briefly, the zero-dimensional nature of PCLake presented a problem for simulating the effect of changes in water level (i.e., desiccation) on macrophyte beds, a factor which is likely to be highly significant in controlling *Ruppia* dynamics in the lagoon (Johnson & Partridge 1998; Robertson & Funnell 2012). Originally, PCLake was chosen because of its ability to simulate dynamic feedback mechanisms between macrophytes and bed shear stress, and thus sediment resuspension; a mechanism not included in the original version of DYRESM-CAEDYM. For this model application, we developed an algorithm to include a *Ruppia*-specific macrophyte group, including dynamic feedback from the macrophyte biomass to bed shear stress, and thus to resuspension in DYRESM-CAEDYM (Section 2.3.1).

2.3 DYRESM-CAEDYM model description

In this study, the one-dimensional (1D) hydrodynamic model DYRESM (version 3.1.0-03) was coupled with the aquatic ecological model CAEDYM (version 3.1.0-06), both developed at the Centre for Water Research, University of Western Australia, to simulate water quality in Waituna Lagoon. DYRESM resolves the vertical distribution of temperature, salinity, and density, and the vertical mixing processes in lakes and reservoirs. CAEDYM simulates time-varying fluxes that regulate biogeochemical variables (e.g. nutrient species, phytoplankton biomass). The model includes comprehensive process representations for carbon (C), nitrogen (N), phosphorus (P), and dissolved oxygen (DO) cycles, and inorganic suspended solids. Several applications have been made of DYRESM-CAEDYM to different lakes (e.g. Burger *et al.* 2008; Gal *et al.* 2009; Trolle *et al.* 2011b; Özkundakci *et al.* 2011) and detailed descriptions of the model equations can be found in Gal *et al.* (2003) and Romero, Antenucci & Imberger (2004).

The biogeochemical variables in CAEDYM may be configured according to the goals of the model application and availability of data. In this study, three groups of phytoplankton were included in CAEDYM, representing generically small unicells/cyanophytes that are largely unaffected by sedimentation (without N-fixation), cryptophytes that form a "background" population and diatoms (sinking phytoplankton), based on monthly phytoplankton sampling carried out by ES (April-October 2011) that indicated the dominance of these three groups.

The interactions between phytoplankton growth and losses, sediment mineralisation and decomposition of particulate organic matter influence N and P cycling in the model (Figure 4). Fluxes of dissolved inorganic and organic nutrients from the bottom sediments are dependent on temperature and concentrations of nitrate and dissolved oxygen (DO) of the water layer immediately above the sediment surface. Model parameters are calibrated to be specific to each new application but with an extensive parameter library now available from the large number of studies undertaken with DYRESM-CAEDYM. Parameters used in this study are given in Appendix 2.

CAEDYM also has provision for simulation of one macrophyte group and up to four macroalgae groups. A number of macrophyte (e.g. *Ruppia polycarpa, Ruppia megacarpa, Myriophyllum triphyllum*) and macroalgae species (e.g. *Enteromorpha* sp., *Ulva* sp., *Bachelotia antillarum*, *Cladophora* sp.) have been identified in Waituna Lagoon (Schallenberg & Tyrrell 2006; Stevens & Robertson 2007; Robertson & Stevens 2009; Stevens & Robertson 2010). *Ruppia* is recognised as a keystone species in Waituna Lagoon and recent surveys have also indicated increased presence of nuisance macroalgae that may be affecting *Ruppia* growth by growing epiphytically, thus shading the macrophyte species (Robertson, Robertson & Stevens 2011). Data for parameterisation and calibration/validation were not considered adequate to have confidence in simulating the dynamics of individual species, thus one group of macrophytes (*Ruppia*) and one group of macroalgae were simulated. The macrophyte module in CAEDYM was initially configured for an estuarine seagrass species, and so was not entirely suitable for modelling *Ruppia*. Furthermore, there was no feedback between seagrass biomass and shear stress, requiring substantial model development for the current application.

2.3.1 *Ruppia*-specific developments in CAEDYM

The macrophyte algorithm in CAEDYM initially related productivity to changes in light, temperature and salinity (Hipsey, Antenucci & Hamilton 2011). As macrophytes are rooted in the sediment, it is assumed that there is no nutrient limitation. In this study, a self-shading function was added to the macrophyte growth algorithm, and the salinity limitation function was altered to better represent the currently-understood responses of *Ruppia* to salinity. Although *R. polycarpa* and *R. megacarpa* grow in fresh/brackish to hypersaline coastal lakes, lagoons and estuaries in Australia (NSW, Victoria, SA, WA, and Tasmania), suggesting they may be tolerant to a wide range in salinity, studies in Te Waihora (New Zealand) suggest that growth of both *R. megacarpa* and *R. polycarpa* was reduced at salinities above 6-8 (Gerbeaux 1989). *Ruppia* abundance in Waituna Lagoon tends to decline sharply following lagoon opening, which is coincident with high salinities (> 25). However, *Ruppia* beds are mostly located at c. 0.25 to 1 m above mean sea level (masl) elevation in Waituna (and this represents a large proportion of total lagoon area), thus desiccation of the macrophyte beds when the lagoon is opened is also likely to be a very significant factor affecting *Ruppia* dynamics (Johnson & Partridge 1998). DYRESM (the 1D hydrodynamic driver to CAEDYM) was set up to represent the lagoon as a series of vertical layers (of between 0.1 and 0.5 m depth). Desiccation of *Ruppia* is represented by the reduction of *Ruppia* biomass to a minimum value when the water level lowers and depth in a layer decreases to zero. In this study, *R. megacarpa* and *R. polycarpa* are modelled as one group (*Ruppia*) due to limited information on differences between the two species in Waituna Lagoon.

In this study, *Ruppia* growth is modelled as:

$$\partial RV/_{\partial t} = \mu max_{RV} \cdot f(T_{VP}) \cdot f(I_V) \cdot f(RV_V) - kr_{RV} \cdot f(T_{VR}) \cdot f(S_V)$$

where, RV = *Ruppia* biomass (g C m^{-2}), μmax_{RV} = maximum potential growth rate (day^{-1}), $f(T_{VP})$ is a temperature function (applied to production) that assumes that the growth response follows the standard relationship (with θ_T the temperature multiplier) up to a temperature T_{STD}, that maximum productivity occurs at a temperature T_{OPT}, but above T_{OPT} the function decreases to zero at the temperature T_{MAX}. Equations describing the temperature function are available in the CAEDYM manual (Hipsey *et al.* 2011).

$f(I_V)$ is the light limitation function (Webb *et al.*, 1974):

$$f(I_V) = 1 - \exp\left(\frac{-I}{I_k}\right)$$

where, I is the photosynthetically active radiation (µE m^{-2} s^{-1}) and I_k is the irradiance constant.

$f(RV_v)$ is a maximum biomass, i.e., a self-shading function, (Haese & Pronk 2011):

$$f(RV_v) = 1 - \left(\frac{RV}{RV_{max}}\right)$$

where, RV = *Ruppia* biomass and RV_{max} is the maximum *Ruppia* biomass (g C m^{-2}).

kr_{RV} is the respiration/mortality rate (day^{-1}) and $f(T_{VR})$ is a temperature function (applied to respiration) that assumes the function follows the standard relationship (with θ_B the temperature multiplier).

$f(S_V)$ is the salinity limitation function for *Ruppia* (identical to the salinity limitation function for freshwater phytoplankton in CAEDYM (Hipsey *et al.* 2011)):

$$f(S_V) = 1 \qquad \text{for } S < S_{opt}$$

$$f(S_V) = \frac{(\beta-1)\cdot S^2}{(S_{max}-S_{opt})^2} - \frac{2(\beta-1)\cdot S_{opt}\cdot S}{(S_{max}-S_{opt})^2} + \frac{(\beta-1)\cdot S_{opt}{}^2}{(S_{max}-S_{opt})^2} + 1 \qquad \text{for } S > S_{opt}$$

where, S_{opt} is the salinity above which salinity limitation occurs, S_{max} is the maximum salinity, and β is the salinity limitation value at S_{max}.

Macrophytes are known to influence resuspension of inorganic and organic sediments (Hamilton & Mitchell 1996), but quantitative relationships between macrophyte coverage and resuspension or shear stress are scarce, and likely to be species-specific. The model PCLake assumes a linear decline in sediment resuspension with increasing macrophyte biomass (Janse 2005). Research in New Zealand lakes indicates, however, that the relationship between macrophyte biomass and shear stress may be logarithmic (Hamilton & Mitchell 1996). We applied a limitation function directly to bottom shear stress, $f(SS_{RV})$, which takes the form:

$$f(SS_{RV}) = 10^{(-kSS\cdot RV)}$$

where, kSS is a constant (~ 0.01 m^2 g^{-1} C) and RV is the *Ruppia* biomass (g C m^{-2}).

The interactions between the primary producer groups (i.e. phytoplankton, macroalgae and *Ruppia*) and N and P cycling for Waituna Lagoon are conceptualised in Figure 4.

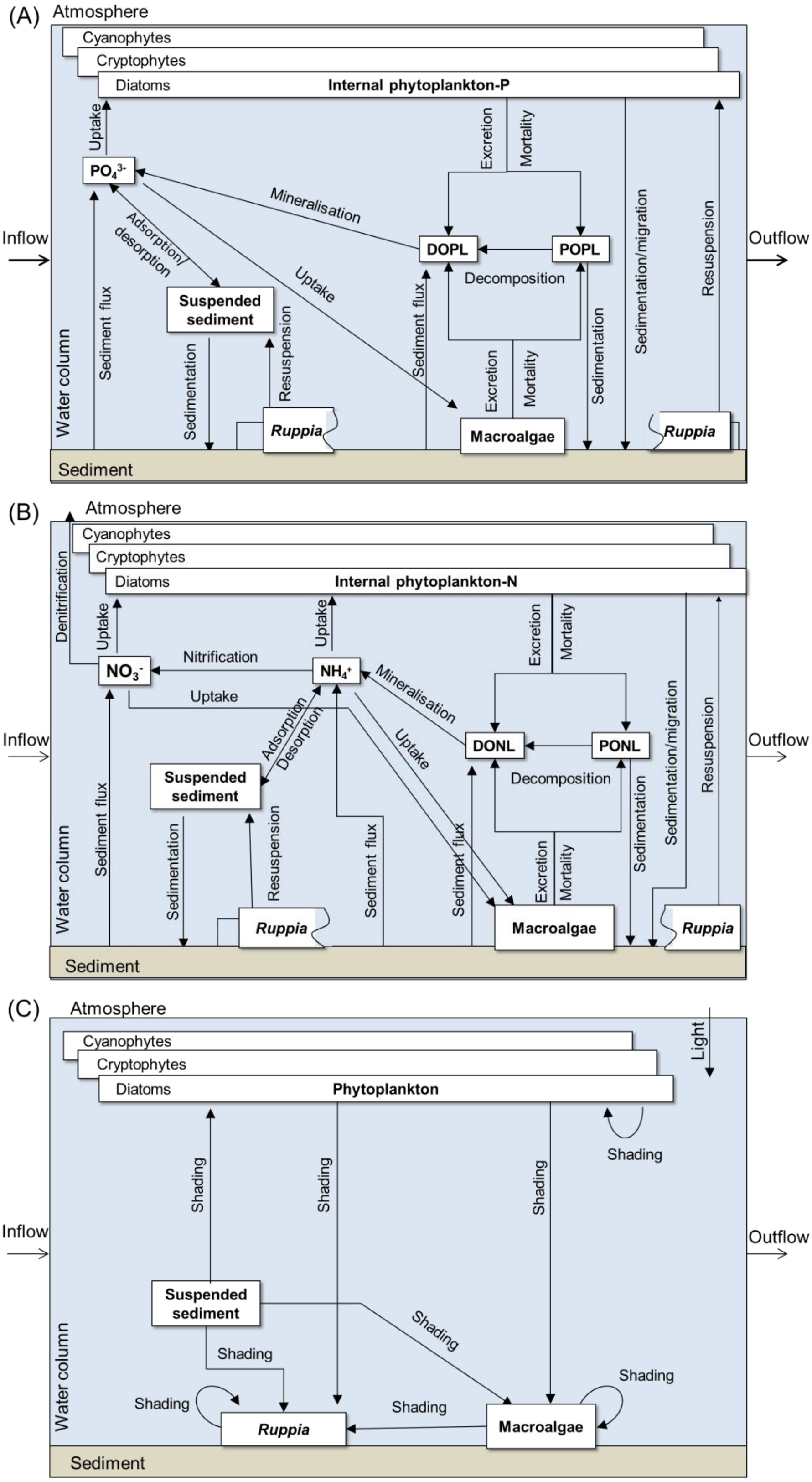


Figure 4: Conceptual model of the (A) phosphorus and (B) nitrogen cycles represented in DYRESM-CAEDYM for the present study, and (C) the light interaction of simulated biological components. POPL, PONL, DOPL and DONL represent particulate labile organic phosphorus and nitrogen, and dissolved labile organic phosphorus and nitrogen, respectively. For clarity, salinity limitation for *Ruppia* and phytoplankton were omitted.

2.4 DYRESM-CAEDYM model inputs

In this study, DYRESM-CAEDYM was run at hourly time steps between October 2001 and October 2011, with daily averaged input data and daily output data at midday. To overcome uncertainty in the initialisation data the calibration and validation periods were run after a one-year "warm up" period that looped the first year of input data.

2.4.1 Meteorology

Meteorological data required for the simulation period were obtained from the National Climate Data Base (http://cliflow.niwa.co.nz) for the Tiwai Point climate station (-46.587 ºS 168.376 ºE) located c. 15 km west of Waituna Lagoon (Figure 5). The data included air temperature (ºC), shortwave radiation (W m^{-2}), cloud cover (fraction of whole sky), vapour pressure (hPa), wind speed (m s^{-1}) and rainfall (m) (Figure 6). Data are collected at Tiwai Point at hourly intervals, and for the purposes of the model input were standardised to daily average values except for rainfall, which was provided as a daily total value. Daily values for theoretical clear sky and full cloud-cover shortwave radiation (W m^{-2}) were estimated by fitting seasonal sinusoidal curves to the maximum and minimum observed daily shortwave radiation values across the entire simulation period. Subsequently, average daily cloud cover was estimated by calculating the fractional difference between observed total daily shortwave radiation and the estimated theoretical daily maximum and minimum. Occasional values below zero (clear sky) or above one (full cloud cover) were set as 0 and 1, respectively.

2.4.2 Water balance

Surface inflow discharges to the lagoon were obtained from ES. Flows for four streams of the major Waituna Lagoon sub-catchments were included (i.e. Waituna Creek, Moffat Creek, Carren Creek and Carren Creek tributary), along with two other inflows representing the sum of all minor surface flows from around the lagoon (and defined as having catchment areas similar to the Moffat Creek catchment, and areas similar to Carren Creek tributary catchment, respectively). Hourly flow data for Waituna Creek was provided for most of the simulation period (with data gaps filled using a gauging relationship between Waituna and Waihopai catchments) and flow for other inputs based on concurrent gaugings (Chris Jenkins, pers. comm.). Groundwater inputs directly into the lagoon were estimated based on an annual catchment water balance and groundwater seepage estimates provided by ES (Rissmann *et al.*, 2012). Annual average water loads for each of the inflows are given in Table 1. Change in lake storage (ΔS) was calculated from water level recorder data provided by ES (from the Waghorn's Rd gauge), multiplied by the water level-dependent lake area derived from hypsographic curves. Hypsographic curves were derived from bathymetry and LIDAR data provided by ES (Figure 7).

Table 1: Total annual flows and nutrient loads for phosphorus (PO_4-P and organic P) and nitrogen (NH_4-N, NO_3-N and organic N) for Waituna Lagoon inflows (derived from measured data) and outflow (derived from model (DYRESM-CAEDYM) output), averaged over the calibration/validation period (2001–2011).

Inflow	**Flow (m^3 yr^{-1})**	**PO_4-P (t yr^{-1})**	**Organic P (t yr^{-1})**	**NH_4-N (t yr^{-1})**	**NO_3-N (t yr^{-1})**	**Organic N (t yr^{-1})**
Waituna Creek	50,887,511	1.09	3.44	4.79	99.33	40.94
Moffat Creek	10,309,810	0.71	0.88	0.68	6.49	10.97
Carren Creek	12,838,949	0.56	1.33	1.41	8.92	11.08
Carren Creek tributary	4,076,090	0.17	0.09	0.10	0.19	2.53
Other surface 1*	18,351,461	1.27	1.57	1.20	11.55	19.53
Other surface 2*	15,354,630	0.64	0.32	0.36	0.70	9.52
Groundwater	43,822,337	0.31	2.02	0.57	10.52	18.27
Total freshwater inflows	*155,640,787*	*4.75*	*9.64*	*9.11*	*137.70*	*112.85*
Tidal inflow	352,060,290	3.52	3.52	1.76	10.56	86.25
Total inflows	*507,701,077*	*8.27*	*13.16*	*10.87*	*148.26*	*199.10*
Outflow	518,221,964	5.14	4.04	8.82	135.43	61.26
Difference between total inflows and outflow	*-10,520,887*	*3.13*	*9.12*	*2.04*	*12.83*	*137.84*
Nutrient retention (recycling and burial) as a proportion of total inflow	*NA*	*0.38*	*0.69*	*0.19*	*0.09*	*0.69*

* Refers to areas not included in other surface inflows that are similar to Moffat Creeks, and Carren Creek tributary, respectively.

N.B. Tidal inflows and outflow are dependent on lagoon opening regime. These data are for 2001–2011 only.

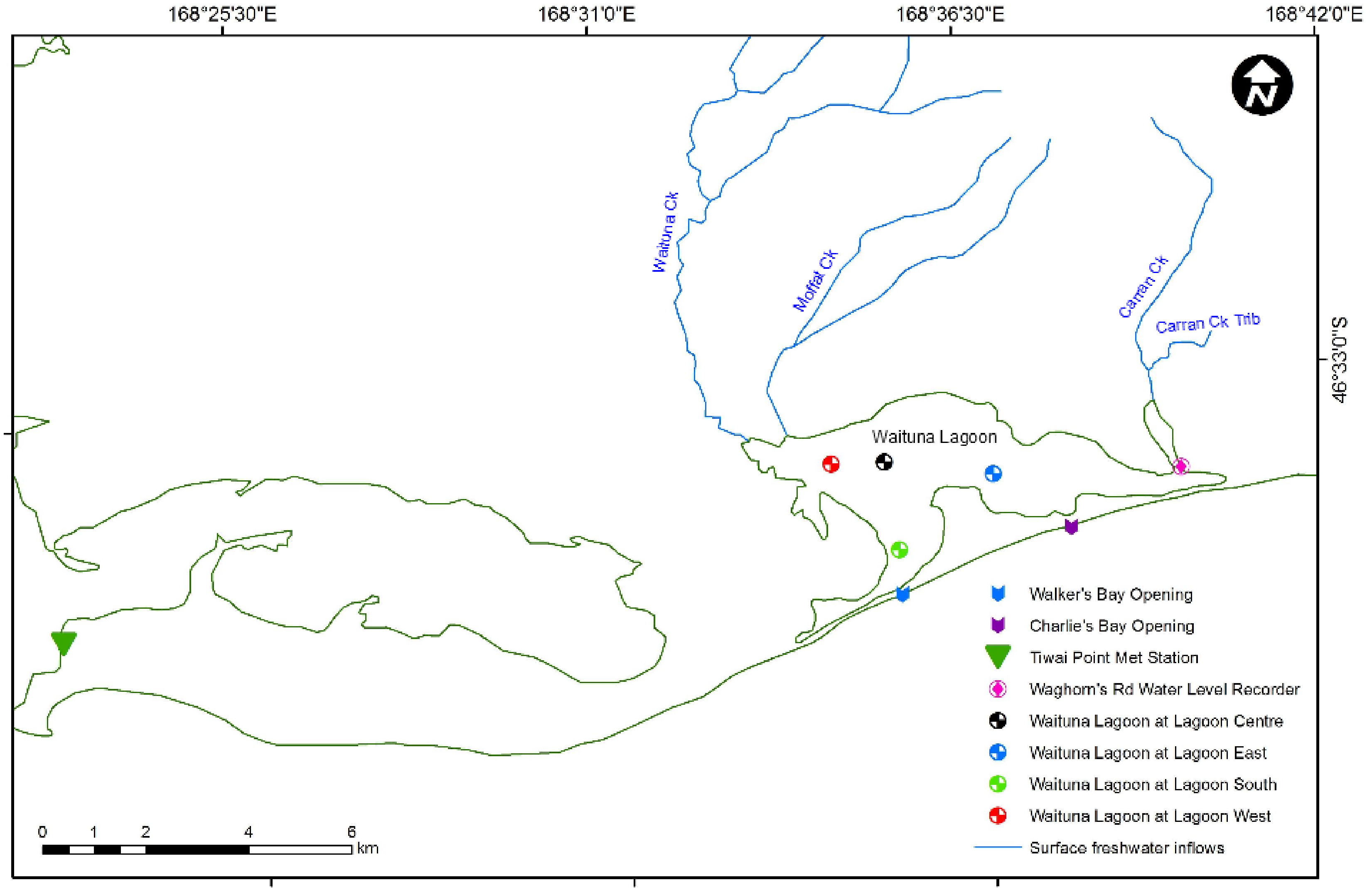


Figure 5: Location of sampling and opening sites, water level recorder, and main freshwater inflows in Waituna Lagoon, and Tiwai Point meteorological station

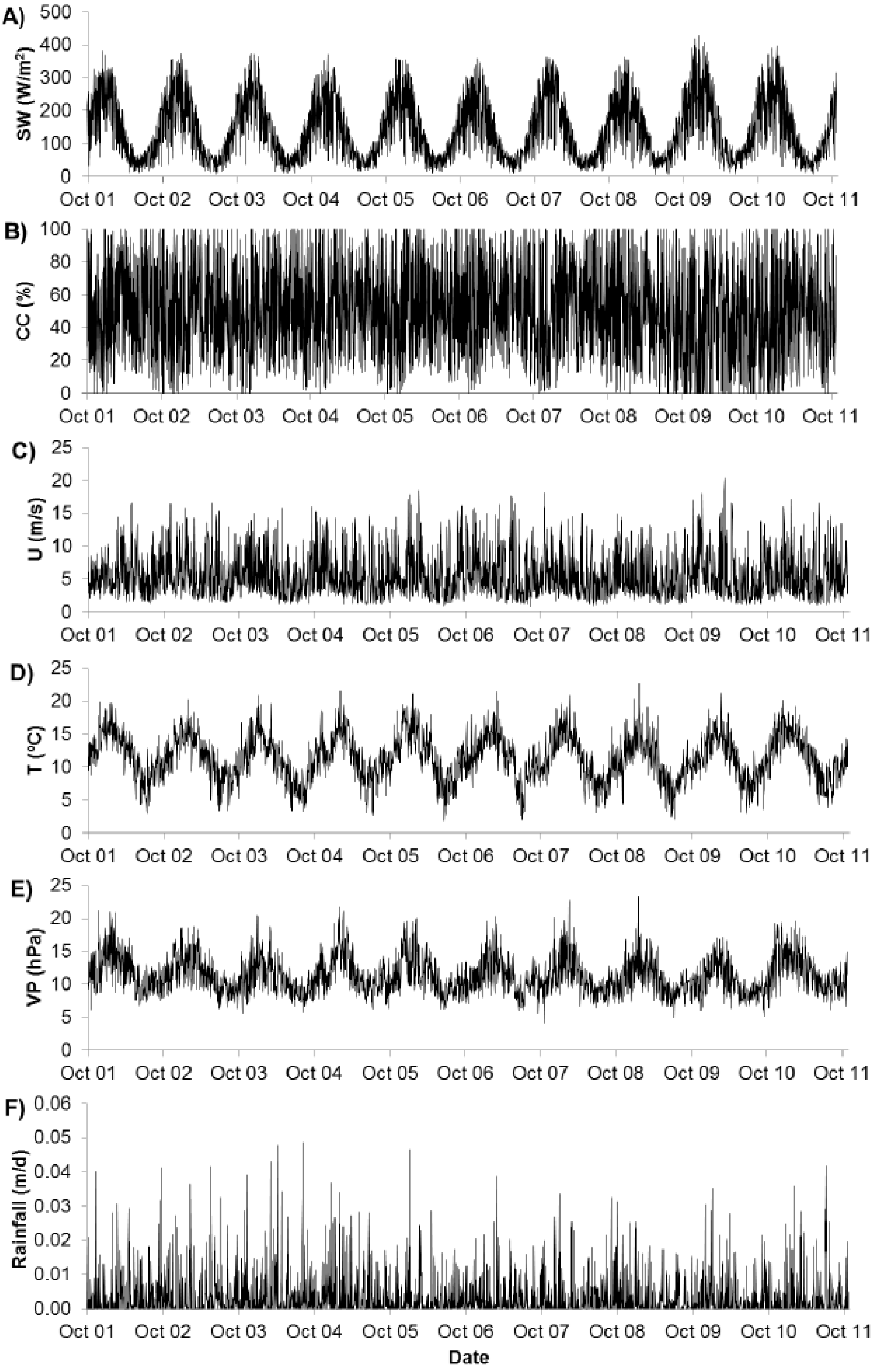


Figure 6: Meteorological data used as input to the DYRESM model (October 2001–October 2011). A) Short wave radiation (SW; W/m^2), B) cloud cover (CC; %), C) air temperature (T; ºC), D) vapour pressure (VP; hPa), E) wind speed (U; m/s), and F) rainfall (m/day). Data were obtained from the Tiwai Point climate station.

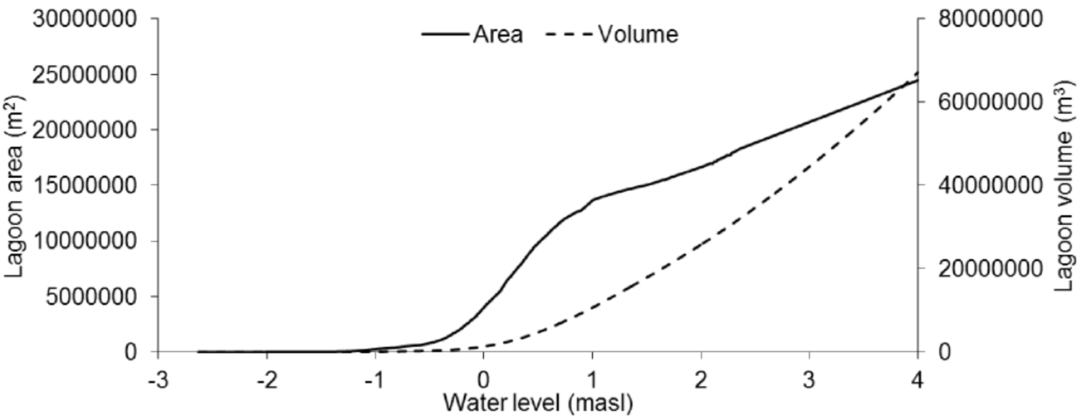


Figure 7: Waituna Lagoon hypsograph

Evaporation from the lake was calculated as a function of wind speed and air vapour pressure from the daily average evaporative heat flux (Fischer *et al.*, 1979; Eqn. 6.20 in Imerito, 2007) using meteorological input data and water temperature:

$$Q_{lh} = minimum\left(0, \frac{0.622}{P} C_L \rho_A L_E U_a (e_A - e_S(T_S))\Delta t\right)$$

where,
Q_{lh} is the evaporative heat flux in J m^{-2} s^{-1},
P is atmospheric pressure in hPa,
C_L is the latent heat transfer coefficient for wind speed at a height of 10 m (1.3×10^{-3}),
ρ_A is the density of air in kg m^{-3},
L_E is the latent heat evaporation of water (2.453×10^6) in J kg^{-1},
U_a is the wind speed at 10 m height above ground level in m s^{-1},
$e_S(T_S)$ is the saturation vapour pressure at the water surface temperature in hPa,
e_A is the vapour pressure of the air in hPa.

The condition that $Q_{lh} < 0$ assumes that condensation does not occur.

For the purposes of determination of water evaporated from the lagoon surface, a daily lagoon water temperature was estimated as for surface inflows (see Section 2.4.4) using methods described in Mohseni *et al.* (1998), from daily air temperature and available lagoon *in situ* temperature measurements provided by ES. The saturated vapour pressure $e_S(T_S)$ was calculated via the Magnus-Tetens formula (TVA, 1972; Eqn. 4.1 in Imerito, 2007):

$$e_S(T_S) = exp\left(2.3026\left(\frac{7.5T_S}{T_S + 237.3} + 0.7858\right)\right)$$

where, T_s is the water surface temperature in ºC.

The change in mass in the surface layer (layer N) due to latent heat flux is calculated as

$$\Delta M_N^{lh} = \frac{-Q_{lh} A_N}{L_V}$$

where, ΔM_N^{lh} is the change in mass in kg s^{-1}, A_N is the surface area of the lagoon in m², and L_V is the latent heat of vaporisation for water (2.258 x 10^6 J kg^{-1}).

The result of this calculation was multiplied by 86.4 to give daily evaporation (E_L) in m^3 d^{-1}.

Outflow may occur as barrier seepage when the lagoon is closed and as outflow through the opening when the lagoon is open. In the absence of measurements for the outflow, daily values for the outflow volume were calculated as a residual term of a complete water balance for the simulation period:

$$Outflow = \sum(surface\ inflows) + groundwater + rainfall - evaporation - \Delta S$$

where, ΔS is change in storage in m^3 d^{-1}.

Negative values, which represented < 2% of total outflows, were averaged and added to surface inflows. The derived outflow was used for the DYRESM simulation over the period 2001–2011; the lagoon level output compared to ES water level recorder data (Figure 8) was closely matched (Pearson's R = 0.97, root mean square error = 0.14 m).

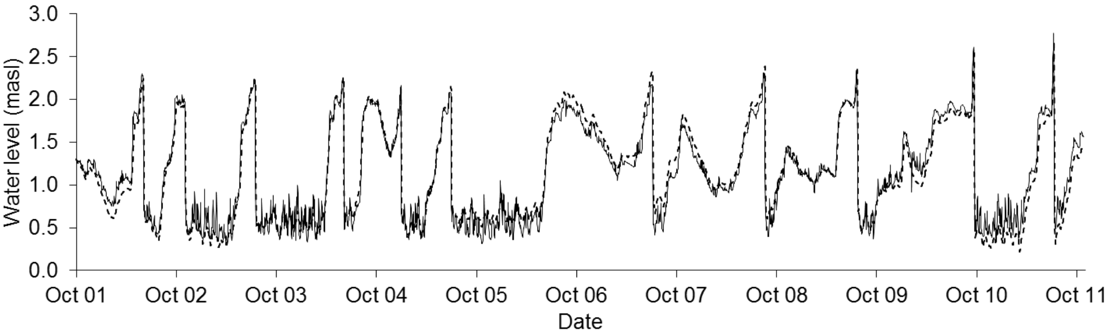


Figure 8: Simulated Waituna Lagoon water level (dashed line), and measured water level (solid line)

There were small discrepancies between the simulated and observed water levels which may be partly attributable to differences between the estimated surface lake water temperature used to derive evaporation for the water balance, and the surface water temperature simulated within DYRESM. Also, our estimates of lagoon storage may be affected by placement of the water level recorder in the far eastern end of the lagoon (at Waghorn's Rd). The prevailing wind direction is westerly, which can cause wind setup (a localised raising of the lagoon level) at the Waghorn's Rd recorder of up to 30 cm (Chris Jenkins, pers. comm.). Moreover, the recorder is c. 7 km east of the Walkers Bay opening location and the tidal response at Waghorn's Rd is delayed by c. 3 to 5 hr compared with the timing of tidally-driven water level variations at the opening.

2.4.3 Tidal inflows/outflows

Tidal inflows and outflows were included as input in the model for periods when the lagoon was open, with lagoon opening and closing dates provided by ES (Table 2). The lagoon was closed at the beginning of the model run (October 2001) and opened ten times at Walker's Bay and once at Charlie's Bay over the simulation period (October 2001–October 2011). In the absence of other data, (i.e., water levels or flows recorded near the lagoon opening), tidal flows were estimated from tidal prism data supplied by ES, and tide heights for Bluff obtained from the NIWA tide forecaster tool (http://www.niwa.co.nz/services/online-services/tide-forecaster). Input data (in m^3 day^{-1}) were derived assuming that tides were semi-diurnal and that tidal inflow was equal to outflow each day.

Table 2: Waituna Lagoon opening and closing dates (Oct 2001-Oct 2011)

Opening location	Date opened	Date closed	Water level*	Days open
Walker's Bay	10/06/2002	8/08/2002	2.30	59
Walker's Bay	9/11/2002	4/05/2003	2.00	176
Walker's Bay	24/07/2003	1/04/2004	2.20	252
Walker's Bay	10/06/2004	15/07/2004	2.20	35
Walker's Bay	5/01/2005	2/04/2005	2.20	87
Walker's Bay	7/07/2005	2/06/2006	2.00	330
Walker's Bay	12/07/2007	21/08/2007	2.16	39
Walker's Bay	25/08/2008	5/10/2008	2.30	41
Walker's Bay	29/07/2009	4/10/2009	2.33	68
Walker's Bay	27/09/2010	31/03/2011	2.60	185
Charlie's Bay	15/07/2011	18/08/2011	2.80	34

* Water level at Waghorn's Rd recorder when opened (masl)

2.4.4 Freshwater inflow parameterisation

Temperature

Surface inflow temperatures for all major inflows were estimated using the method described in Mohseni, Stefan & Erickson (1998):

$$T_s = \frac{\alpha}{1 + e^{\gamma(\beta - T_a)}}$$

where,
T_s is the estimated stream temperature,
T_a is the measured air temperature,
α is the coefficient for the estimated maximum stream temperature,
γ is a measure of the steepest slope of the function,
β represents the air temperature at the inflection point.

Quality of fit was defined by the difference between modelled water temperature and available (typically, monthly) *in situ* ES measurements for each stream. Model parameters were adjusted in order to minimise the root-mean-square error (RMSE) and maximise the Pearson correlation co-efficient (R), using Microsoft Excel Solver.

Dissolved oxygen

Dissolved oxygen concentrations of all inflows were estimated as a function of water temperature (Mortimer (1981) based on data from Benson & Krause (1980)):

$$DO = exp(7.71 - 1.31ln(T + 45.93))$$

where, DO is dissolved oxygen at saturation in mg L^{-1} and T is water temperature in ºC.

Nutrients

Daily nitrate, ammonium and phosphate concentrations for all major inflows were derived by linear interpolation between monthly samples from ES stream monitoring data. This method has been used in other model applications (e.g. Burger *et al.* 2008; Trolle *et al.* 2011b; Özkundakci *et al.* 2011) but potentially underestimates the effect of storm events that may not be captured by routine monitoring. The ES stream monitoring data did include higher resolution coverage of one storm event in May 2011 in which measurements were taken at c. 2 hourly intervals over 5 days (and which was included as model input), but was otherwise comprised of monthly measurements over the 2001 – 2011 simulation period. In the absence of field data on nutrient speciation of labile organic nitrogen and phosphorus concentrations (ONL and OPL, respectively), these species were calculated from ES monthly stream nutrient measurements of total nutrient concentrations, and were evenly divided into dissolved (D) and particulate (P) fractions using the equations:

DONL or PONL = (TN – NH_4-N – NO_3-N) / 2

DOPL or POPL = (TP – PO_4-P) / 2

As in previous DYRESM-CAEDYM applications labile dissolved and particulate organic carbon (DOCL and POCL) concentrations were calculated using inflow labile organic nitrogen concentrations and a “Redfield” molar ratio of 106:16 for C:N (Burger *et al.* 2008; Trolle *et al.* 2011b; Özkundakci *et al.* 2011), where:

DOCL or POCL = (DONL or PONL*106*M(C)) / (16*M(N))

where, M(C) is the molar mass of carbon and M(N) is the molar mass of nitrogen.

High dissolved organic carbon concentrations in Waituna Lagoon inflows and in the lagoon itself are evident from recent sampling by ES (i.e., in April 2012 DOC concentrations in Waituna, Moffat and Carren Creeks ranged from 10.9 to 24 mg L^{-1} and in the lagoon from 12 to 13.5 mg L^{-1}), although DOC was not included in monitoring prior to these samples. To account for what may be an important influence on Waituna Lagoon water quality (i.e., likely to be an important contributor to the colour of the water in the lagoon) we assigned an average value for refractory DOC (DOCR) to all inflows, based on the April 2012 sample concentrations, and which included a component of light attenuation by DOCR.

Atmospheric deposition of N and P directly to the lagoon was not explicitly accounted for in the model. Typical rates for atmospheric deposition in the Southern Hemisphere are 0.35 t N km^{-2} yr^{-1} and 0.017 t P km^{-2} yr^{-1} (Hamilton 2005). Thus, assuming an average lagoon surface area of 13 km^2, atmospheric deposition of N and P is likely to represent just 2-3% of dissolved nutrient loads relative to surface and groundwater inflows (Table 1).

Inorganic suspended solids
As for nutrients, daily values for total suspended solids (TSS) concentrations for all major inflows were derived by linear interpolation between (typically) monthly samples from ES stream monitoring data.

2.4.5 Tidal inflow parameterisation

Water temperature and dissolved oxygen concentrations for the tidal inflow were estimated as for freshwater inflows using available *in situ* data provided by ES (Oreti Beach bathing beach surveys and Riverton Rocks consent monitoring). *In situ* data for nutrients, suspended solids and chlorophyll *a* were scarce (e.g. Riverton Rocks monitoring was six-monthly for samples for NH_4-N, TN and TSS only) preventing estimation of daily values by linear interpolation. Therefore, for PO_4-P, TP, NH_4-N, NO_3-N, TN, TSS an average value for the tidal inflow was estimated based on the Riverton Rocks consent monitoring and samples taken in

Fouveaux Strait during oceanographic research cruises (Bradford, Cranfield & Michael 1991; Vincent *et al.* 1991). Chlorophyll *a* concentrations in inshore waters off the Southland coast tend to show a weak annual cycle, ranging from c. 0.2 µg L^{-1} in winter to 2-3 µg L^{-1} in late summer (Bradford *et al.* 1991; Vincent *et al.* 1991; Murphy & Pinkerton 2001), which was assigned to the diatom group in the tidal inflow for this study.

2.4.6 Groundwater parameterisation

Groundwater is likely to discharge directly into the lagoon around the northern boundaries, particularly around Moffat Creek (Rissmann *et al.*, 2012). The unconfined aquifer (located in Quaternary gravels) is most likely to influence waters discharging into the lagoon, and water quality data for this aquifer was provided by ES (from bore logs and sampling undertaken in March 2011). This information indicated higher nitrate than ammonium concentrations (c. 0.17 to 0.61 µg L^{-1} NO_3-N *cf.* 0.01 to 0.12 µg L^{-1} NH_4-N), as would be expected for water that was not anoxic. An average value for all water quality parameters was assumed based on these data.

Preliminary simulations revealed the model to be consistently under-predicting salinity following lagoon closing events. Field data suggested that following closure salinity often declined quite gradually, taking on average c. 70 days to reach fresh/brackish conditions (< 4). We hypothesised that groundwater may be a source of saline water to the lagoon at these times, and assigned an exponentially declining salinity (from 22 on the day of closing to 4, 14 days after) to the groundwater inflow. This resulted in a better match between modelled and measured salinity following lagoon closure, e.g. Figure 9.

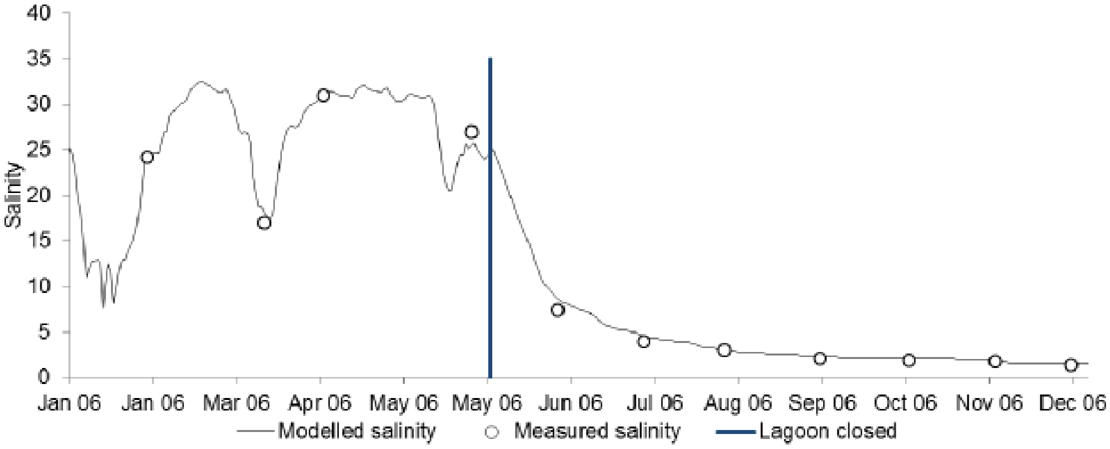


Figure 9: Modelled (DYRESM-CAEDYM) and measured salinity at the Waituna Lagoon sampling station for 2006, after assigning exponentially declining salinity to groundwater inflow following lagoon closure (see text).

2.5 DYRESM-CAEDYM calibration and validation

DYRESM-CAEDYM was calibrated against field data (monthly surface samples collected in the centre of the lagoon by ES) over the six-year period between October 2001 and October 2007 for variables of temperature, salinity, DO, chlorophyll *a*, PO_4-P, TP, NH_4-N, NO_3-N, TN, TSS at the water surface (0 m). The three simulated phytoplankton groups (i.e. cyanophytes, cryptophytes and diatoms) collectively contributed to a total simulated chlorophyll *a* concentration, which was calibrated against measured surface chlorophyll *a*. Model parameters were adjusted manually using a trial and error approach with values set to within literature ranges (e.g., Schladow & Hamilton 1997; Trolle *et al.* 2011b). The model error was represented by a series of model performance statistics, including the root-mean-square-error (RMSE), Pearson correlation coefficient (R), mean absolute error (MAE), mean signed difference (MSD) and comparison of the means of both the observations ($\mathrm{Mean_{obs}}$) and model output ($\mathrm{Mean_{mod}}$).

The RMSE is a frequently used measure of the difference between values predicted by a model and the values actually observed from the environment that is being modelled. These individual differences are also called residuals, and the RMSE serves to aggregate them into a single measure of predictive power. The RMSE of a model prediction with respect to the estimated variable X_{model} is defined as the square root of the mean squared error:

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(y_i - x_i)^2}{n}}$$

where, y_i is observed values and x_i is modelled values at time/place *i*. The calculated RMSE values have units, and RMSE for phosphorus concentrations, for example, cannot for this reason be compared directly to RMSE values for chlorophyll *a* concentrations. However, RMSE values can be used to distinguish model performance or a variable in a calibration period with that of a validation period, as well as to compare the individual model performance to that of other predictive models.

Correlation, often measured with a correlation coefficient, indicates the strength and direction of a linear relationship between two variables (for example model output and observed values). A number of different coefficients are used for different situations. The best known, which was used in this study, is the Pearson product-moment correlation coefficient (also called Pearson correlation coefficient or the sample correlation coefficient), which is obtained by dividing the covariance of the two variables by the product of their standard deviations. For a series of n observations and n model values, the Pearson product-moment correlation coefficient can be used to estimate the correlation between model and observations:

$$R = \frac{\sum_{i=i}^{n}(x_i - \bar{x}) \times (y_i - \bar{y})}{\sqrt{\sum_{i=i}^{n}(x_i - \bar{x})^2 \times \sum_{i=1}^{n}(y_i - \bar{y})^2}}$$

The correlation is +1 in the case of a perfect increasing linear relationship, and -1 in case of a perfect decreasing linear relationship, and the values in between indicate the closeness of fit to a linear relationship between, for example, model and observations. In modelling, a correlation coefficient between simulations and observations of +1 may be ideal, whereas 0 means the there is no linear relationship amongst variables, and -1 represents the poorest possible model fit. However, the correlation between model output and field observations can approach 1 whether or not there is a consistent offset between the two.

The MAE is a measure of average error magnitude, which derives from the unaltered magnitude (absolute values) of each difference:

$$MAE = \frac{1}{n}\sum_{i=1}^{n}|x_i - y_i|$$

The MSD quantifies if the model consistently over and/or under predicts:

$$MSD = \frac{1}{n}\sum_{i=1}^{n} x_i - y_i$$

For each output variable the model statistics were quantified after each simulation for which model parameter values were adjusted. Calibration continued until there was negligible improvement in model statistics with repeated model simulations. Model statistics were also compared to modelling studies in the literature to assess an acceptable model error for prediction purposes. The final model parameters from the calibration were then fixed for model validation over the four-year period October 2007 – October 2011.

There was limited data available on macrophytes and macroalgae for extensive calibration and validation of model output. Four macrophyte surveys were carried out during the 2001 to 2011 model calibration/validation period (in February/March of 2007, 2009, 2010 and 2011). The methodology was not consistent between the 2007 survey and the later surveys, making data comparison difficult, and surveys were conducted for the Department of Conservation (DOC) by Wriggle Coastal Management Ltd in 2007, 2008 and 2009, and by NIWA in 2011. However, the reports do indicate that *Ruppia* coverage in the lagoon declined, whereas macroalgae coverage increased, from 2007 onwards (Stevens & Robertson 2007; Robertson & Stevens 2009; Stevens & Robertson 2010; Sutherland & Taumoepeau 2011). The decline in *Ruppia* coverage from 2009 to 2011 was attributed to increased duration of lagoon opening, leading to desiccation and salinity stress (Robertson & Funnell 2012). Quantitative data on macrophyte percent cover was gathered in all surveys, but no consistent, quantitative data on macroalgae were available for model calibration. An overall macrophyte (*Ruppia*) percent cover estimate for the lagoon was provided by Wriggle Ltd (for all four years) and by DOC (for 2009, 2010 and 2011) to compare with model output (Leigh Stevens (Wriggle), pers. comm. and Emily Funnell (DOC) pers. comm.). *Ruppia* biomass has not been measured in Waituna Lagoon, and available data on the relationship between percent cover and biomass are variable. For example, in Wilson Inlet, in the south of Western Australia, *R. megacarpa* biomass ranged from c. 5 to 70 g C m^{-2}, and percent cover from c. 20 to 90% (Carruthers, Walker & Kendrick 1999). In a model of the same system the maximum *Ruppia* biomass was assumed to be c. 180 g C m^{-2} (Haese & Pronk 2011). In the absence of any Waituna-specific data, percent cover was converted to biomass (in g carbon m^{-2}), based on a linear relationship with 100% cover equal to 100 g C m^{-2} (and 0% cover equal to 0 g C m^{-2}). Model error in simulating macrophytes and macroalgae was not assessed with model statistics, due to the small number of field data points and uncertainty around their derivation; rather the simulated variables were visually compared with the field data and the trends identified from the macrophyte monitoring reports.

2.6 DYRESM-CAEDYM sensitivity analysis

2.6.1 Model complexity

The influence of model complexity on simulation output was examined by gradually reducing the conceptual complexity in consecutive simulations. The macroalgae variable was first removed from the model configuration, followed by *Ruppia* and then the sediment resuspension process. Output for nutrients (dissolved species and total nitrogen and phosphorus), chlorophyll *a*, total suspended sediments and *Ruppia* (for the simulation that removed macroalgae only from model configuration) was compared between the simulations and the more complex calibrated DYRESM-CAEDYM model. Furthermore, the simulation for which *Ruppia* was removed from the model configuration may provide some insight into the effect of a total loss of this species on phytoplankton, suspended sediments, and nutrient concentrations in the lagoon.

2.6.2 Sensitivity to input data

Models are seldom constructed using ideal input data. The sensitivity of model output to uncertainty in input data was explored by adjusting various input data within what were considered to be reasonable bounds (e.g. ± 10%). Meteorological input data for the model were obtained from a climate station located some distance (c. 15 km) west of the lagoon, so the sensitivity of the model to these input data was assessed by altering wind speed (by ± 10%) and air temperature (by ± 0.5 ºC) in successive model simulations. Inflow nutrient and sediment concentrations were measured (typically) monthly in four major surface inflows (but not in the tidal inflow when the lagoon was open), so the sensitivity of the model to these input data was assessed by altering nutrient and sediment concentrations in all inflows (by ± 10%) in successive model simulations.

2.6.3 Sensitivity to key parameters: Uncertainty analysis

For the purpose of this study, a local sensitivity analysis was conducted on selected model parameters using a one-factor-at-a-time method (e.g. Morris 1991). This method was designed to quantify the effect of the variation of a given parameter on the model output while other parameters were kept at their calibrated values for the simulation period. This approach is computationally very efficient compared to more demanding techniques for sensitivity analysis such as Monte Carlo methods.

Parameter selection for the sensitivity analysis was based on considerations of Schladow & Hamilton (1997), Makler-Pick *et al.* (2011) and the experience gained throughout the calibration efforts of the current model application, which encompassed some 500 model runs. We selected 23 parameters for the local sensitivity analysis (Appendix 3). Schladow & Hamilton (1997) carried out a sensitivity analysis on an early version of DYRESM-CAEDYM for Prospect Reservoir, Australia. They found that the sensitivity of model output to changes in calibration parameter values with reference to chlorophyll *a* concentrations was mostly related to parameters that directly alter phytoplankton growth rates or indirectly affect growth rates through their ability to take up or utilise nutrients. Similarly, the sensitivity analysis carried out for Lake Kinneret, Israel, by Makler-Pick *et al.* (2011) on a later version of DYRESM-CAEDYM suggested that some of the most sensitive parameters were those affecting phytoplankton growth rates, but also some potentially site specific sensitivities related to particulate organic material.

The 23 model parameters were changed by ± 10% of their calibrated values. This range was thought to be comparable with values found in the literature and for Waituna Lagoon. The sensitivity of the model output to changed values of the model parameters was assessed and quantified using two sensitivity indicators. The relative change (RC) of model output variables for any given change in model parameter values was calculated as:

$$RC = \frac{1}{n}\sum_{i=1}^{n}\left(\frac{y_{2_i} - y_{1_i}}{y_{1_i}}\right) \times 100\%$$

where, y_1 is the value of the model output variable on day *i* for the baseline simulation using calibration parameter values and y_2 is the value of the model output variable after changing the value of the selected model parameter. A sensitivity ratio (SR), equal to the average percentage change in a model output variable compared to the baseline mode output for the respective variable divided by the percentage change in model parameters was calculated as:

$$SR = \frac{\frac{1}{n}\sum_{i=1}^{n}\left(\frac{y_{2_i} - y_{1_i}}{y_{1_i}}\right) \times 100\%}{\left(\frac{x_{2_j} - x_{1_j}}{x_{1_j}}\right) \times 100\%}$$

where, x_1 is the parameter value of the baseline simulation for parameter j and x_2 is the changed value of the model parameter. A value of SR equal to 1 or -1 would indicate a proportional change (i.e. increase or decrease, respectively) of the model output for the respective change of a model parameter value. We defined parameters of low sensitivity as $-1 < SR < 1$ whereas sensitive parameters were those for which SR was outside this range. Values of RC and SR were interpreted with reference to 14 simulated state variables (temperature, salinity, dissolved oxygen, phosphate, total phosphorus, nitrate, ammonium, total nitrogen, cyanophytes, cryptophytes, diatoms, suspended minerals, *Ruppia* and macroalgae).

2.6.4 Sensitivity of *Ruppia* to salinity limitation parameter

Ruppia spp. are typically described as salt tolerant and are abundant in a wide variety of habitats (i.e. fresh, brackish and saline waters) throughout temperate and tropical regions in New Zealand and Australia (e.g. Mason 1967; Brock 1982; Carruthers *et al.* 2007; Champion and Clayton 2004). However, germination experiments with *R. polycarpa* have indicated that germination and growth of seedlings may be depressed at high salinities (Gerbeaux 1989; Sim *et al.* 2006), and increases in salinity in Waituna Lagoon have been linked with the decline in *Ruppia* spp. observed between 2009 and 2011, as the length of time during which the lagoon was open to the sea increased during the same period (Robertson and Funnell 2012).

The response of modelled *Ruppia* biomass to varying degrees of salinity limitation was investigated by varying the parameter β, i.e. salinity limitation value at maximum salinity. The value of β was varied between 1, which represents no salinity limitation at all, and 20, which represents very severe salinity limitation. ± 10% of the calibrated value for β was also included in this sensitivity analysis. The calibrated model (with salinity limitation parameter $\beta = 1.6$) was also used to replicate the experimental conditions of Sim *et al.* (2006), which measured the biomass of *R. polycarpa* seedlings subjected to differing salinity treatments. The effect of the model salinity limitation function on *Ruppia* biomass was then compared with the experimental results of Sim *et al.* (2006).

2.7 DYRESM-CAEDYM scenarios

The Lagoon Technical Group (LTG) consists of a group of experts appointed by ES to provide scientific advice and to comment on technical matters that affect the functioning of the lagoon. The LTG, together with Environment Southland, decided upon 15 scenarios to simulate with the DYRESM-CAEDYM model, in order to better understand the range of responses of Waituna Lagoon to different possible environmental and management regimes. The scenarios were not intended to provide definitive future quantitative values for variables because it is not possible *a priori* to know, for example, the timing of lagoon closures, climate and its interaction with defined lagoon opening triggers, and other water quality triggers. Instead the scenarios were designed to contribute to a body of knowledge to allow for informed decision. The scenarios were decided upon through a voting system based on contributions from the LTG and Environment Southland and allocated to (i) hydrological, (ii) nutrient and sediment input, (iii) ecological and (iv) cultural categories. The scenarios as provided to us by ES on 16 July 2012 are described in Table 3. The scenarios were simulated using input data as for the 2001–2011 calibration/validation period, with particular inputs (e.g. outflows, tidal inflows, nutrient loads) adjusted as per the requirements of each scenario. Of the fifteen scenarios requested by ES, one was conceptually identical to the calibration/validation period (which could also be described as the "status quo", or base scenario), and two were identical and represented a "natural" opening regime (i.e., no artificial openings, with the lagoon left to overflow or breach at water levels > 4 masl). There were difficulties simulating the ecological scenarios due to unsuitability for conceptualisation or lack of input data (Section 2.7.3). Additional to the scenarios provided by the LTG, we included one nutrient/sediment input scenario to explore the effect of managing only phosphorus loads. Furthermore, as requested by ES staff/LTG members Karen Wilson, Barry Robertson and Greg Ryder on 21 August 2012, we simulated additional hydrological scenarios (involving winter openings) and combined scenarios comprising of the natural opening regime or winter opening scenario combined with nutrient input reductions. In total, 23 scenarios were simulated using DYRESM-CAEDYM (Table 4).

Table 3: Scenarios as provided by the LTG/ES on 16 July 2012 (see text for description of scenarios simulated using DYRESM-CAEDYM)

	Water level trigger (masl)	Description
Hydrological scenarios		
1[+]	> 2	"Natural" opening regime scenario: "Natural" is defined here as leaving the lagoon to breach itself probably at water levels over 4.0msl
2	> 2	Open on at 2.8msl
3*	> 2	Status Quo, open only at current water level trigger of 2.0 m
4	> 1.5	Spring Opening at level between 1.5-1.85 to trial rapid mouth closure*1 month open
5	> 2	Long Summer Opening * 6 months
6	> 2	Climatic (average annual rainfall, water temps, air temps)
7	> 2	Climatic (dry year <25% annual rainfall, increased evaporation)
8	> 2	Climatic (wet year >200% annual rainfall, decreased evaporation)
Nutrient and sediment input scenarios		
9		Nutrient decrease 10% of non-natural inputs
10		Nutrient decrease 25%
11		Nutrient & Suspended Sediment decrease 10%
Ecological scenarios		
12		Open only when chlorophyll a is >12 µg/l for at least 14 days or rises to >20 µg/l within 1 week. Closing will be uncertain. In this situation, lagoon would be left to flood surrounding land or overflow to sea.
13		Lagoon Ecological condition in 1976 (Ramsar)
14		Lagoon Ecological condition in 1995
Cultural health scenarios		
15[+]		Leave closed and breach by its self, to allow movement of Taonga Species and to enhance and maintain Mauri

[+] Both of these scenarios represent the lagoon in a "natural" condition (i.e. left to overflow/breach at water levels > 4 masl).

* This scenario represents the base scenario (i.e. calibration/validation period).

2.7.1 Hydrological scenarios

These scenarios required adjustments to the water balance to derive inflows and outflows based on opening timing and duration, and are briefly described below.

- A "Natural" opening regime scenario in which the barrier is not artificially opened and water breaches the lagoon when water levels are > 4 masl. Water losses occur naturally through barrier seepage and evaporation. Barrier seepage is determined based on water levels (Section 2.4.2) and evaporation is accounted for as a component of the computation within the DYRESM-CAEDYM model. Based on 2001-2011 input data, water level did not exceed 4 masl, (i.e. losses from barrier seepage and evaporation were enough to maintain water levels < 4 masl), so the lagoon was not opened at all in this scenario.
- A “High-level Opening”, based on a lagoon water level trigger of 2.8 masl. This regime required modification of the water balance involving openings for two months after water levels reached 2.8 masl (which occurred twice in this scenario based on 2001–2011 data, with barrier seepage and evaporation accounted for as above).
- A “Winter Opening” scenario, where the lagoon is opened for 3 months in winter each year.
- A “Variable Winter Opening”, where the lagoon is opened in in winter each year, with the length of the opening varied (at random) for between 1 and 6 months.
- A “Status Quo” scenario, where the lagoon is opened only at the current water level trigger of 2 masl. This regime equates to the period for which the model has been calibrated and validated, and so is represented by the base scenario.
- A “Spring Opening” of one-month duration, that may be optional for water levels between 1.5 and 1.85 masl, and obligatory for water levels > 1.85 masl, as requested by the LTG. However, based on 2001–2011 input data the 1.85 masl water level trigger for opening was always exceeded in spring, due to high rainfall and surface discharge in the months preceding the spring opening. Thus, this scenario involved a 1 month opening in September at a variable trigger level (1.8 – 2.7 masl).
- A “Long Summer Opening” involving a six-month opening from November to April each year.
- Climatic modifications, involving three separate one year simulations based on annual (a) “average rainfall”, (b) “low rainfall” and (c) “high rainfall”, with other meteorological input variables unaltered. In order to carry out these simulations median, minimum and maximum total rainfall and surface inflow years were selected from the simulation period (2001–2011), with the “high” and “low” rainfall years having c. 20% more or less rainfall than the “average” year. (N.B. The “wet year” scenario requested by LTG/ES (i.e. >200% annual rainfall) was not considered to be realistic). Each of the “average”, “high” and “low” years was run with a one-month opening in spring. Climate and inflow data were specific to the year allocated. The objective of these simulations was to contribute information on the lagoon response to prolonged periods of drought or high rainfall.

2.7.2 Nutrient and sediment input scenarios

In these scenarios, external nutrient loads were adjusted over the entire 2001–2011 period. All other inputs used measured 2001–2011 data (e.g., opening timing and duration, meteorology data were as for the base scenario). Nutrient and sediment loads were adjusted in the freshwater inflows only (i.e. not tidal inflows).

- A decrease in the nutrient load by 10% of the non-natural (anthropogenic)-related load. To formulate this scenario we used the Waituna Lagoon catchment area of 17,973 ha and considered that areal nutrient loads before humans were in the catchment were 0.2 kg N ha^{-1} yr^{-1} and 0.1 kg P ha^{-1} yr^{-1}, yielding a total catchment load of 3594.6 kg N and 1797.3 kg P. These areal loads are based on current estimates provided by AgResearch for Department of Conservation and forestry land in the Waituna Lagoon catchment, and assume that the entire catchment was peatland. The estimates may vary considerably from this. For example, indigenous scrub and forest is commonly given areal nutrient loads of 2–4 kg N ha^{1} yr^{-1}. According to AgResearch estimates, the current (2011) nutrient load to Waituna Lagoon is 399,534 kg N and 19,396 kg P. These values compare with our model input loads of 358,640 kg N and 17,480 kg P for 2011 (the average values for 2001–2011 (259,660 kg N and 14,390 kg P) are provided in Table 1). Slight differences may be attributable to errors inherent in the load derivation approach, e.g. monthly interpolation of measured data which may underestimate storm flow loads. The difference between the 2011 and "natural" loads represents what might be considered to be "manageable", i.e. 395,939 kg N and 17,599 kg P. No modifications were made to the discharge for these scenarios though some alterations might be expected with different rates of evapotranspiration between "natural" vegetation and "human-altered" vegetation. Moreover, much of the dissolved organic load is likely to be difficult to manage because of destruction of the original peatbog, which has resulted in a largely unmanageable flux of dissolved organic carbon, nitrogen and phosphorus that now arises from the agricultural catchment. Further consideration could be given to the definition of what component of the load could be manageable. However, under the assumption that the difference between the 2011 and "natural" load is manageable we applied a factor (0.9) to that fraction of the concentrations of N and P species in the inflow file. From the figures above, 99.1 and 90.7% of the respective N and P loads were considered manageable.
- A decrease in the total nutrient load by 25%. Because loads are strongly dominated by human inputs this figure would not be dissimilar to the above scenario but using 25% instead of 10% for the percentage reduction in manageable load. For this scenario the nutrient concentrations in the inflows were reduced by applying a factor of 0.75 to all nutrient species in the inflow file, again under an assumption that the hydrology was unaffected from the base case.
- A decrease in the total nutrient and sediment load by 10%. This scenario was designed to consider reductions in sediment loads as well as nutrients. Both nutrient and suspended sediment concentrations in the inflows were reduced by applying a factor of 0.9 to all nutrient species and suspended sediments in the inflow file, with unaltered hydrology.
- A decrease in phosphorus and sediment loads by 25%. This scenario was not explicitly requested by ES, by was added in order to investigate the effect of managing only phosphorus/sediment loads. Phosphorus and sediment concentrations in inflows were reduced by applying a factor of 0.75 to phosphate and organic phosphorus in the inflow files, as well as suspended sediments, again under an assumption that the hydrology was unaffected from the base case.
- A decrease in nitrogen loads by 25% and decrease in phosphorus loads by 50% in freshwater inflows (hydrology unaffected from the base case).
- A decrease in nitrogen loads by 50% and decrease in phosphorus loads by 25% in freshwater inflows (hydrology unaffected from the base case).
- A decrease in total nutrient loads by 50% (hydrology unaffected from the base case).
- A decrease in total nutrient loads by 90% (hydrology unaffected from the base case).

2.7.3 Ecological and cultural health scenarios

- The “managed ecological feedback” scenario that required the lagoon to be opened when chlorophyll *a* concentrations exceeded either 12 µg L^{-1} for at least 14 days or 20 µg L^{-1} for 7 days (with closure assumed to occur when chlorophyll *a* concentrations were < 10 µg L^{-1} for a period of one week), was not simulated because there were no occurrences of simulated chlorophyll *a* exceeding the trigger concentrations prescribed above.
- The “1976 ecological condition” and “1995 ecological condition” scenarios that required nutrient load reductions based on land use in 1976 and 1995 (as compared to current land use) were not simulated because of uncertainties around the land use data available at the time of writing to prescribe the nutrient load reductions.
- A “Cultural Health Scenario” was designed to simulate natural breaching and closure of the lagoon. This option was identical to the first scenario, and is not reported separately. For simulation results pertaining to this scenario, readers should refer to the “Natural opening regime” scenario simulation results.

Table 4: Scenarios simulated using DYRESM-CAEDYM in this study

	Scenario	**Description**
Hydrological scenarios		
1	-O	Natural opening regime/Cultural health scenario
2	O2p8	Lagoon opens (for 2 months) when water level > 2.8 masl
3	OW3	Lagoon opens for 3 months in winter each year
4	OWV	Winter opening of variable length (1-6 months)
5	OSp1	Lagoon opens for 1 month in spring each year
6	OS6	Lagoon opens for 6 months in summer each year
7	CAve	Average rainfall year
8	CDry	Low rainfall year
9	CWet	High rainfall year
Nutrient and sediment reduction scenarios		
10	NP-10	10% decrease in “non-natural” nitrogen and phosphorus in freshwater inflows
11	NPS-10	10% decrease in nitrogen, phosphorus and suspended sediments in freshwater inflows
12	NP-25	25% decrease in nitrogen and phosphorus in freshwater inflows
13	PS-25	25% decrease in phosphorus and suspended sediments in freshwater inflows
14	N25P50	25% decrease in nitrogen and 50% decrease in phosphorus in freshwater inflows
15	N50P25	50% decrease in nitrogen and 25% decrease in phosphorus in freshwater inflows
16	NP-50	50% decrease in nitrogen and phosphorus in freshwater inflows
17	NP-90	90% decrease in nitrogen and phosphorus in freshwater inflows
Combined hydrological and nutrient reduction scenarios		
18	-ONP-50	Natural opening regime and 50% decrease in nitrogen and phosphorus in freshwater inflows
19	-ONP-70	Natural opening regime and 70% decrease in nitrogen and phosphorus in freshwater inflows
20	-ONP-90	Natural opening regime and 90% decrease in nitrogen and phosphorus in freshwater inflows
21	OW3N50P25	Lagoon opens for 3 months in winter each year and 50% decrease in nitrogen and 25% decrease in phosphorus in freshwater inflows
22	OW3NP-50	Lagoon opens for 3 months in winter each year and 50% decrease in nitrogen and phosphorus in freshwater inflows
23	OW3NP-90	Lagoon opens for 3 months in winter each year and 90% decrease in nitrogen and phosphorus in freshwater inflows

2.7.4 Combined hydrological and nutrient input scenarios

The natural opening regime scenario and the winter opening scenario were combined with nutrient load reductions in freshwater inflows.

- A "Natural" opening regime scenario in which the barrier is not artificially opened and water breaches the lagoon when water levels are > 4 masl was combined with nutrient load (nitrogen and phosphorus) reductions in freshwater inflows of 50%, 70% and 90%.
- A “Winter opening” scenario, where the lagoon is opened for 3 months in winter each year was combined with nutrient load reductions in freshwater inflows of 50% nitrogen and 25% phosphorus, 50% for both nitrogen and phosphorus, and 90% for both nitrogen and phosphorus.

2.8 ELCOM model description

ELCOM (Estuary and Lake Computer Model) is a three-dimensional hydrodynamic model developed by the Centre for Water Research at the University of Western Australia, based on the unsteady, viscous Navier-Stokes equations for incompressible flow (Hodges & Dallimore 2011). For this study, ELCOM was used to simulate temperature, salinity, water levels and current velocity in Waituna Lagoon, to investigate the effect of opening location on these variables. Although, ELCOM can be coupled with the aquatic ecological model CAEDYM to simulate three-dimensional transport and interactions between flow physics, biology and chemistry in a system (Hipsey 2011), this coupling is computationally very demanding and time constraints prevented coupling of ELCOM with CAEDYM in this study.

2.9 ELCOM model inputs

As ELCOM is a three-dimensional model, runs times are significantly greater than for the one-dimensional model, DYRESM. Thus, it was not possible to run ELCOM for the same time period as DYRESM-CAEDYM (i.e. 10 years; Oct 2001 to Oct 2011). Instead, three periods were chosen which covered a range in opening locations and durations.

1) July 2003 to July 2004, including two openings (242 and 35 days long) at Walker’s Bay,
2) July 2007 to July 2008, which included a short (40 days) opening at Walker’s Bay, and
3) May to October 2011, which included a short (34 days) opening at Charlie’s Bay.

For each of these periods, ELCOM was run at 60 s time steps, with 2 hourly outputs for water level, temperature, salinity and current velocity. Input data for freshwater inflows was derived as for the DYRESM-CAEDYM model. Hourly meteorological data (i.e. rainfall, wind speed and direction, solar radiation, air temperature, relative humidity and cloud cover) for Tiwai was acquired from NIWA’s Cliflo service (National Institute of Water and Atmospheric Research National Climate database, http://cliflow.niwa.co.nz). Outflow when the lagoon was closed (i.e. barrier seepage) was derived from a water balance, (as for DYRESM-CAEDYM), but when the lagoon was open the opening was treated as an open boundary and outflows simulated by ELCOM were based on water level. There are no measurements for water level at the opening (the only water level recorder being at the far eastern end of the lagoon at Waghorn’s Rd), so water level at the opening was derived from tide height and times at Bluff, acquired from NIWA’s tide forecaster (http://www.niwa.co.nz/services/online-services/tide-forecaster).

2.10 ELCOM bathymetry and model setup

A hydrographic survey of Waituna Lagoon was undertaken by ES in December 2011 when the lagoon water level was c. 1.65 masl, providing high resolution coverage at elevations up to c. 1.4 masl. However, lagoon levels frequently exceed 1.4 masl, and 4 masl is the estimated maximum lagoon level based on barrier elevation (Johnson & Partridge 1998), so LIDAR data (also provided by ES) was used to extend bathymetry to 4 masl (Figure 10). Bathymetry points were interpolated to produce a horizontal model grid of 50 m x 50 m (vertical grid resolution was set to 0.5 m). The inflow boundary conditions included four surface freshwater inflows (Waituna, Moffat and Carren Creeks, and Carren Creek tributary), and a groundwater inflow, assumed to enter the northern side of the lagoon at sea level. When the lagoon was closed the outflow boundary conditions included outflow cells in the gravel barrier in the southwest and southeast ends of the lagoon. Two opening locations were defined as open boundary cells, where inflows and outflows were simulated by ELCOM based on water level (i.e., subject to tidal forcing). One open boundary was at Walker’s Bay and one at Charlie’s Bay (Figure 5). In reality, lagoon opening width and depth tend to be highly variable, ranging from 10 to 200 m wide and 1 to 5 m deep (at Walker’s Bay), although the width at Charlie’s Bay tends to be confined to 50 to 70 m due to lagoon morphology (Greg Larkin, pers. comm.). It should be noted, however, that the model does not allow for alteration of open boundary size during the course of simulations. The open boundaries were set to be 150 and 50 m for the Walker’s Bay and Charlie’s Bay openings, respectively.

Model output was generated for a number of locations (stations) around the lagoon, and as two-dimensional (2D) cross-sections/maps, to aid with model calibration and with visualisation of output data. These were:

- Profiles for the four ES sampling sites in the lagoon (Figure 5)
- Profiles for the water level recorder at Waghorn’s Rd (Figure 5)
- Profiles for the Walker’s and Charlie’s Bay openings (Figure 5)
- A “curtain” (i.e. cross-section) extending west to east across the lagoon
- Surface and bottom “sheets” (i.e. 2D maps)

Following preliminary simulations that indicated salinity stratification was sometimes present in the deeper channel in the eastern arm of the lagoon, a profile in this location was added to the model output. Co-ordinates for each of the stations are included in Table 5.

Table 5: Stations included in ELCOM output

	New Zealand Map Grid	
	Easting	Northing
ES sampling sites		
Lagoon Centre	2171404	5395825
Lagoon East	2173508	5395599
Lagoon West	2170376	5395790
Lagoon South	2171698	5394133
Extra profile (in ELCOM, not an ES sampling site)		
Lagoon East Arm	2175562	5395264
Water level recorder		
Waghorn's Rd	2177154	5395726
Opening locations		
Walker's Bay	2171745	5393260
Charlie's Bay	2175030	5394565

2.11 ELCOM calibration and output

ELCOM was calibrated against field data, i.e., water level measured at Waghorn's Rd, and temperature and salinity collected at four sites within the lagoon (centre, east, west and south) for each of the three simulation periods. As for DYRESM-CAEDYM, the model error was represented by a series of model performance statistics, the root-mean-square-error (RMSE), normalised root-mean-square-error (NRMSE), and Pearson correlation coefficient (R).

Surface and bottom sheets were used to visualise movement of saline water throughout the lagoon following openings, and a curtain (i.e. vertical transect from west to east across the lagoon) was used to illustrate the effect of wind setup on lagoon level.

To investigate the influence of opening location on salinities and current velocities in the lagoon each of the calibration periods was run again, but with the other opening location used as the open boundary, i.e. for the 2003 and 2007 simulations the opening location was changed from Walker's Bay to Charlie's Bay, and for the 2011 simulation the opening location was changed from Charlie's Bay to Walker's Bay. This allowed us to quantify the effect of opening location on hydrological variables around the lagoon.

2.12 Remote Sensing

The remote sensing preliminary investigation used two Landsat 5 images captured on 20 December 2010 and 21 January 2011. Also one Landsat 7 image was analysed which was captured on 20 March 2012. Images were investigated visually using true colour composites of Landsat bands 1, 2 and 3 (blue, green and red). Further investigation included a two-stage image classification (K-means classification), in order to group spectrally similar pixels. The first stage identified land areas and eliminated them from further analysis. The second stage classified water pixels only.

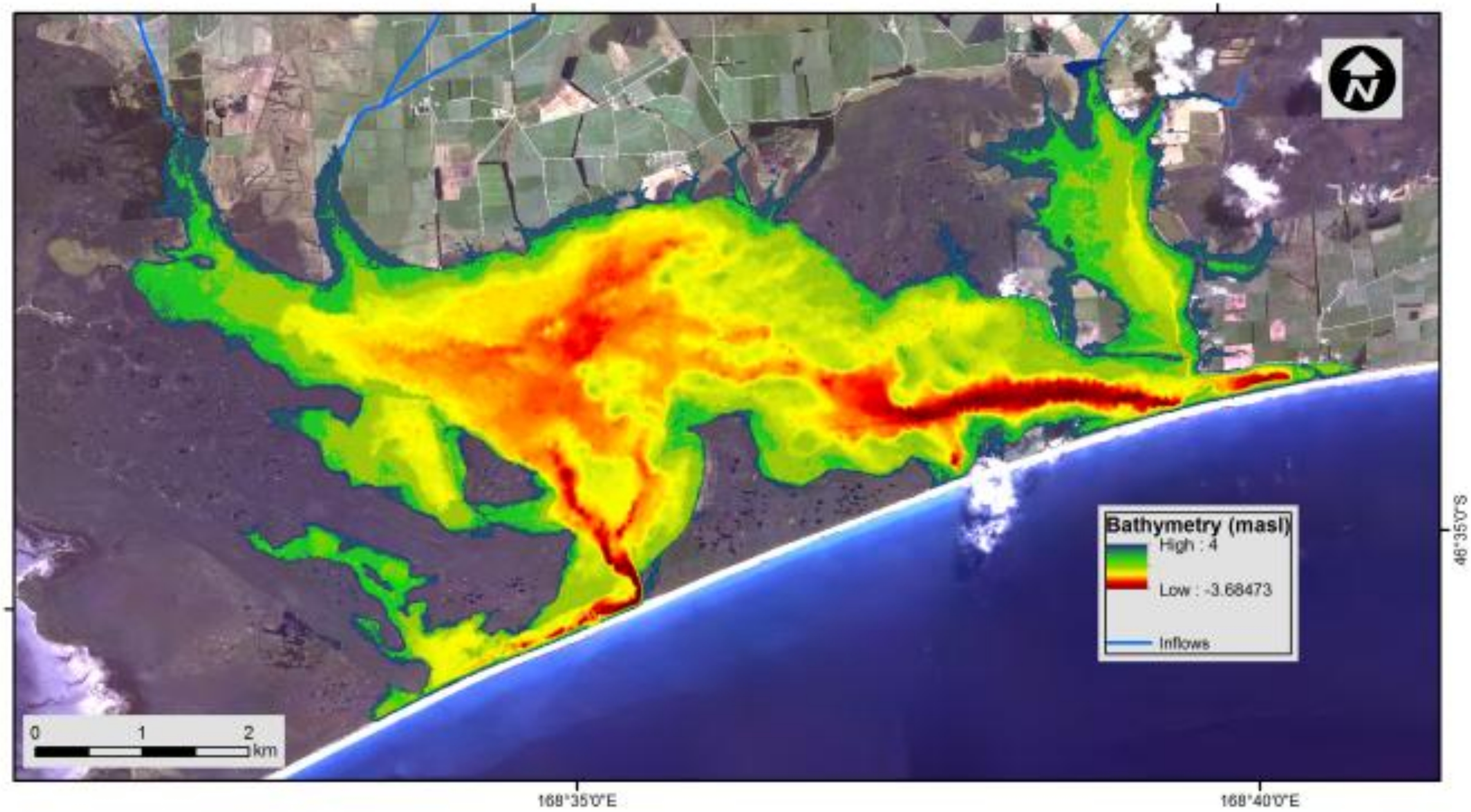


Figure 10: Waituna Lagoon bathymetry

3 Results

3.1 DYRESM-CAEDYM calibration and validation

The model parameters adjusted during the calibration of DYRESM-CAEDYM are included in Appendix 2. Parameter values were assigned within the range found in the literature (e.g. Schladow & Hamilton 1997; Trolle *et al.* 2011b; Özkundakci *et al.* 2011). Visual comparisons of modelled temperature, salinity, dissolved oxygen, PO_4-P, TP, NO_3-N, NH_4-N, TN, total chl *a*, TSS, *Ruppia* and macroalgae biomass with available field measurements are shown in Figures 11-13. *Ruppia* and macroalgae biomass is shown as a function of elevation in the lagoon in Figure 14.

The overall model performance was assessed statistically using several model statistics (Pearson's R, root mean square error, mean absolute error, normalised mean absolute error, mean signed difference and the mean of both the field observations and modelled values; Table 6). These values suggest that the model was able to reproduce the magnitude and dynamics of field measurements, and performed well compared to other published model applications for similar systems (e.g. Burger *et al.* 2008; Fragoso *et al.* 2011; Gal *et al.* 2009; Trolle *et al.* 2011b; Özkundakci *et al.* 2011). The model successfully reproduced moderate chlorophyll *a* variations associated with phytoplankton blooms, and periods with very low biomass (< 3 µg L^{-1}), but did not capture some of the high (> 12 µg L^{-1}) chl *a* field measurements. Model statistics suggest the model was most successful at simulating variables such as NO_3-N, TN, temperature, salinity, and dissolved oxygen. Where model statistics suggested that the model performed less well, e.g., for PO_4-P, NH_4-N and chl *a*, closer examination of field measurements revealed that a high proportion (i.e. 25 – 40%) were below detection limits, which can restrict the ability of model statistics to define model error (Table 7).

Comparison of modelled *Ruppia* biomass with field survey data suggests that the model was able to reproduce the declining trend (from 2007 – 2011) identified in field surveys. Though macroalgae biomass was not quantified, the surveys suggested that macroalgae abundance increased from 2007 until 2010, but was then decreased in the summer of 2011, a trend which was also simulated in the model.

Table 6: Statistical comparison of DYRESM-CAEDYM model simulations with field data (monthly measurements) of surface water in Waituna Lagoon, using Pearson correlation coefficient (R), root mean square error (RMSE), mean absolute error (MAE), normalised MAE (NMAE), mean signed difference (MSD), mean of observations (Mean$_{obs}$), and mean of model values (Mean$_{mod}$), for each variable.

	Calibration (2001 – 2007)							**Validation (2007 – 2011)**						
Variable	**R**	**RMSE**	**MAE**	**NMAE**	**MSD**	**Mean$_{Obs}$**	**Mean$_{Mod}$**	**R**	**RMSE**	**MAE**	**NMAE**	**MSD**	**Mean$_{Obs}$**	**Mean$_{Mod}$**
Temperature (^{o}C)	0.94	1.57	1.24	0.11	0.06	11.70	11.42	0.97	2.66	1.43	0.13	0.54	11.41	11.40
Salinity	0.97	3.10	1.99	0.14	-0.97	14.52	14.10	0.96	3.37	2.07	0.21	-0.69	10.08	8.49
Dissolved oxygen (mg L^{-1})	0.46	1.27	1.00	0.11	0.25	9.47	9.68	0.43	1.60	0.82	0.08	-0.47	10.65	9.99
Phosphate (mg L^{-1})	0.16	0.015	0.009	0.75	-0.005	0.013	0.008	0.67	0.011	0.006	1.04	0.005	0.006	0.008
Total phosphorus (mg L^{-1})	0.14	0.031	0.020	0.48	-0.006	0.041	0.035	0.34	0.031	0.021	0.55	0.004	0.038	0.039
Nitrate (mg L^{-1})	0.89	0.206	0.161	0.57	0.090	0.283	0.380	0.92	0.254	0.185	0.37	-0.005	0.504	0.474
Ammonium (mg L^{-1})	0.57	0.016	0.013	0.64	0.004	0.020	0.024	0.77	0.021	0.017	0.84	0.011	0.020	0.029
Total nitrogen (mg L^{-1})	0.77	0.364	0.284	0.36	0.001	0.798	0.803	0.89	0.427	0.314	0.28	-0.162	1.128	0.932
Chlorophyll *a* (µg L^{-1})	0.39	6.18	3.82	0.71	-1.99	5.38	3.36	0.30	6.20	3.38	0.70	-1.01	4.85	4.06
Total suspended solids (mg L^{-1})	0.30	5.40	3.93	0.63	-1.03	6.25	5.43	0.11	6.05	4.25	0.78	-0.32	5.45	5.30

Table 7: Laboratory detection limits for field data

Variable	**ES laboratory/Cawthron (09/10/2001 – 30/04/2005)**	**MLS Envirolab/Cawthron (01/05/2005 – 30/06/2008)**	**Hills Laboratory (01/07/2008 – 31/10/2011)**	**Percentage of observations below detection limits (2001–2011)**
Phosphate (mg P L^{-1})	0.005	0.005	0.004	39
Total phosphorus (mg L^{-1})	0.01	0.01	0.004	3
Nitrate (mg N L^{-1})	0.01	0.01	0.002	26
Ammonium (mg N L^{-1})	0.01	0.01	0.01	40
Total nitrogen (mg L^{-1})	0.08	0.08	0.05	0
Chlorophyll *a* (µg L^{-1})	0.6	0.6	3	25
Total suspended solids (mg L^{-1})*			3	34

*Total suspended solids collected from August 2008 onwards only. Prior to this, TSS was estimated from turbidity (NTU) measurements.

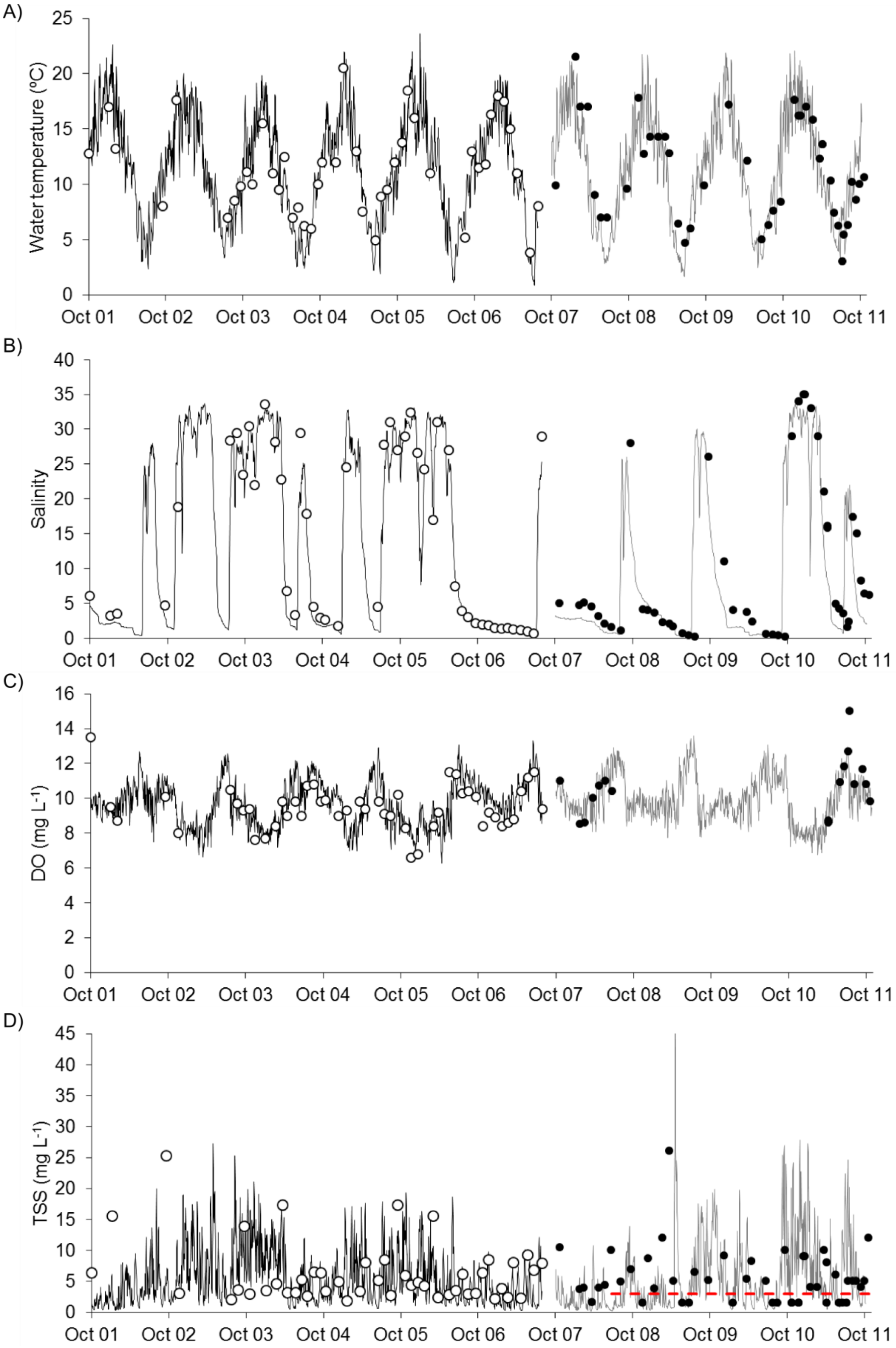


Figure 11: Modelled (DYRESM-CAEDYM) variables (black line = calibration and grey line = validation period) compared with field data (open circles = calibration and filled circles = validation period). A) Temperature (ºC), B) salinity, C) dissolved oxygen (DO, mg L^{-1}), and D) total suspended solids (TSS, mg L^{-1}). Dashed red line represents the detection limit for TSS field data. (N.B. TSS was only measured from August 2008. Prior to this, TSS was estimated from turbidity (NTU) measurements).

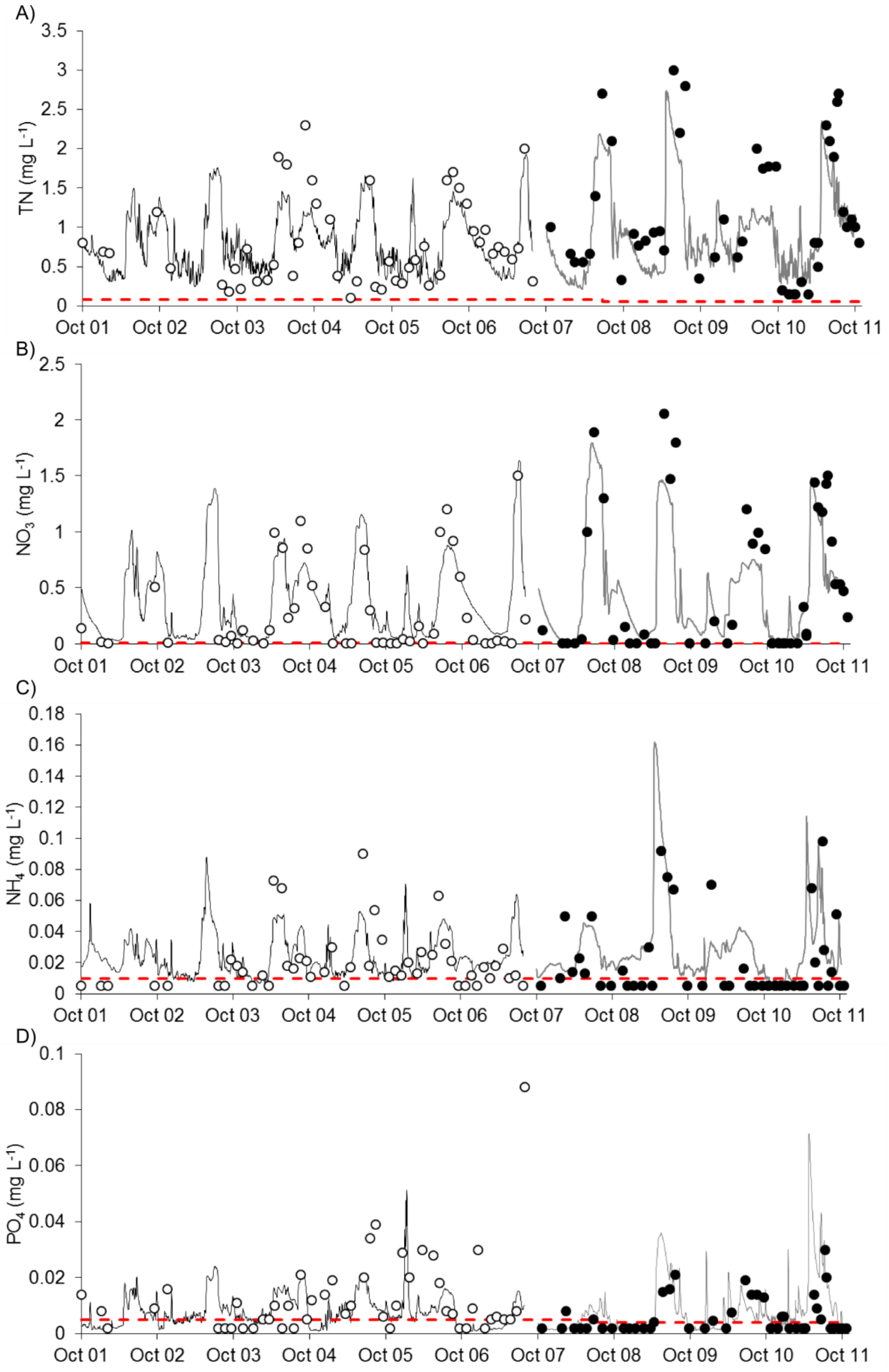


Figure 12: Modelled (DYRESM-CAEDYM) variables (black line = calibration and grey line = validation period) compared with field data (open circles = calibration and filled circles = validation period). A) Total nitrogen (TN), B) nitrate (NO_3-N), C) ammonium (NH_4-N), and D) phosphate (PO_4-P; all mg L^{-1}). Dashed red line represents the detection limit for field data.

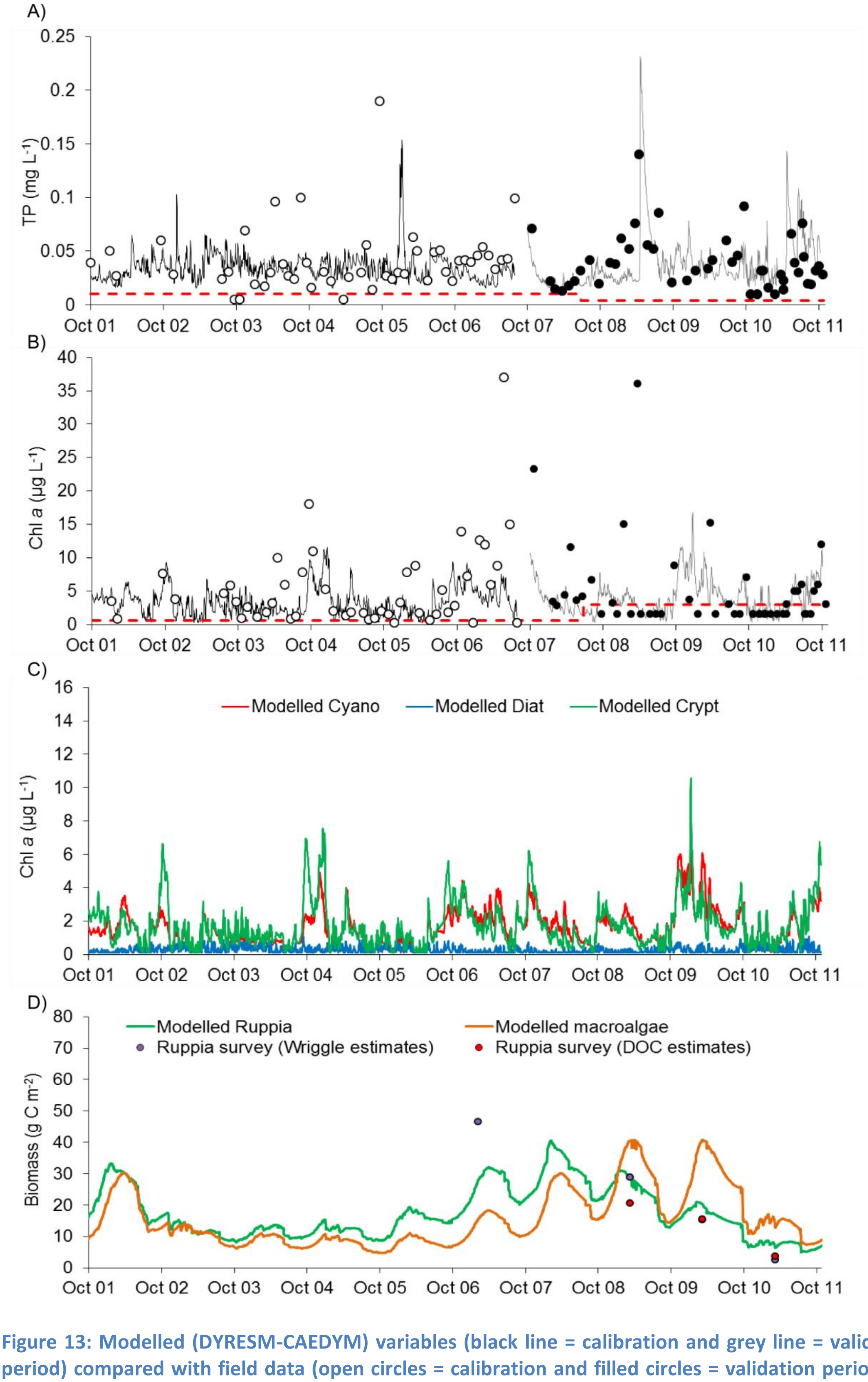


Figure 13: Modelled (DYRESM-CAEDYM) variables (black line = calibration and grey line = validation period) compared with field data (open circles = calibration and filled circles = validation period). A) Total phosphorus (TP; mg L^{-1}) and B) total chlorophyll *a* (chl *a*; µg L^{-1}). Modelled primary producer groups C) phytoplankton (cyanophytes, diatoms and cryptophytes) and D) *Ruppia* and macroalgae (circles represent measured *Ruppia* data derived from surveys). Dashed red line represents the detection limit for TP and chl *a* field data.

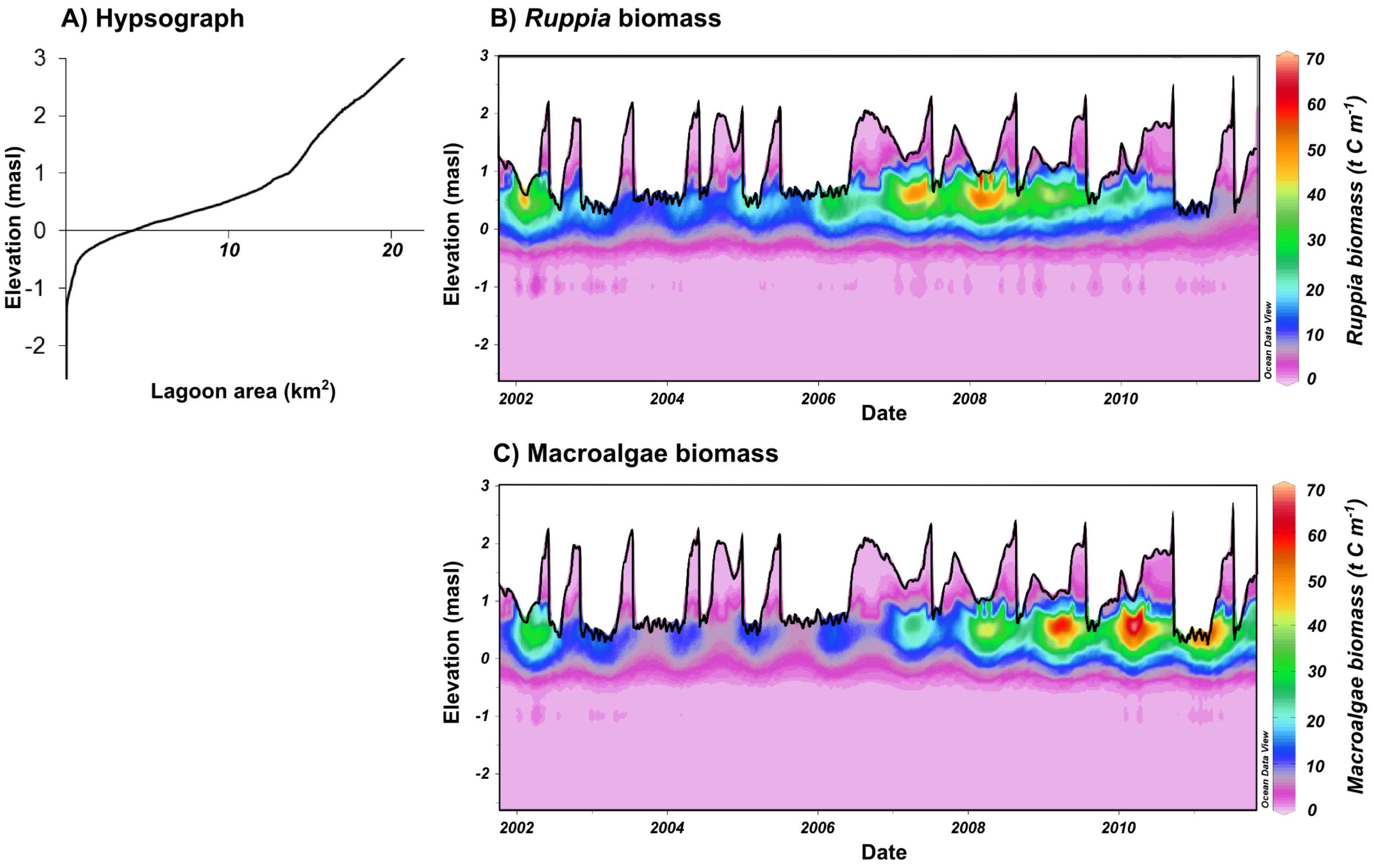


Figure 14: Waituna Lagoon hypsograph (A) and modelled (DYRESM-CAEDYM) *Ruppia* biomass (B) and macroalgae biomass (C) as a function of elevation (masl) for the calibration/validation period (Oct 2001 to Oct 2011). Low biomass at low elevations represents lower growth limits for *Ruppia* and macroalgae due to light limitation, while low biomass at high elevations represents upper growth limits due to desiccation stress. Seasonal patterns in *Ruppia* and macroalgae growth often apparent as high biomass in summer and lower biomass in winter, particularly for years with higher water levels.

3.2 DYRESM-CAEDYM sensitivity analysis

Visual comparison of model output for the calibrated model (base scenario) with output from sensitivity analyses (that manipulated model complexity and input data) are shown in Figures 15-18. Comparison of the daily mean value (± standard deviation) for each of the modelled variables, for the calibrated model and sensitivity analyses are shown in Figures 31-34.

3.2.1 Model complexity

Removal of macroalgae from the model produced simulations in which *Ruppia* biomass increased, and chlorophyll *a* decreased slightly, whilst removal of both macroalgae and *Ruppia* produced simulations in which chlorophyll *a* increased (Figure 16). Removal of macroalgae and *Ruppia* also appeared to increase total suspended sediments, total nitrogen and total phosphorus. Removal of macroalgae, *Ruppia* and resuspension (i.e., the least complex model configuration) resulted in chlorophyll *a* and total suspended sediment concentrations much lower than the calibrated model. Dissolved nutrients were affected to a lesser degree, but phosphorus concentrations were typically greater in simulations with reduced complexity (Figure 15).

3.2.2 Sensitivity to input data

A slight change (± 10%) in all nutrient and suspended sediment concentrations in all inflows had little effect on modelled variables (Figure 17, and Figures 31-34). Dissolved and particulate nutrients and suspended sediments, and chlorophyll *a* appeared to increase or decrease slightly with the corresponding increase or decrease in nutrients/sediment. The simulations indicate that *Ruppia* and macroalgae biomass may not be greatly affected by slight changes in nutrient/sediment concentrations (although macroalgae biomass did increase in the final part of the simulations in both cases).

In contrast, the modelled variables appeared to be more sensitive to changes in meteorological input data. Although, the effect of a 10% decrease in wind speed on modelled variables was slight, a 10% increase in wind speed resulted in a slight increase in chlorophyll *a* and total suspended solids, a decrease in macroalgae, and increase in *Ruppia* biomass (Figure 18, and Figures 33-34). Increasing/decreasing air temperature by 0.5 ºC had little effect on most model variables, except for macroalgae and *Ruppia*. In both cases, macroalgae biomass increased, whilst *Ruppia* decreased.

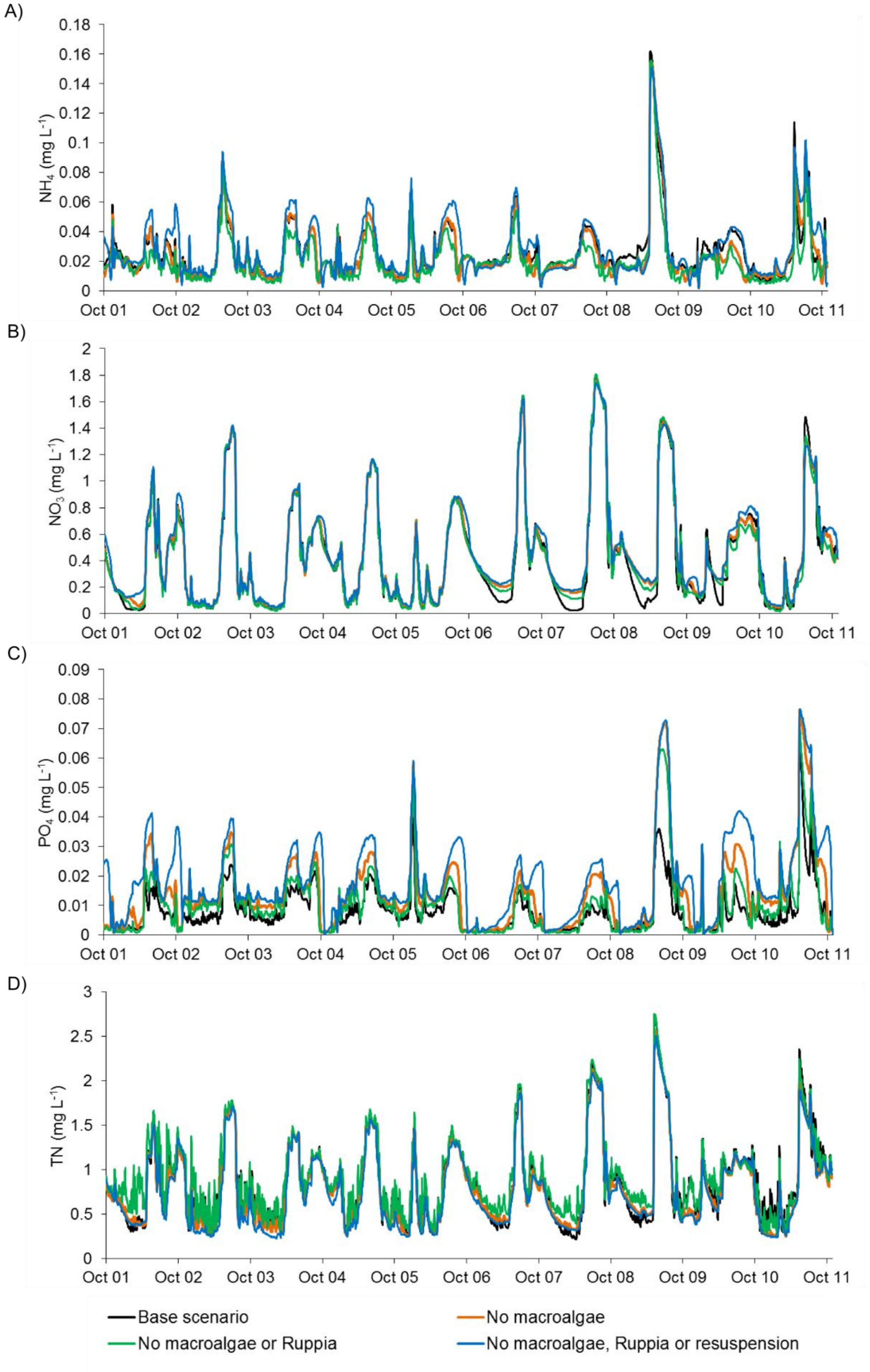


Figure 15: Effect of reducing model complexity on A) ammonium, B) nitrate, C) phosphate, and D) total nitrogen.

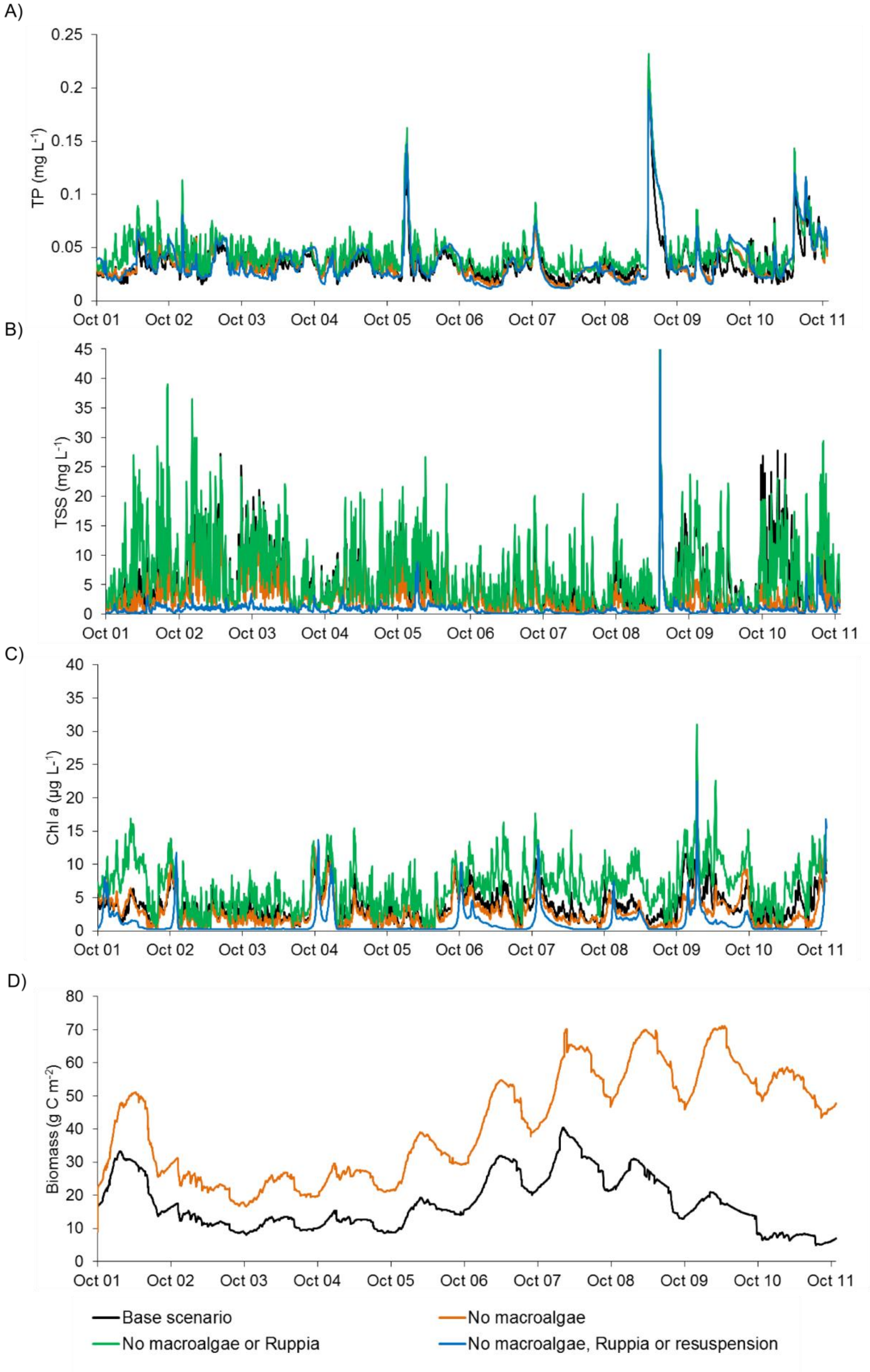


Figure 16: Effect of reducing model complexity on A) total phosphorus, B) total suspended solids, C) chlorophyll *a*, and D) *Ruppia* biomass.

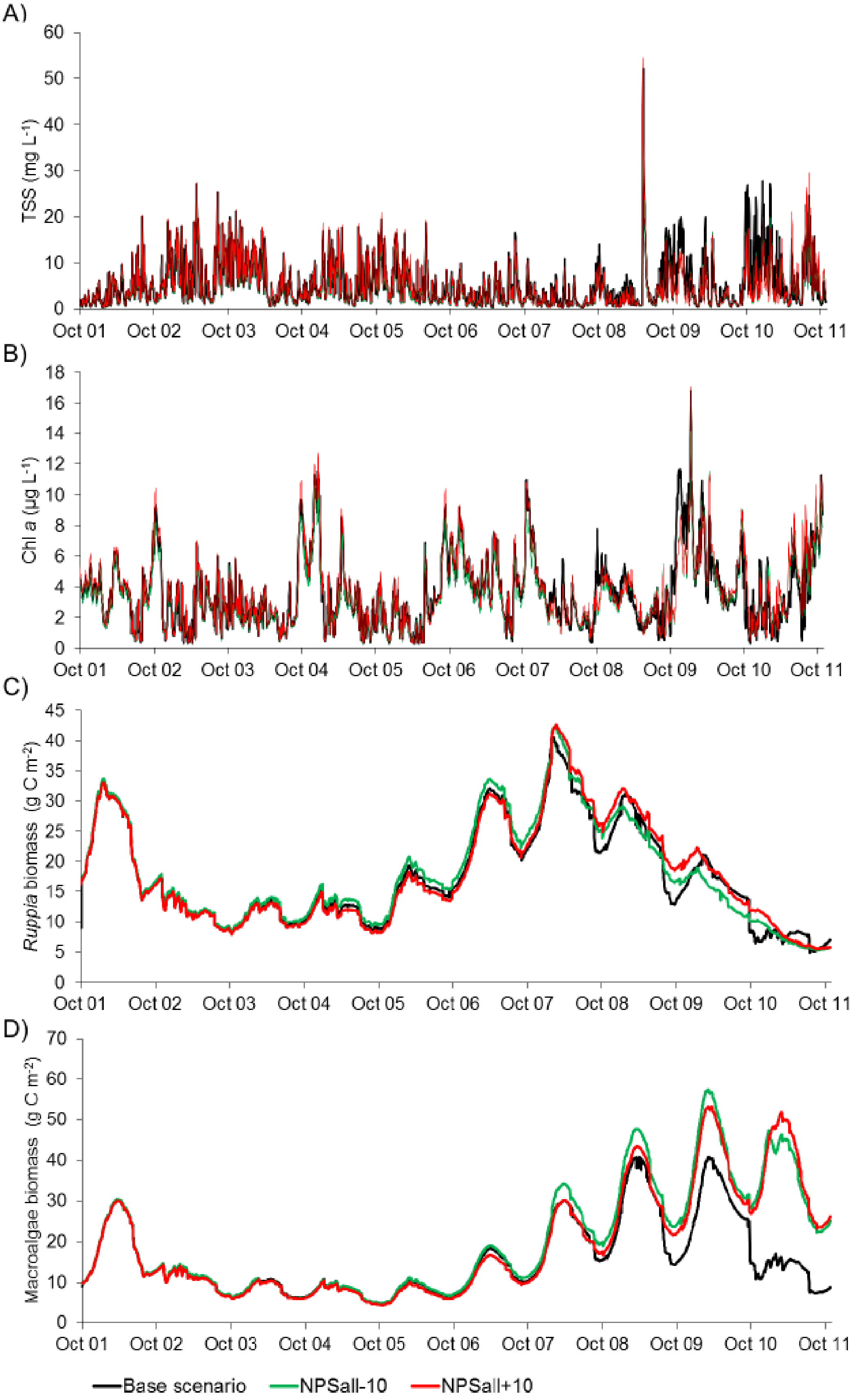


Figure 17: Effect of 10% increase or decrease in nutrients and sediments (NPSall+10 and NPSall-10, respectively) on A) total suspended solids, B) chlorophyll *a*, C) *Ruppia*, and D) macroalgae biomass.

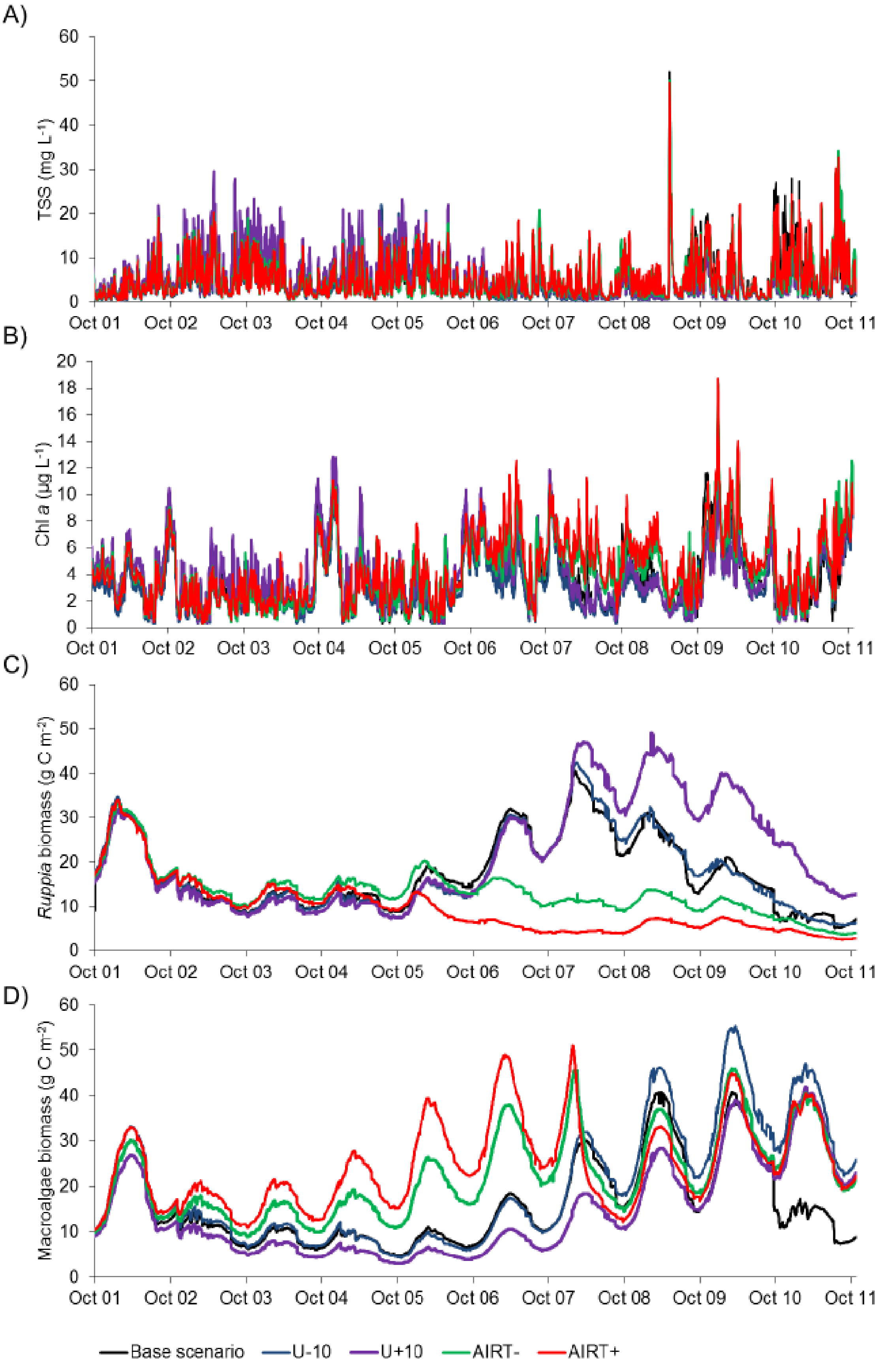


Figure 18: Effect of 10% increase or decrease in wind speed (U+10 and U-10) and 0.5 ºC increase or decrease in air temperature (AIRT+ or AIRT-) on A) total suspended solids, B) chlorophyll *a*, C) *Ruppia*, and D) macroalgae biomass.

3.2.3 Sensitivity to key parameters: Uncertainty analysis

Sensitivity analysis revealed that the 14 state variables considered were sensitive to many of the 23 parameters. In total, the sensitivity analysis resulted in 644 values of RC (relative change) and SR (sensitivity ratio). Values of RC ranged from -85% to 155% (Figure 19) whereas values of SR ranged from -8.5 to 15.6, with 10% of all values being either below -1 or above 1 (Figure 20). Values of RC indicated that phytoplankton species were moderately sensitive to changes in parameters which directly affect growth and/or mortality, and this effect was more pronounced for cyanophytes and cryptophytes. Changing phytoplankton growth parameters also affected water column nutrient concentrations; whereas parameters related to phytoplankton nutrient uptake had a minimal effect on nutrient concentrations. *Ruppia* and macroalgae biomass was highly sensitive to parameters related directly to the respective growth and mortality rates of these groups. Changes of these parameters also exposed some potential complex interactive effects of species-specific parameter changes on other biota and/or nutrient concentrations. For example, increasing the maximum potential growth rate for macroalgae resulted in a disproportionate increase of all simulated phytoplankton species and a decrease in phosphate and nitrate concentrations. Complex interactions were also observed for changes in parameters affecting *Ruppia* growth rate, which also resulted in a disproportionate change in cyanophyte and cryptophyte concentrations but had lesser effect on diatom concentrations. While 18 parameter changes (out of 46) resulted in a disproportionately large change in model output (i.e. SR outside the range -1 to 1) for at least one of the 14 state variables, most parameter changes resulted in SR values within the insensitive range (i.e. $-1 < SR < 1$).

3.2.4 Sensitivity of *Ruppia* to salinity limitation parameter

There is little effect of varying the *Ruppia* salinity limitation parameter β by ± 10% on *Ruppia* biomass (Figure 21 and Figure 23). In contrast, with no salinity limitation at all, modelled *Ruppia* biomass is increased over much of the simulation period, and appears to closely match observed data. With salinity limitation increased compared to the calibrated model (β = 3-20) simulated *Ruppia* biomass is decreased to very low levels throughout much of the simulation period.

The modelled salinity function (with the calibrated salinity limitation parameter $\beta = 1.6$) compares very well to the experimental results of Sim *et al.* (2006), although the influence of mid-range salinities is slightly reduced in the model compared to the experimental results (Figure 22).

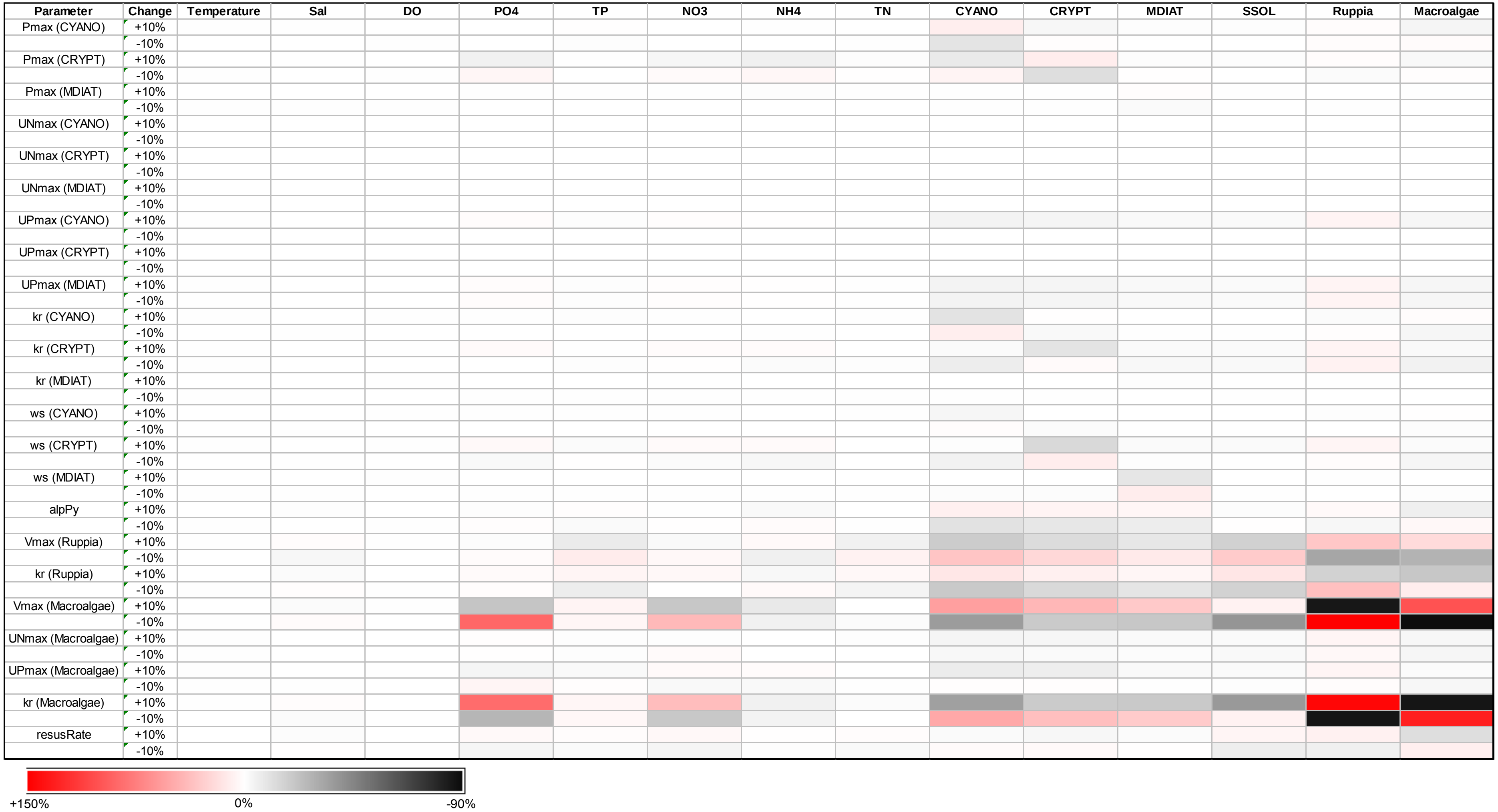


Figure 19: Summary of parameter sensitivity analysis using relative change (RC) as a sensitivity indicator. Details on calculations are described in the main body of the text. See Appendix 3 for descriptions of the parameter names.

Parameter	Change	Temperature	Sal	DO	PO4	TP	NO3	NH4	TN	CYANO	CRYPT	MDIAT	SSOL	Ruppia	Macroalgae
Pmax (CYANO)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟦	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
Pmax (CRYPT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟦	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟥	🟩	🟩	🟩	🟩
Pmax (MDIAT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
UNmax (CYANO)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
UNmax (CRYPT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
UNmax (MDIAT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
UPmax (CYANO)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
UPmax (CRYPT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
UPmax (MDIAT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
kr (CYANO)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟦	🟩	🟩	🟩	🟩	🟩
kr (CRYPT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
kr (MDIAT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
ws (CYANO)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
ws (CRYPT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟥	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟦	🟩	🟩	🟩	🟩
ws (MDIAT)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟦	🟩	🟩	🟩
alpPy	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟥	🟩	🟩	🟩	🟩	🟩
Vmax (Ruppia)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟥	🟥	🟩	🟥	🟦	🟦
	-10%	🟩	🟩	🟩	🟩	🟦	🟩	🟩	🟩	🟦	🟦	🟦	🟦	🟥	🟥
kr (Ruppia)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟦	🟩	🟩	🟦	🟥	🟥
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟥	🟥	🟩	🟥	🟦	🟦
Vmax (Macroalgae)	+10%	🟩	🟩	🟩	🟥	🟩	🟥	🟩	🟩	🟦	🟦	🟦	🟩	🟥	🟦
	-10%	🟩	🟩	🟩	🟦	🟩	🟦	🟩	🟩	🟥	🟥	🟥	🟥	🟦	🟥
UNmax (Macroalgae)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
UPmax (Macroalgae)	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
kr (Macroalgae)	+10%	🟩	🟩	🟩	🟦	🟩	🟦	🟩	🟩	🟥	🟥	🟥	🟥	🟦	🟥
	-10%	🟩	🟩	🟩	🟥	🟩	🟥	🟩	🟩	🟦	🟦	🟦	🟩	🟥	🟦
resusRate	+10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟥
	-10%	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟦

🟩 -1 < SR < 1
🟥 SR < -1
🟦 SR > 1

Figure 20: Summary of parameter sensitivity analysis using a sensitivity ratio (SR). A value of SR equal to 1 or -1 would indicate a proportional change (i.e. increase or decrease, respectively) of the model output with for the respective change of a model parameter value. Insensitive parameters were defined as -1 < SR < 1 whereas sensitive parameters were those for which SR was outside this range. Details on calculations are described in the main body of the text See Appendix 3 for descriptions of the parameter names.

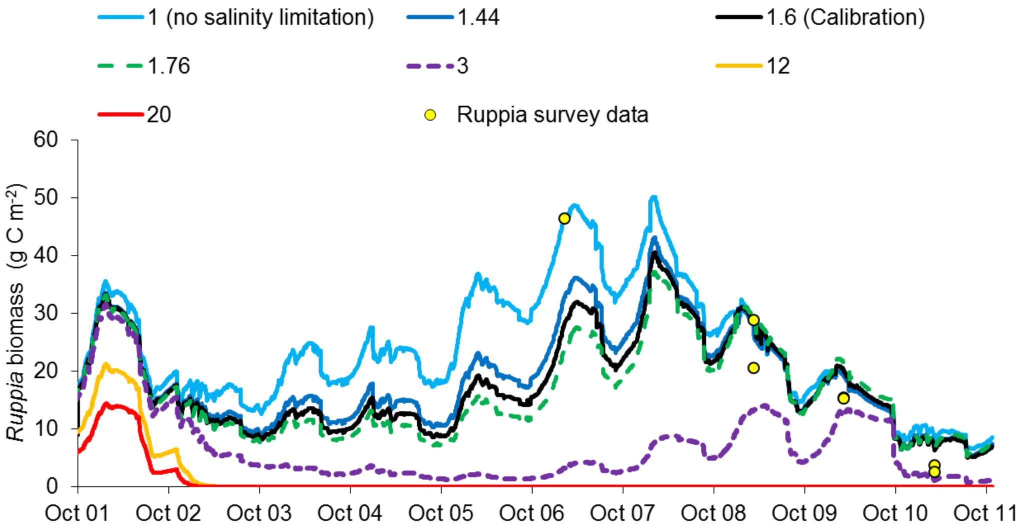


Figure 21: Modelled (DYRESM-CAEDYM) *Ruppia* biomass for the calibrated model (black line) and with varying salinity limitation. Numbers in legend refer to the value of the salinity limitation parameter *β* (i.e. salinity limitation at maximum salinity). 1 = no salinity limitation, 1.44 = calibrated value -10%, 1.6 = calibrated value, 1.76 = calibrated value +10%, 3, 12 and 20 = increasingly severe salinity limitation. *Ruppia* survey data included as yellow circles, as interpreted by Emily Funnell (DOC; using mid-points for % cover classes for 2009, 2010, and 2011) and by Leigh Stevens (Wriggle; using maximum for % cover classes, for 2007, 2009, 2010 and 2011).

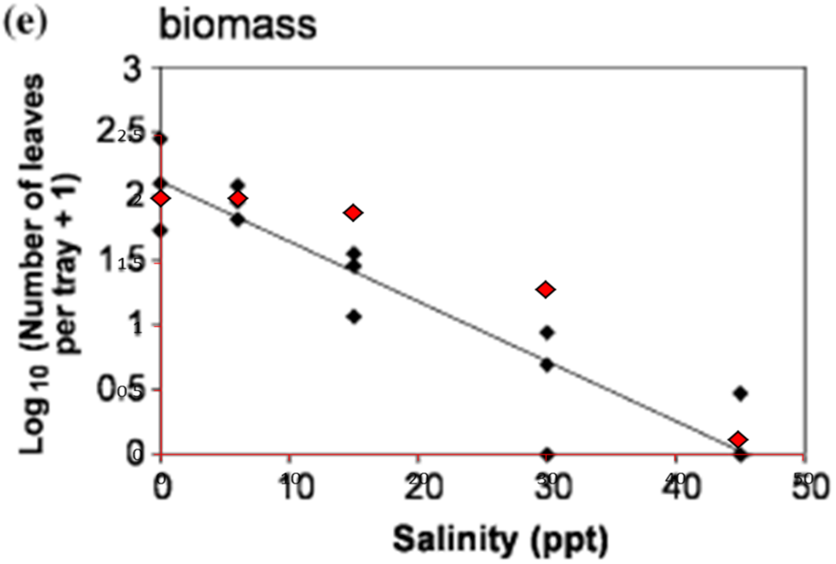


Figure 22: Experimental response of *R. polycarpa* biomass to increasing salinity (black diamonds) from Sim *et al.* (2006) compared with the response of *Ruppia* biomass to increasing salinity using the DYRESM-CAEDYM salinity limitation function (red diamonds).

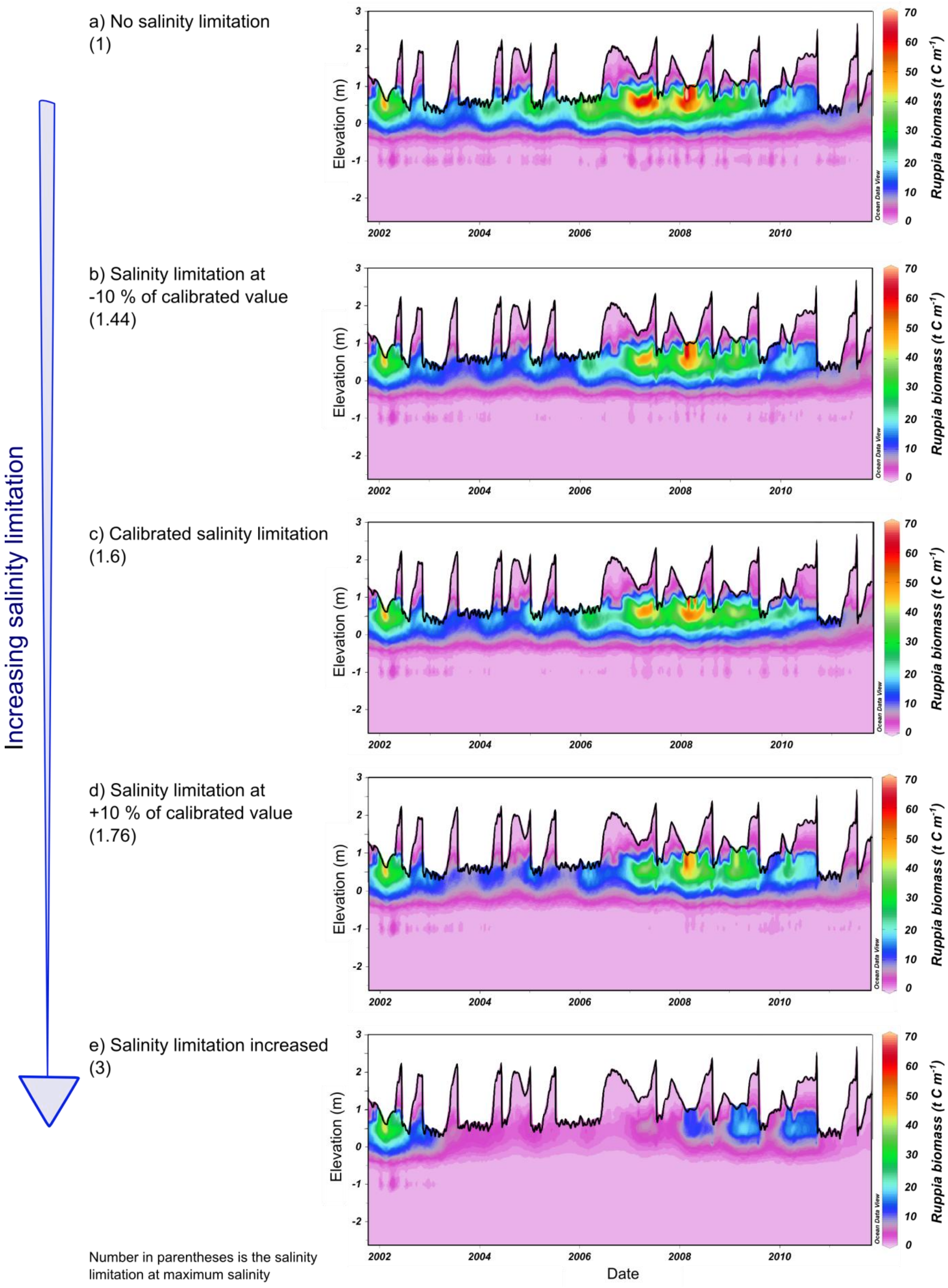


Figure 23: Modelled (DYRESM-CAEDYM) *Ruppia* biomass as a function of elevation at varying salinity limitation. a) no salinity limitation, b) -10% of calibrated model value, c) calibrated model, d) +10% of calibrated model, e) salinity limitation increased.

3.3 DYRESM-CAEDYM scenarios

Visual comparisons of model output for the calibrated model (base scenario) and hydrological, climate, nutrient/sediment, and combined scenarios are shown in Figures 24-29. *Ruppia* depth distributions for selected scenarios are shown in Figure 30. Comparison of the daily mean value (± standard deviation) for each of the modelled variables, for the calibrated model, sensitivity analyses and scenarios are shown in Figures 31-34.

3.3.1 Hydrological scenarios

The six hydrological scenarios that involved alterations to the lagoon opening regime had substantial effects on model variables (Figures 24-25 and Figures 31-34). For both the “natural opening regime” (where the lagoon is assumed to overspill/breach only at water levels > 4 masl) and the regime that triggered openings at water levels of 2.8 masl, simulation results indicate that chlorophyll *a* and macroalgae biomass would both increase, and *Ruppia* biomass decrease. In both simulations *Ruppia* biomass declined to very low levels (< 2 g C m^{-2}) and did not recover. It should be noted that there were no openings simulated in the “natural opening regime” scenario, as the water level did not reach 4 masl. In the “open at 2.8 masl” scenario there was only two periods of opening.

For the scenario that involved opening the lagoon for one month in spring each year, simulations suggested that chlorophyll *a* and macroalgae biomass would increase and *Ruppia* biomass would decrease, compared to the base scenario. In contrast to the “natural opening regime” and “open at 2.8 masl” scenarios, however, *Ruppia* biomass increased again over the final few years of the simulation (coincident with a lower macroalgae biomass). Opening the lagoon for three months in winter each year resulted in *Ruppia* and macroalgae biomass and chlorophyll *a* similar to the base scenario. *Ruppia* biomass was decreased slightly in the variable winter opening scenario, compared to the regular three month winter opening. The scenario that opened the lagoon for six months over summer each year had reduced macroalgae biomass and chlorophyll *a*, whilst *Ruppia* biomass was stable but did not include any summer biomass peaks which were evident in the latter portion of the base scenario and in the scenarios with winter openings.

Three climate scenarios were each run for one year (using actual meteorological and inflow input data for that year, determined to have average, high or low rainfall), and so are not directly comparable with other scenarios or the base scenario (which cover 10 years). Total suspended solids and chlorophyll *a* were increased, and macroalgae biomass decreased, during late summer/autumn of the “dry year” simulation compared to with the “average year” and “wet year”. *Ruppia* biomass was lower in the “dry year” and higher in the “wet year” compared with the average year.

3.3.2 Nutrient and sediment reduction scenarios

Nutrient and sediment scenarios that reduced only nutrients or nutrient and sediments in freshwater inflows by 10% had very little effect on any of the modelled variables, including nutrient concentrations and primary producer biomass in the lagoon (Figure 27 and Figures 31-34). The scenario that reduced phosphorus and sediment concentrations in freshwater inflows by 25% yielded slightly reduced phosphorus and chlorophyll *a*, but slightly increased macroalgae and little change in *Ruppia* biomass, compared to the base scenario. A 25% reduction in both nutrients (i.e. nitrogen and phosphorus) marginally reduced both chlorophyll *a* and macroalgae biomass, and increased *Ruppia* biomass slightly. A 25% reduction in nitrogen and 50% reduction in phosphorus reduced chlorophyll *a*, but slightly increased macroalgae biomass and had little effect on *Ruppia* biomass. In contrast, a 50% reduction in nitrogen and 25% reduction in phosphorus had a substantial effect on modelled variables, reducing both macroalgae biomass and chlorophyll *a* and increasing *Ruppia* biomass. A 50% reduction in both nutrients reduced chlorophyll *a*; macroalgae, although decreased compared to the baseline scenario, were increased compared to the 50% reduction in nitrogen and 25% reduction in phosphorus. A 90% reduction in both nutrients reduced both macroalgae biomass and chlorophyll *a* to low levels (i.e. < 1 g C m^{-2} macroalgae and < 1 µg L^{-1} chlorophyll *a*) and more than doubled *Ruppia* biomass compared to the base scenario.

3.3.3 Combined scenarios

Scenarios involving alterations to the opening regime as well as reductions in nutrient loading had comparably substantial effects on modelled variables (Figures 28-29 and Figures 31-34). With the "natural" opening regime (i.e. no openings) and 50% reduction in nitrogen and phosphorus, *Ruppia* biomass was increased and chlorophyll *a* was reduced compared to the base scenario and natural opening regime with no nutrient reduction; macroalgae biomass was increased compared to the base scenario and decreased compared to the natural opening regime with no nutrient reduction). Additional reductions in nutrient loading (i.e. 70 and 90%) further reduced macroalgae biomass and chlorophyll *a* and increased *Ruppia* biomass substantially compared to the base scenario and natural opening regime with no nutrient reduction.

A three-month winter opening, 50% reduction in nitrogen and 25% reduction in phosphorus resulted in reduced macroalgae and chlorophyll *a*, and a roughly 2-fold increase in *Ruppia* biomass compared with the base scenario or the three-month opening alone. 50% reduction in both nutrients combined with the three month opening further reduced chlorophyll *a* but not macroalgae. A 90% reduction in both nutrients combined with the three month opening resulted in low macroalgae and chlorophyll *a*, and high *Ruppia* biomass.

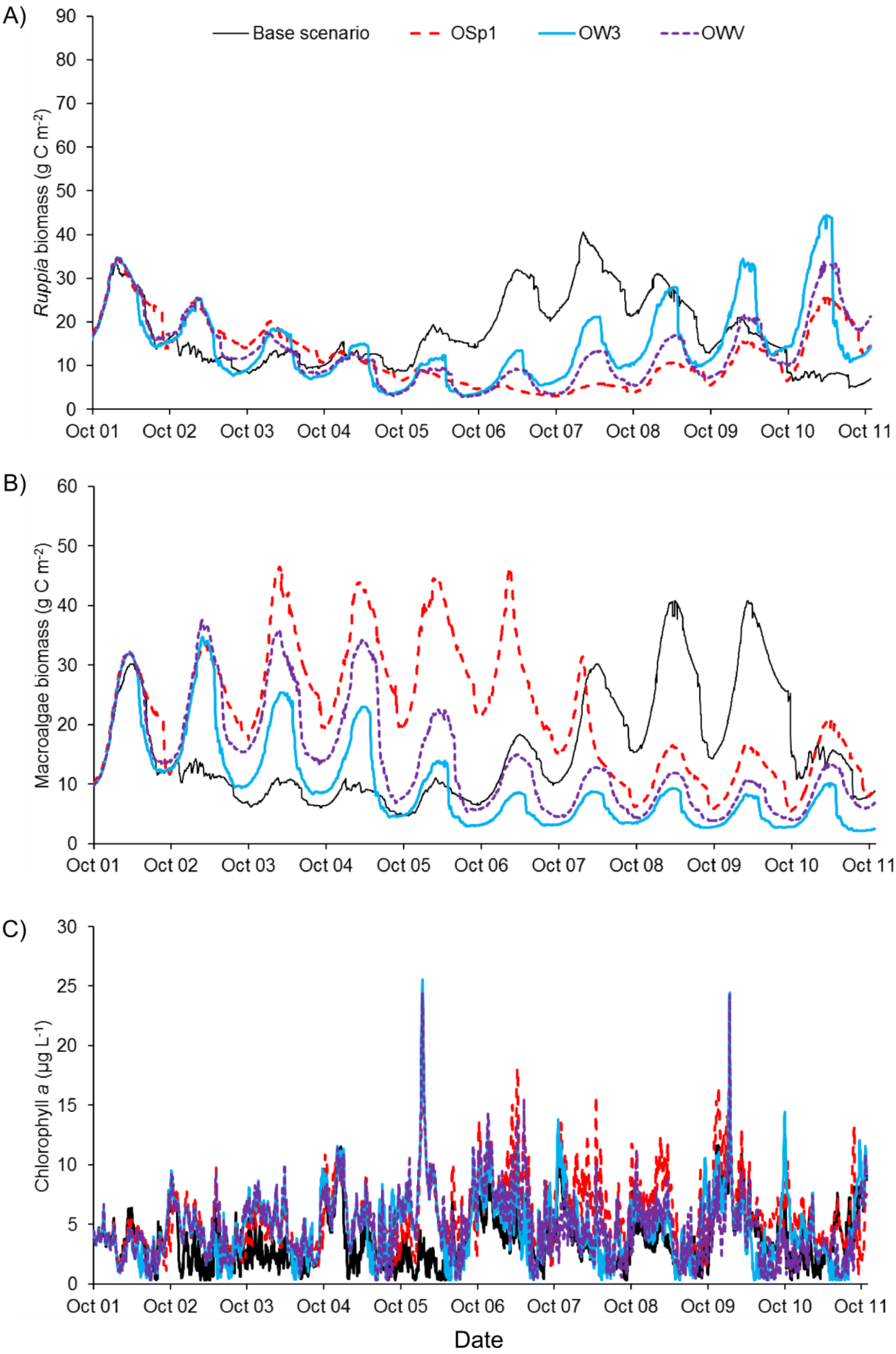


Figure 24: Effect of hydrological scenarios (OSp1 = short spring opening, OW3 = 3 month winter opening, OWV = variable winter opening) on A) *Ruppia* biomass B) macroalgae biomass and C) chlorophyll *a* concentration.

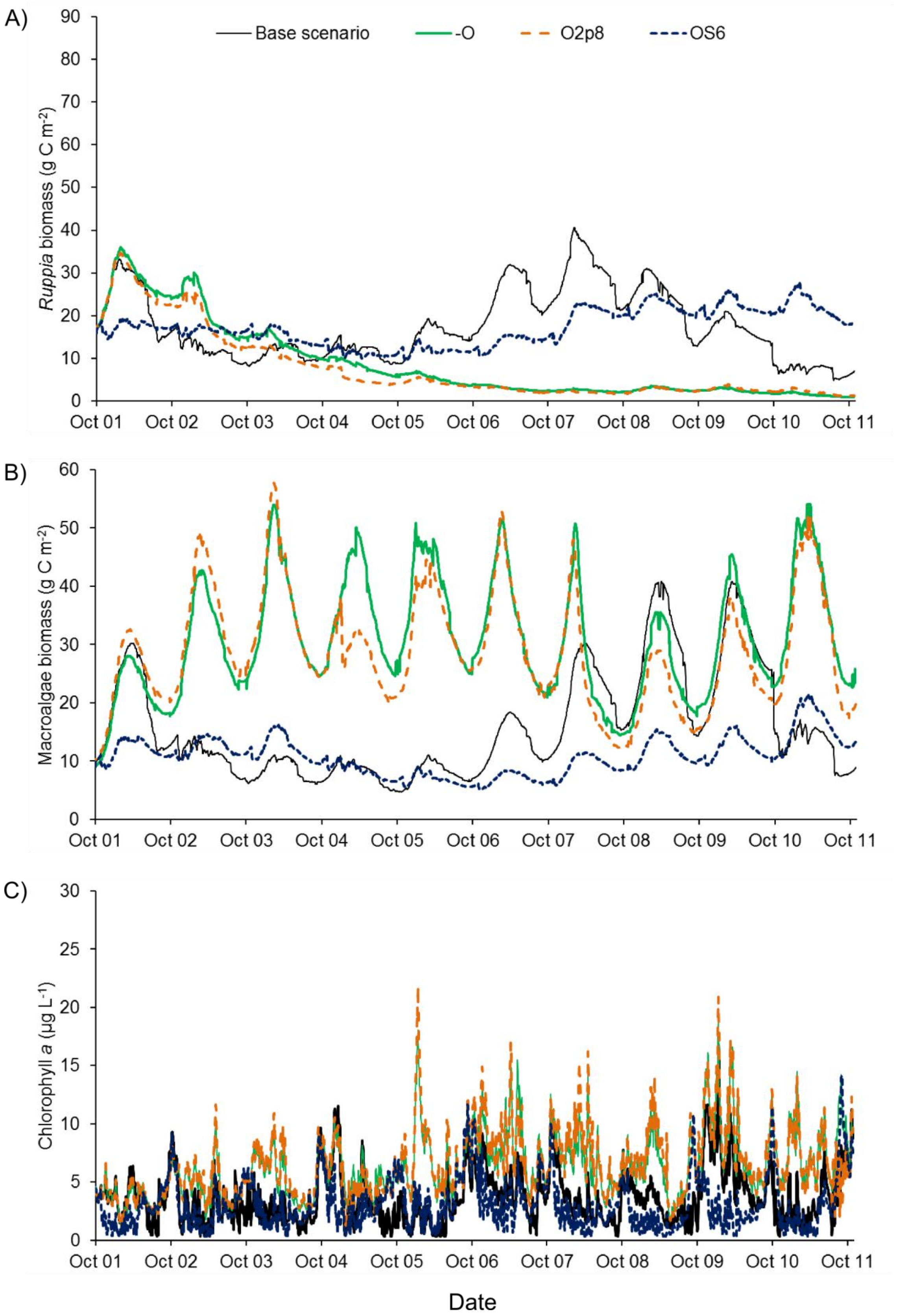


Figure 25: Effect of hydrological scenarios (-O = natural opening regime, O2p8 = opens when water level reaches 2.8 masl, OS6 = long summer opening) on A) *Ruppia* biomass B) macroalgae biomass and C) chlorophyll *a* concentration.

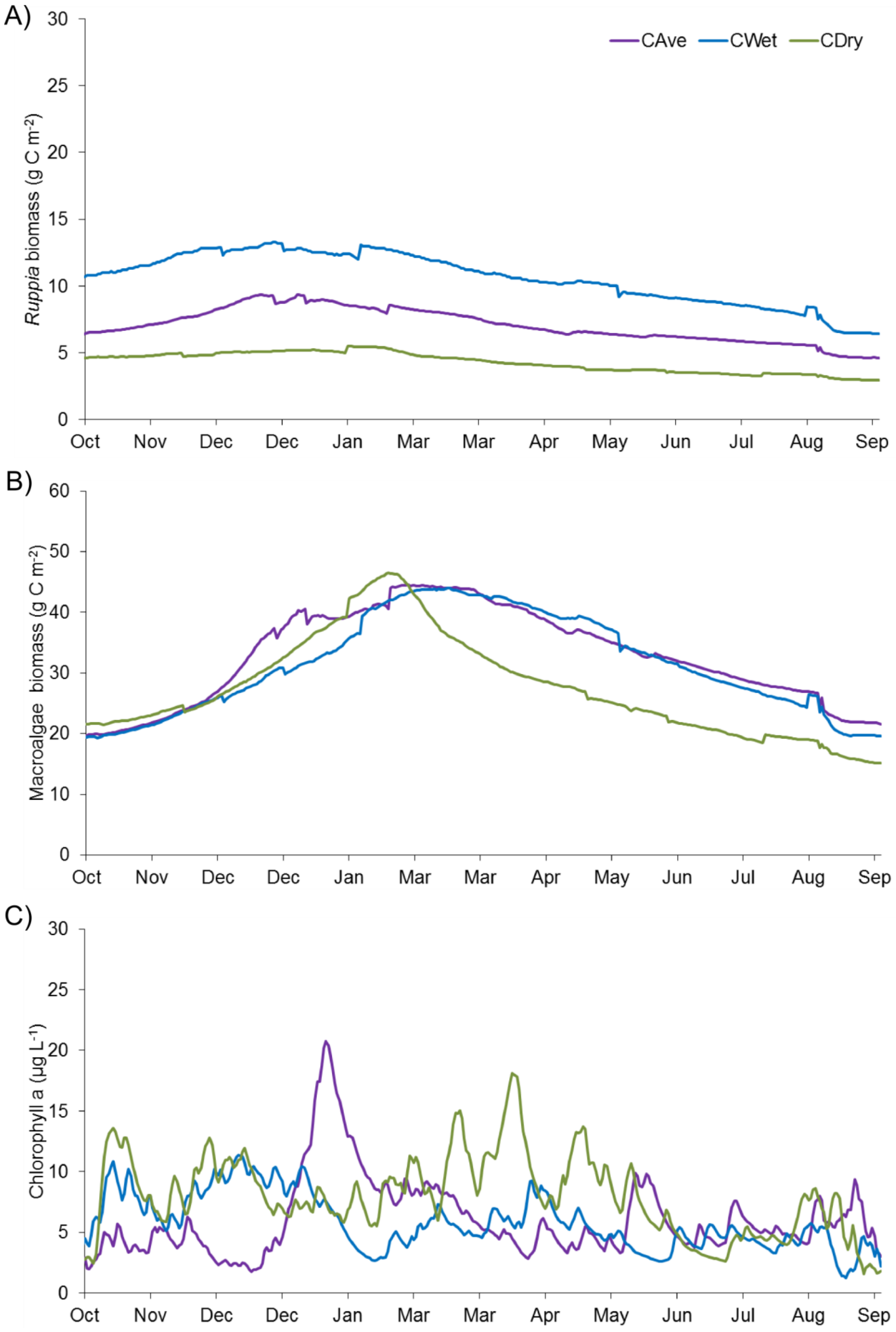


Figure 26: Effect of climate scenarios (CAve = "average rainfall" year, CWet = "wet" year, CDry = "dry" year) on A) *Ruppia* biomass, B) macroalgae biomass and C) chlorophyll *a* concentration.

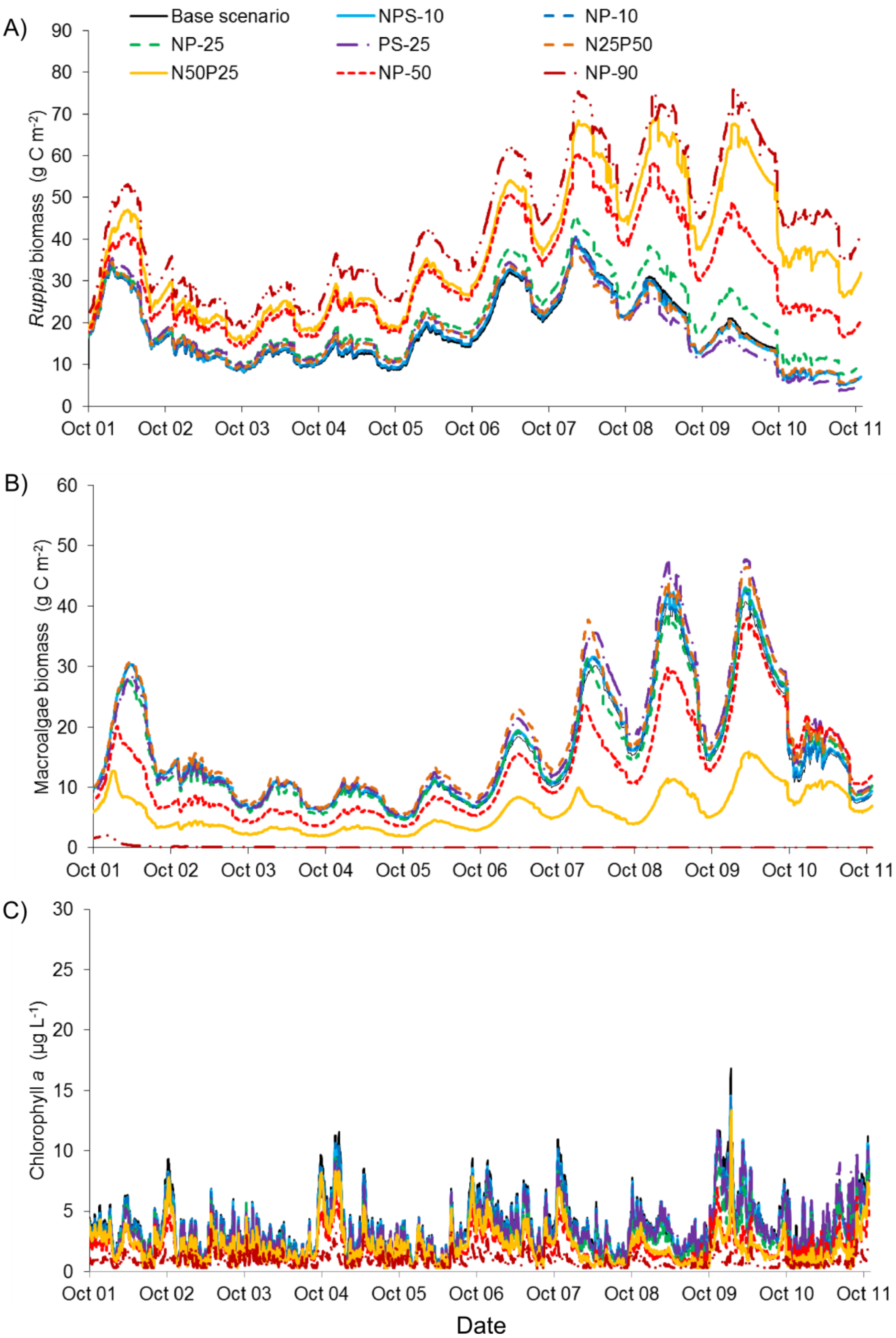


Figure 27: Effect of scenarios involving nutrient and sediment reductions in freshwater inflows (NP-10 = 10% decrease in "non-natural" nitrogen and phosphorus, NPS-10 = 10% decrease in nitrogen, phosphorus and suspended sediments, NP-25 = 25% decrease in nitrogen and phosphorus, PS-25 = 25% decrease in phosphorus and suspended sediments, N25P50 = 25% decrease in nitrogen and 50% decrease in phosphorus, N50P25 = 50% decrease in nitrogen and 25% decrease in phosphorus, NP-50 = 50% decrease in nitrogen and phosphorus, NP-90 = 90% decrease in nitrogen and phosphorus) on A) *Ruppia* biomass, B) macroalgae biomass, and C) chlorophyll *a* concentration.

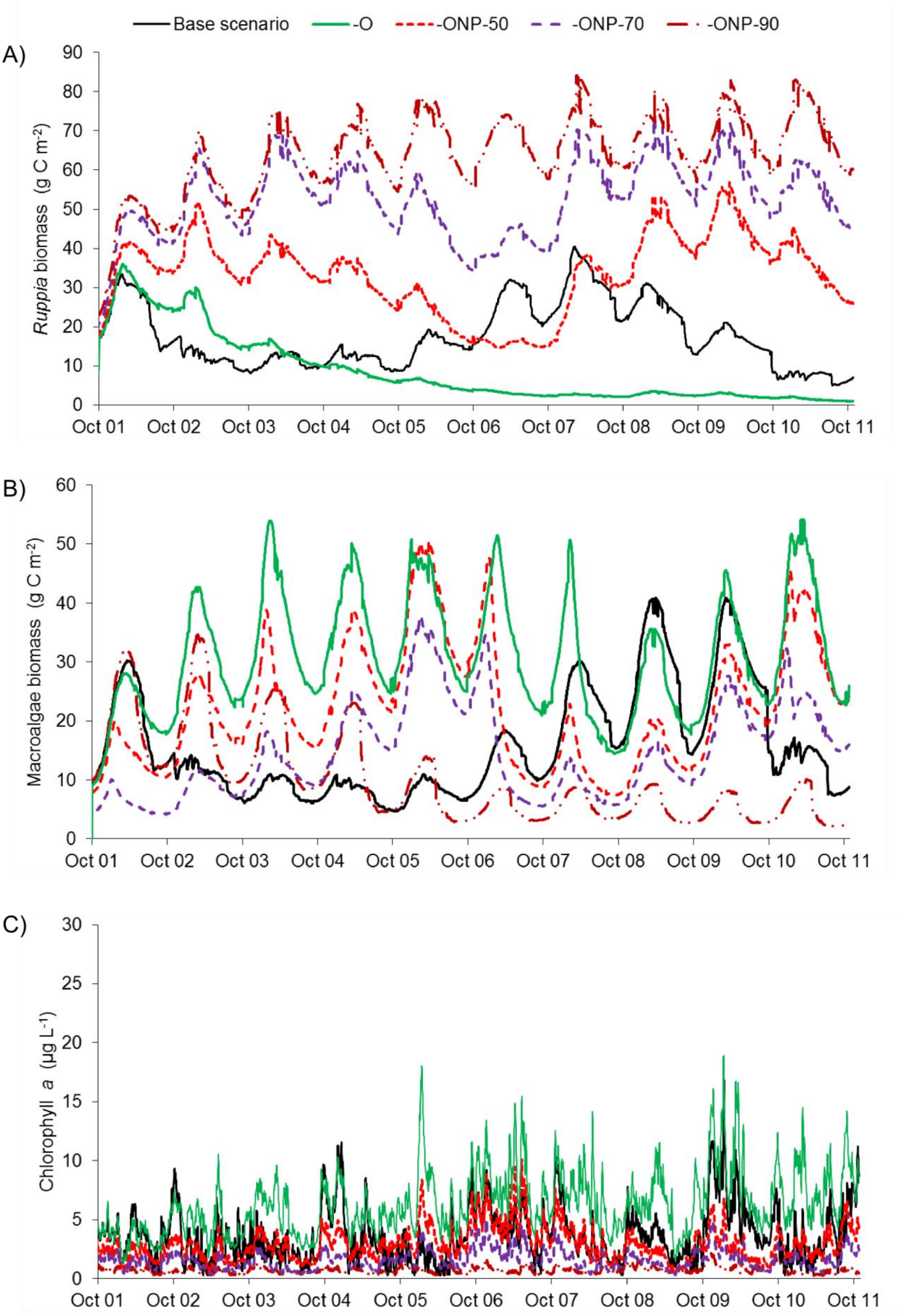


Figure 28: Effect of combined hydrological and nutrient reduction scenarios (-ONP-50 = natural opening regime and 50% reduction in nitrogen and phosphorus, -ONP-70 = natural opening regime and 70% reduction in nitrogen and phosphorus, -ONP-90 = natural opening regime and 90% reduction in nitrogen and phosphorus) on A) *Ruppia* biomass, B) macroalgae biomass, and C) chlorophyll *a* concentration. Effect of hydrological scenario alone (i.e. -0 = natural opening regime) also shown for comparison.

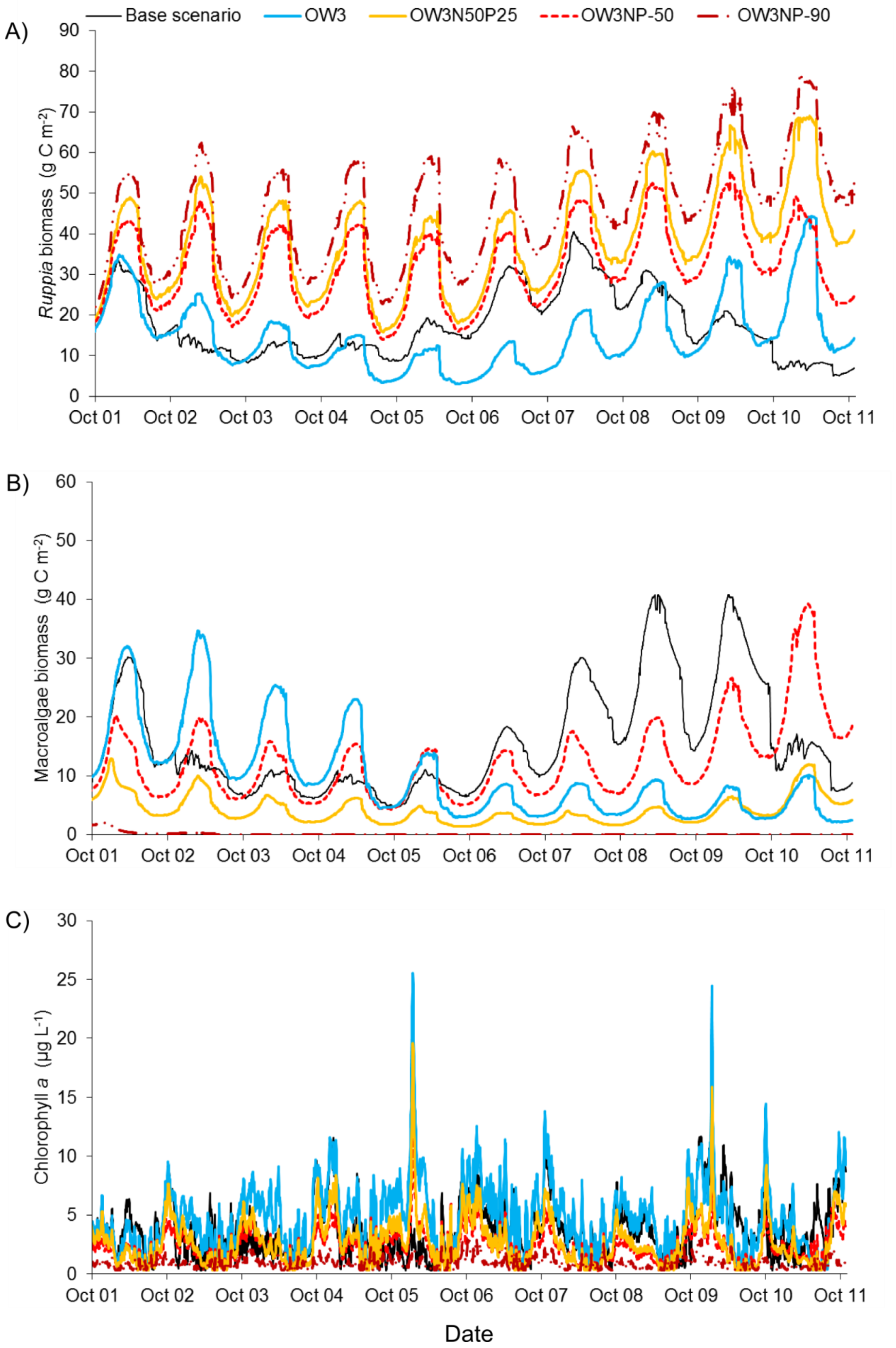
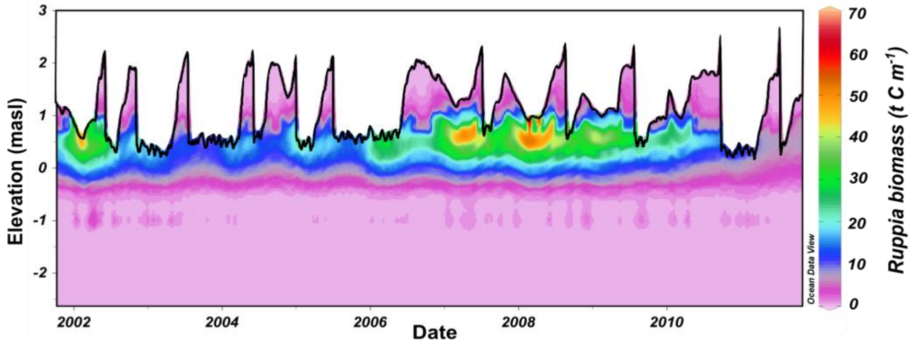
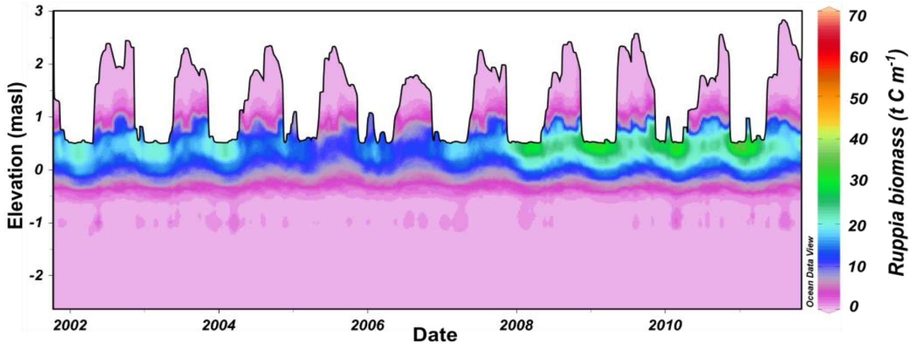


Figure 29: Effect of combined hydrological and nutrient reduction scenarios (OW3N50P25 = 3 month winter opening and 50% reduction in nitrogen and 25% reduction in phosphorus, OW3NP-50 = 3 month winter opening and 50% reduction in nitrogen and phosphorus, OW3NP-90 = 3 month winter opening and 90% reduction in nitrogen and phosphorus) on A) *Ruppia* biomass, B) macroalgae biomass, and C) chlorophyll *a* concentration. Effect of hydrological scenario alone (i.e. OW3 = 3 month winter opening) also shown for comparison.

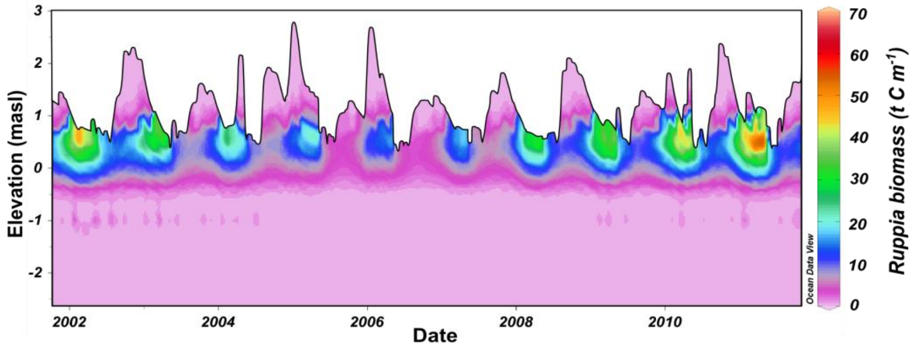
A) Base scenario (calibrated/validated simulation)



B) Scenario: OS6 (Summer opening)



C) Scenario: OW3 (Winter opening)



D) Scenario: OW3N50P25 (Winter opening combined with 50% reduction in nitrogen and 25% reduction in phosphorus loading)

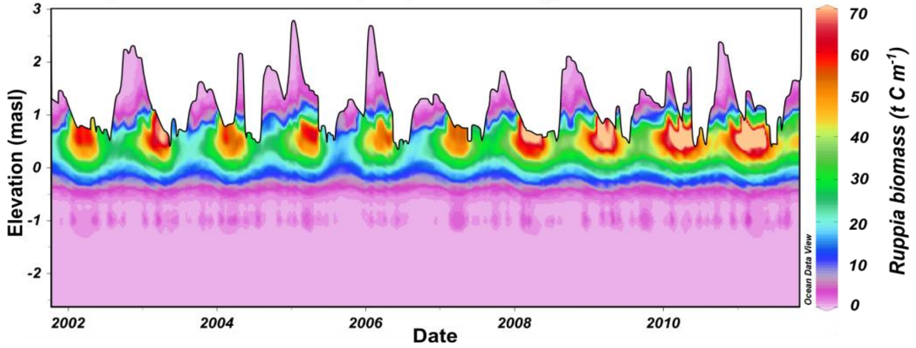
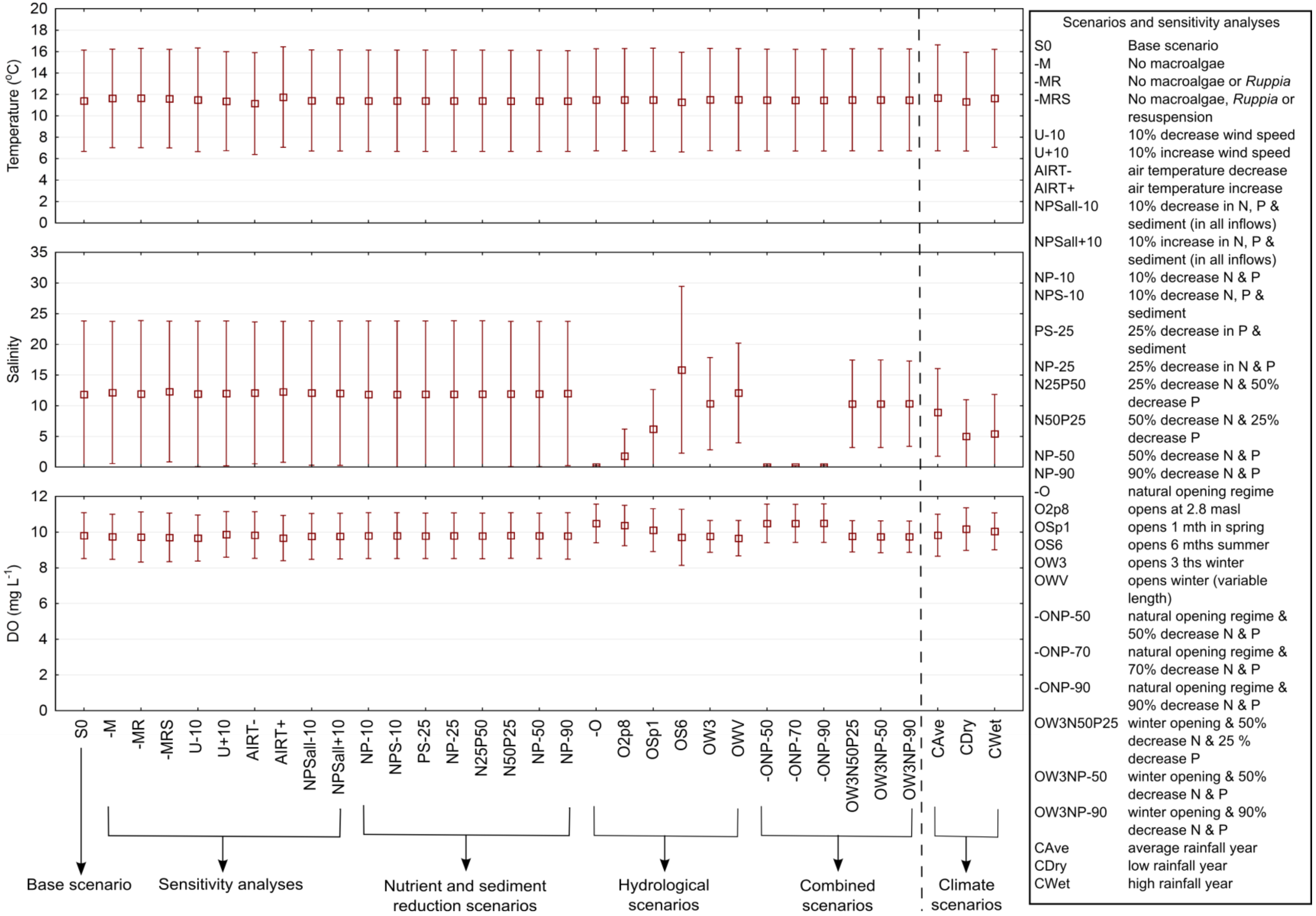


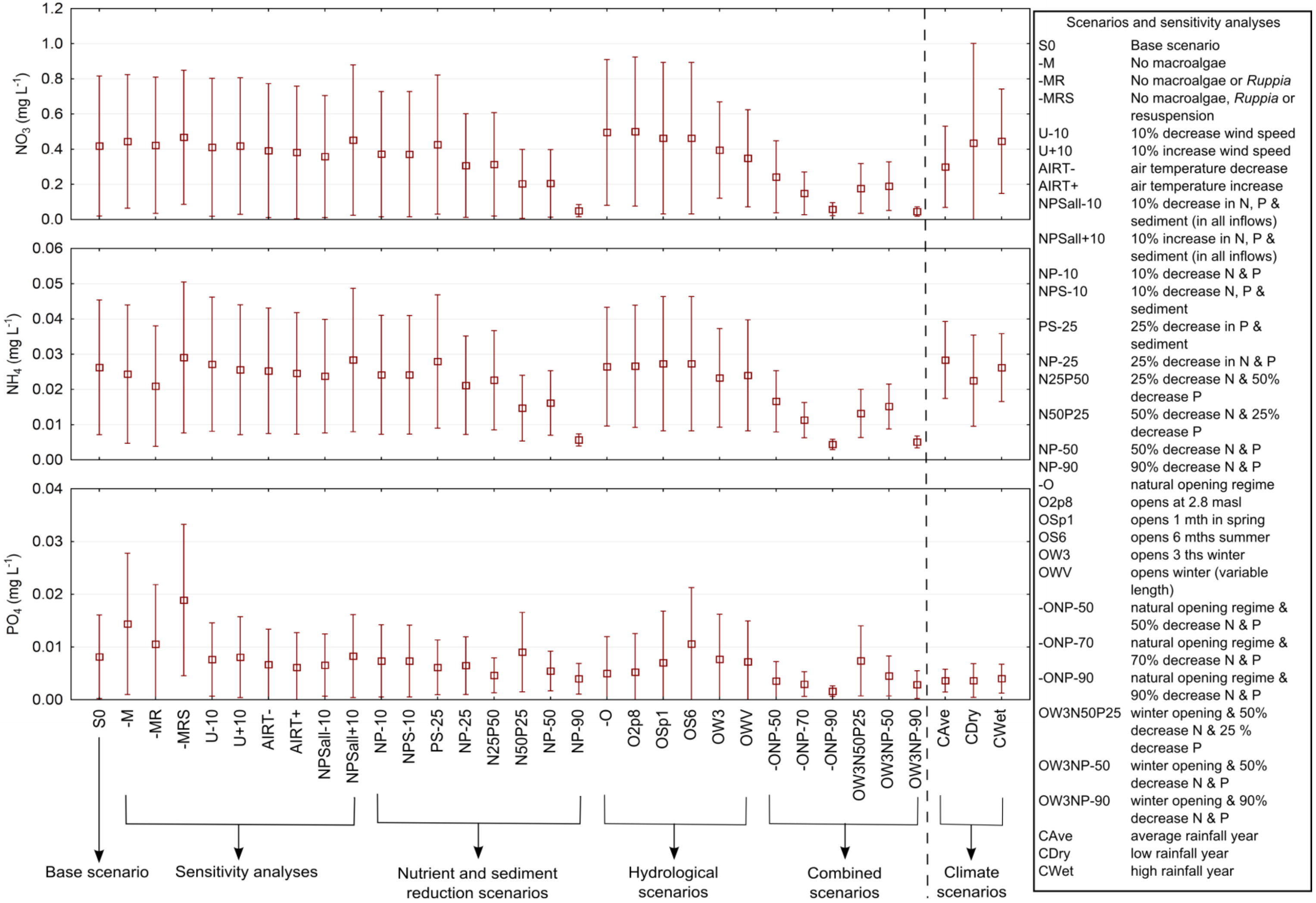
Figure 30: Modelled (DYRESM-CAEDYM) *Ruppia* biomass as a function of elevation (masl) for A) the base scenario (i.e. calibration/validation period), hydrological scenarios B) OS6 (long summer opening), and C) OW3 (winter opening), and D) combined scenario OW3N50P25 (winter opening and50% reduction in nitrogen loads and 25% reduction in phosphorus loads).



Scenarios and sensitivity analyses

S0	Base scenario
-M	No macroalgae
-MR	No macroalgae or *Ruppia*
-MRS	No macroalgae, *Ruppia* or resuspension
U-10	10% decrease wind speed
U+10	10% increase wind speed
AIRT-	air temperature decrease
AIRT+	air temperature increase
NPSall-10	10% decrease in N, P & sediment (in all inflows)
NPSall+10	10% increase in N, P & sediment (in all inflows)
NP-10	10% decrease N & P
NPS-10	10% decrease N, P & sediment
PS-25	25% decrease in P & sediment
NP-25	25% decrease in N & P
N25P50	25% decrease N & 50% decrease P
N50P25	50% decrease N & 25% decrease P
NP-50	50% decrease N & P
NP-90	90% decrease N & P
-O	natural opening regime
O2p8	opens at 2.8 masl
OSp1	opens 1 mth in spring
OS6	opens 6 mths summer
OW3	opens 3 ths winter
OWV	opens winter (variable length)
-ONP-50	natural opening regime & 50% decrease N & P
-ONP-70	natural opening regime & 70% decrease N & P
-ONP-90	natural opening regime & 90% decrease N & P
OW3N50P25	winter opening & 50% decrease N & 25 % decrease P
OW3NP-50	winter opening & 50% decrease N & P
OW3NP-90	winter opening & 90% decrease N & P
CAve	average rainfall year
CDry	low rainfall year
CWet	high rainfall year

Figure 31: Mean ± standard deviation for temperature, salinity and DO (dissolved oxygen) for the base scenario (i.e. the calibrated model; S0), sensitivity analyses and scenarios. N.B. Climate scenarios (one year simulations; CAve, CDry and CWet) are not directly comparable with other scenarios/sensitivity analyses (10 year simulations).



Scenarios and sensitivity analyses

S0	Base scenario
-M	No macroalgae
-MR	No macroalgae or *Ruppia*
-MRS	No macroalgae, *Ruppia* or resuspension
U-10	10% decrease wind speed
U+10	10% increase wind speed
AIRT-	air temperature decrease
AIRT+	air temperature increase
NPSall-10	10% decrease in N, P & sediment (in all inflows)
NPSall+10	10% increase in N, P & sediment (in all inflows)
NP-10	10% decrease N & P
NPS-10	10% decrease N, P & sediment
PS-25	25% decrease in P & sediment
NP-25	25% decrease in N & P
N25P50	25% decrease N & 50% decrease P
N50P25	50% decrease N & 25% decrease P
NP-50	50% decrease N & P
NP-90	90% decrease N & P
-O	natural opening regime
O2p8	opens at 2.8 masl
OSp1	opens 1 mth in spring
OS6	opens 6 mths summer
OW3	opens 3 ths winter
OWV	opens winter (variable length)
-ONP-50	natural opening regime & 50% decrease N & P
-ONP-70	natural opening regime & 70% decrease N & P
-ONP-90	natural opening regime & 90% decrease N & P
OW3N50P25	winter opening & 50% decrease N & 25 % decrease P
OW3NP-50	winter opening & 50% decrease N & P
OW3NP-90	winter opening & 90% decrease N & P
CAve	average rainfall year
CDry	low rainfall year
CWet	high rainfall year

Figure 32: Mean ± standard deviation for PO_4-P, NO_3-N and NH_4-N for the base scenario (i.e. the calibrated model; S0), sensitivity analyses and scenarios. N.B. Climate scenarios (one year simulations; CAve, CDry and CWet) are not directly comparable with other scenarios/sensitivity analyses (10 year simulations).

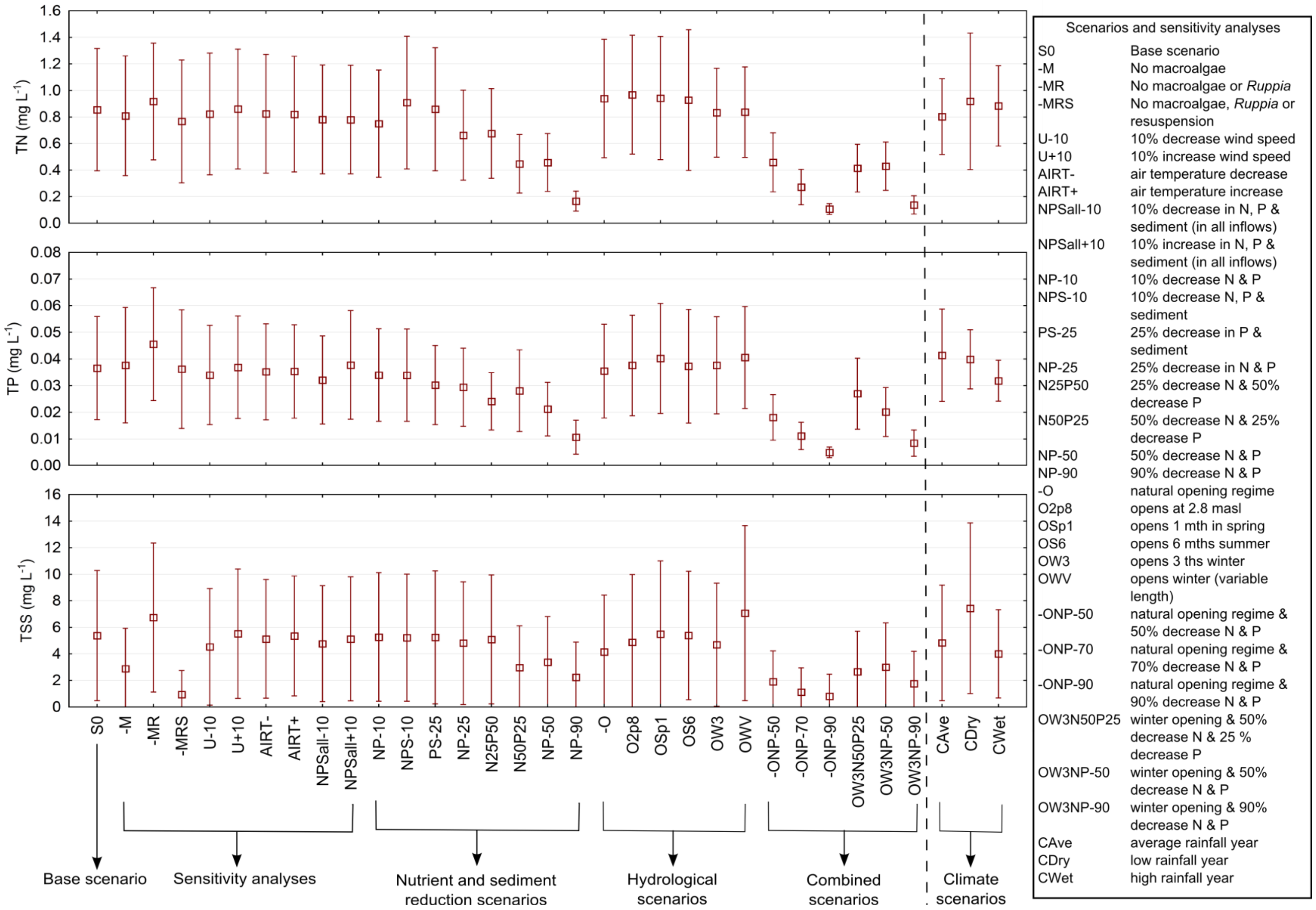


Figure 33: Mean ± standard deviation for TN (total nitrogen), TP (total phosphorus) and TSS (total suspended solids) for the base scenario (i.e. the calibrated model; S0), sensitivity analyses and scenarios. N.B. Climate scenarios (one year simulations; CAve, CDry and CWet) are not directly comparable with other scenarios/sensitivity analyses (10 year simulations).

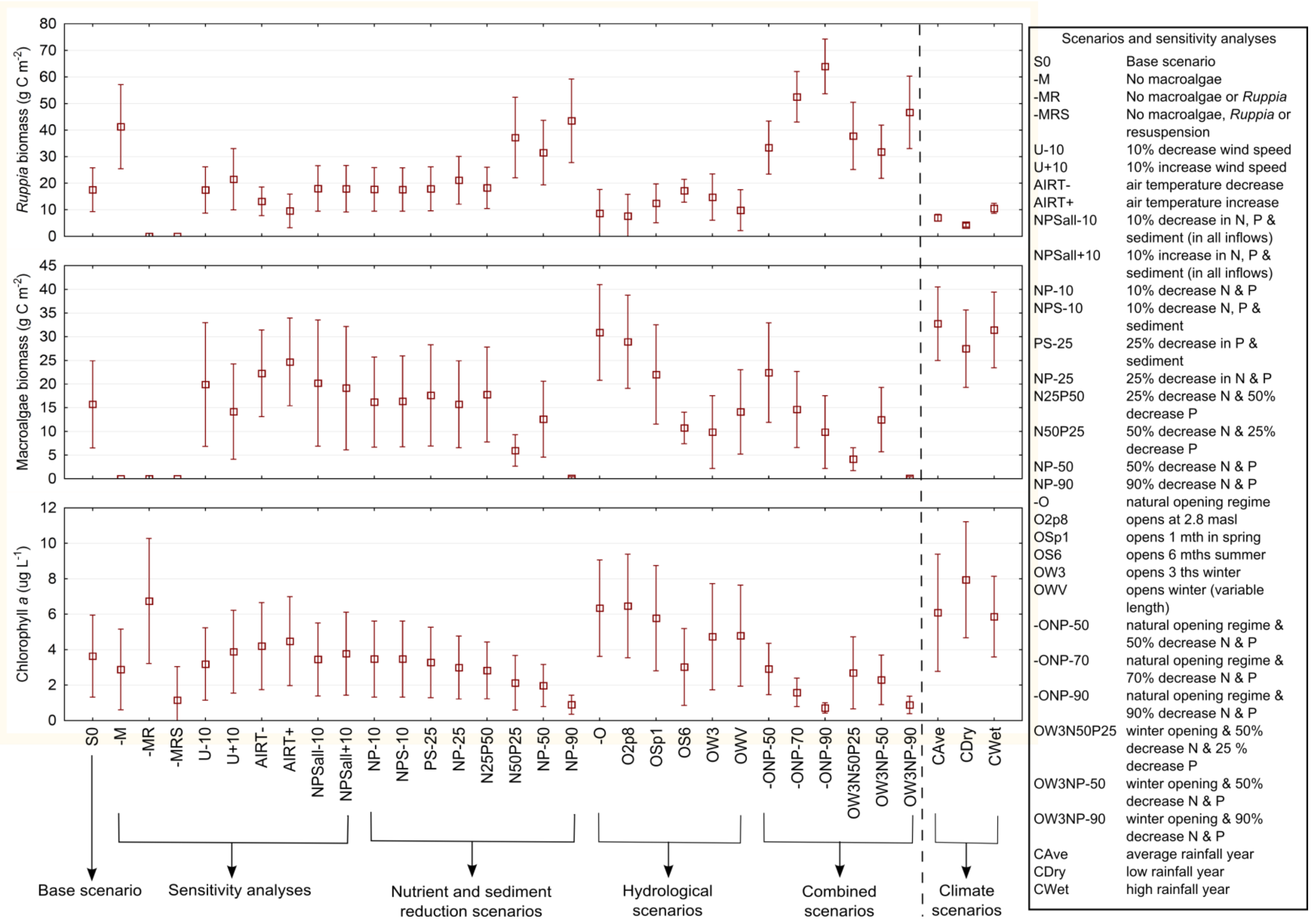


Figure 34: Mean ± standard deviation for *Ruppia* biomass, macroalgae biomass and chlorophyll *a* concentration for the base scenario (i.e. the calibrated model; S0), sensitivity analyses and scenarios. N.B. Climate scenarios (one year simulations; CAve, CDry and CWet) are not directly comparable with other scenarios/sensitivity analyses (10 year simulations).

3.4 ELCOM calibration

Visual comparison of modelled water level, temperature and salinity against field measurements at the Waghorn’s Rd water level recorder and four ES sampling sites is shown in Figures 35-37. Initial model runs using tide heights at Bluff for the open boundary resulted in modelled salinities significantly lower than field measurements. Because water level was not measured near the lagoon opening, salinities in the lagoon were calibrated by altering the level at the open boundary by an iterative process until modelled and measured salinities were matched as closely as possible. The overall model performance, assessed statistically using RMSE, suggests that the model was able to reproduce the lagoon water level to within c. 0.1 m and salinities to within c. 2 to 7 (Table 8). Modelled temperatures were consistently lower (by c. 3 to 4 ^{o}C), however, than field measurements.

ES has recently installed another water level recorder in the centre of the lagoon, allowing some comparison of the difference between time of high water at the centre site and Waghorn’s Rd recorder (when the lagoon is open). Although, there are no data for the same periods simulated using ELCOM in this study, the water level recorders suggest that tides at Waghorn’s Rd are c. 1-2 hours behind the centre site, and tidal amplitude is reduced at Waghorn’s compared to the centre site (e.g. Figure 38). This pattern appears to be reproduced in ELCOM output (e.g. Figure 39).

Table 8: Statistical comparison of ELCOM model simulations with field data (monthly measurements) of surface water in Waituna Lagoon, using Pearson correlation coefficient (R), root mean square error (RMSE), and normalised root mean square error (NRMSE), for water level (at Waghorn’s Rd recorder), salinity and temperature (at the four ES sampling sites, Centre, East, South and West).

Site/Year	Water level			Salinity			Water temperature		
	R	RMSE	NRMSE	R	RMSE	NRMSE	R	RMSE	NRMSE
2003/2004									
Waghorn’s Rd	0.99	0.10	0.12						
Lagoon Centre				0.92	4.96	0.22	0.92	3.34	0.35
Lagoon East				0.67	7.55	0.43	0.95	3.84	0.40
Lagoon South				0.92	4.74	0.20	0.89	3.48	0.35
Lagoon West				0.87	7.22	0.31	0.93	3.78	0.38
2007/2008									
Waghorn’s Rd	0.98	0.09	0.05						
Lagoon Centre				1.00	1.83	0.27	0.98	4.22	0.35
Lagoon East				0.92	1.89	0.46	0.96	4.95	0.39
Lagoon South				0.98	2.85	0.38	0.96	4.21	0.35
Lagoon West				0.94	2.36	0.37	0.96	4.41	0.35
2011									
Waghorn’s Rd	0.99	0.10	0.09						
Lagoon Centre				0.83	2.96	0.43	0.90	2.91	0.43
Lagoon East				0.86	4.30	0.45	0.84	2.94	0.45
Lagoon South				0.92	2.86	0.38	0.92	3.09	0.38
Lagoon West				0.54	3.54	0.55	0.89	3.23	0.55

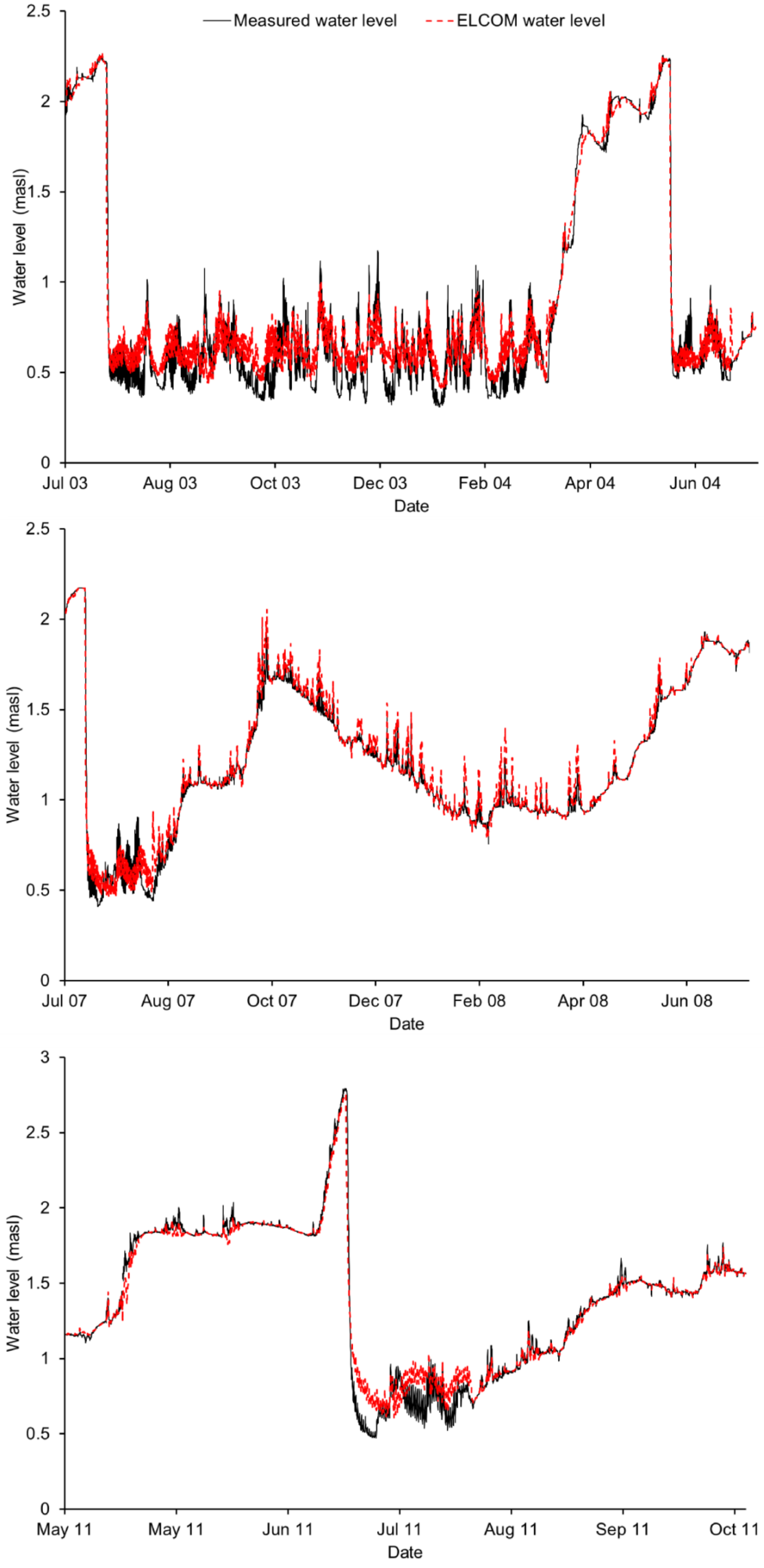


Figure 35: Simulated ELCOM water level (red dashed line) and measured water level at Waghorn's Rd (solid black line)

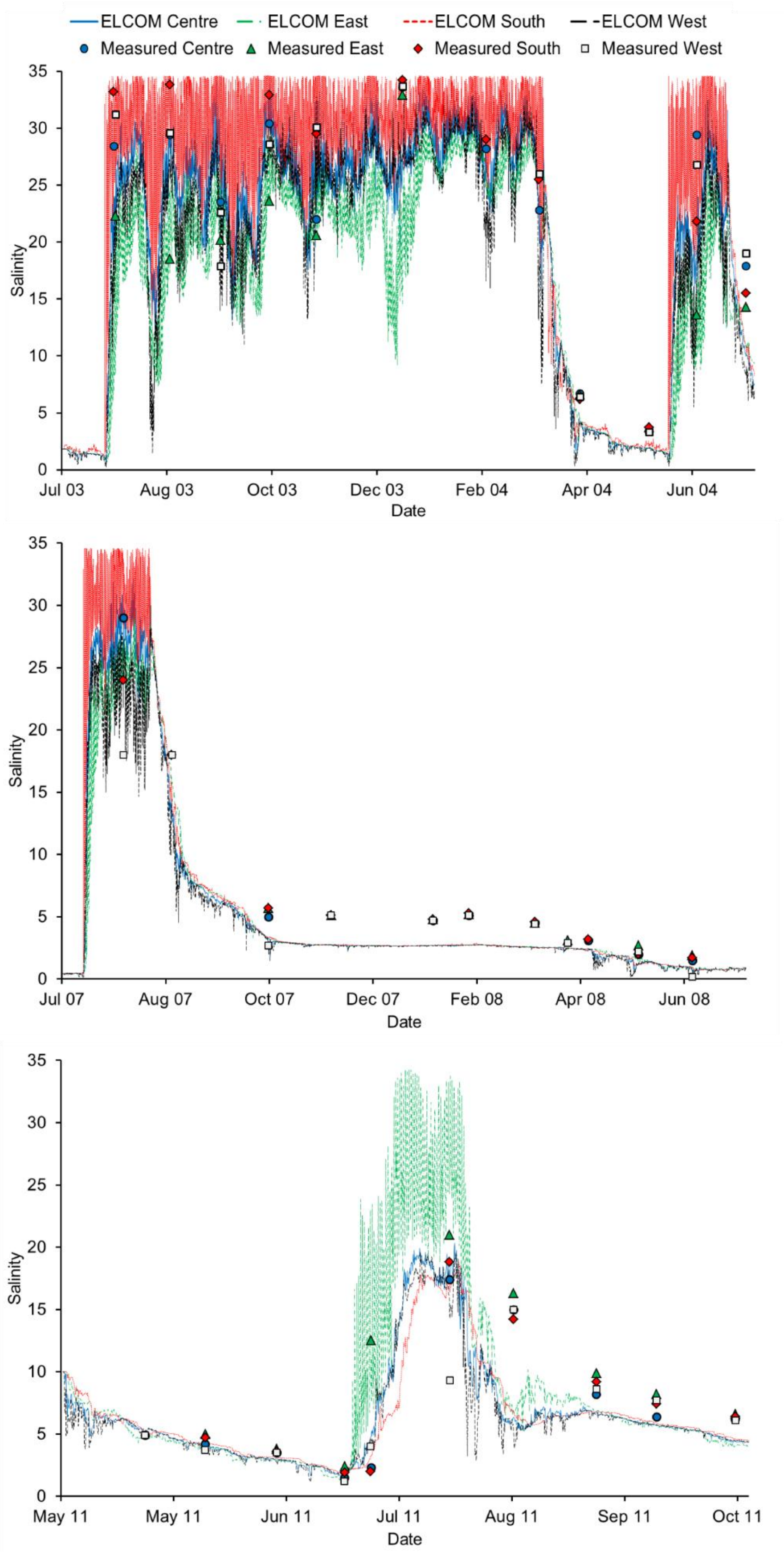


Figure 36: Simulated ELCOM salinity (lines) and measured salinity (symbols) at four ES sampling sites. Centre = blue/circle, East = green/triangle, South = red/diamond, West = white/square.

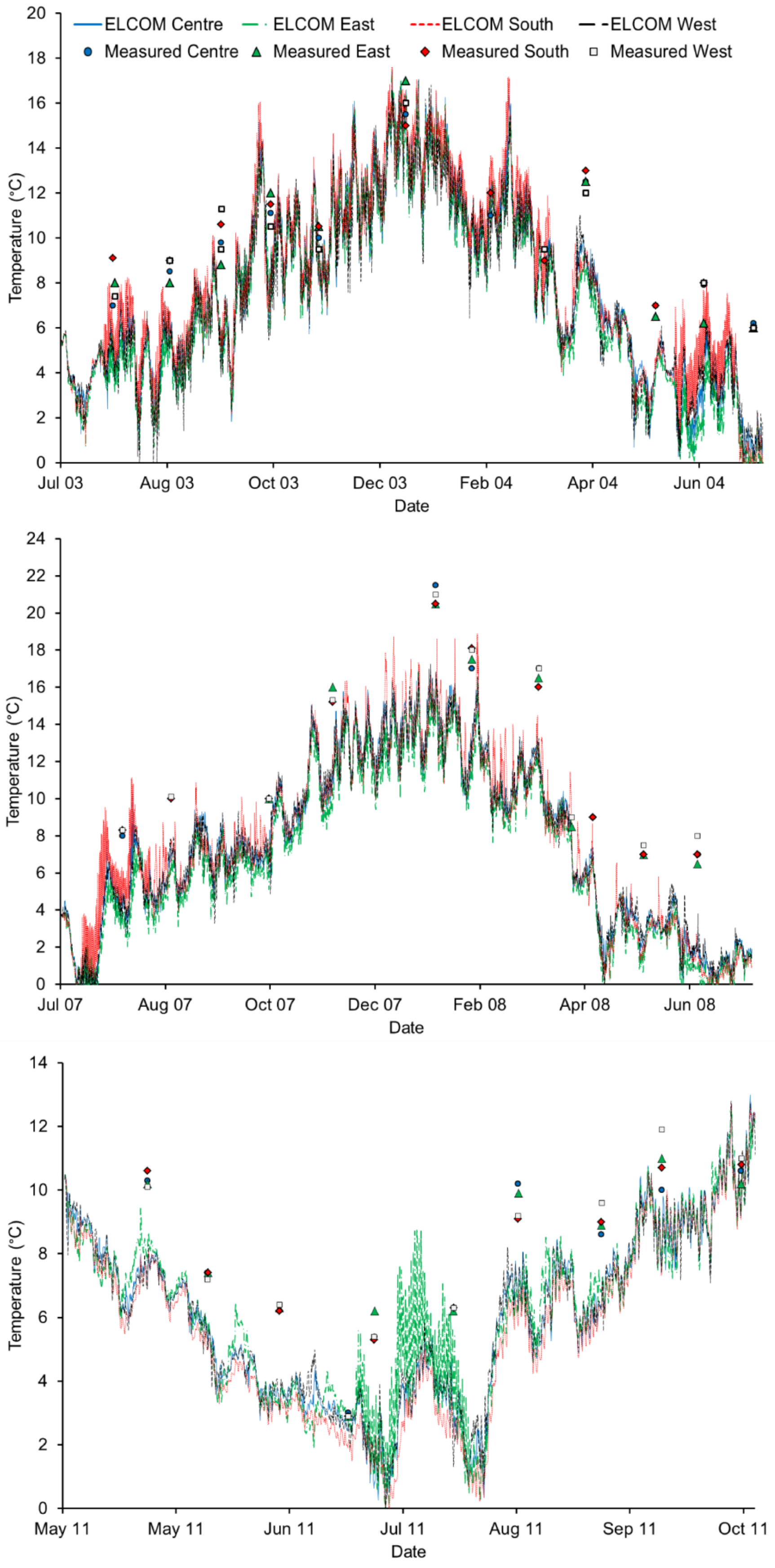


Figure 37: Simulated ELCOM temperature (lines) and measured temperature (symbols) at four ES sampling sites. Centre = blue/circle, East = green/triangle, South = red/diamond, West = white/square.

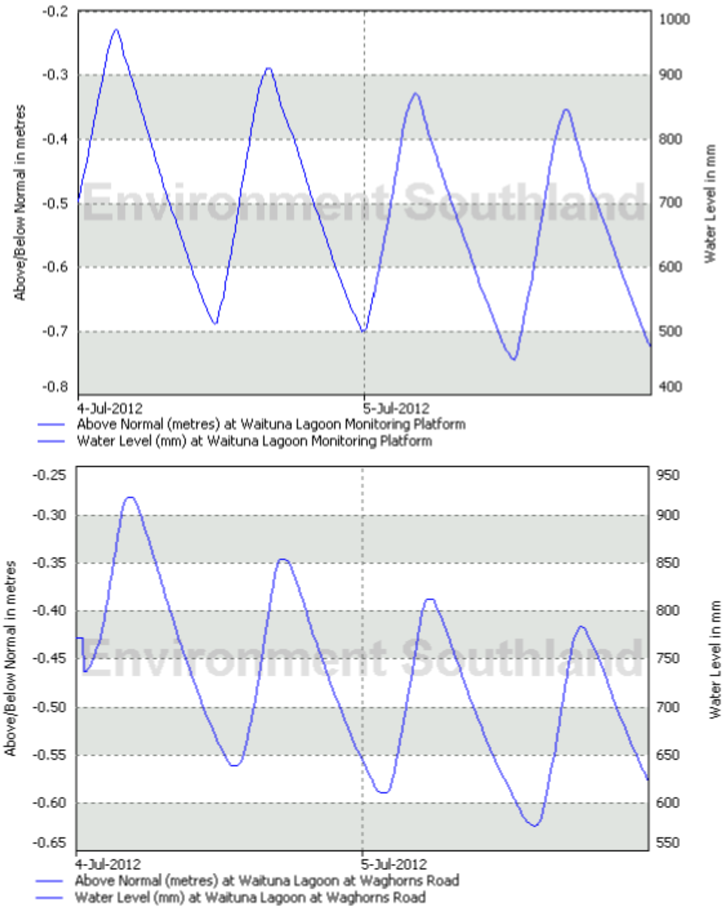


Figure 38: Measured water level at Waituna Lagoon centre site and Waghorn's Rd (source: http://www.es.govt.nz/rivers-and-rainfall)

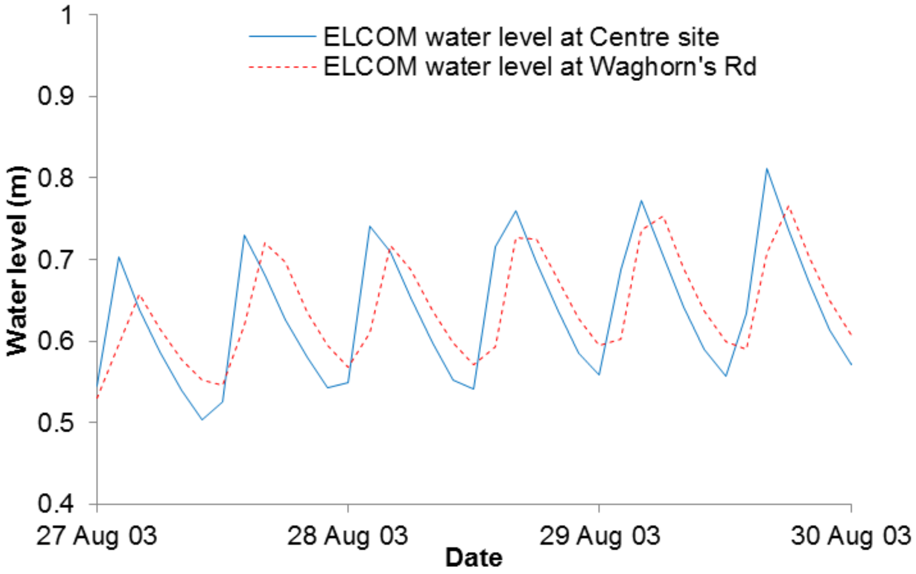


Figure 39: ELCOM-simulated water level at centre site and Waghorn's Rd site

3.5 ELCOM output

The frequent presence of wind setup (uneven water levels across the lagoon) is evident in the ELCOM simulations (e.g. Figures 40-41). When the lagoon is closed water levels tend to be higher at Waghorn’s Rd than at the West sampling site, with a maximum difference of 0.55 m.

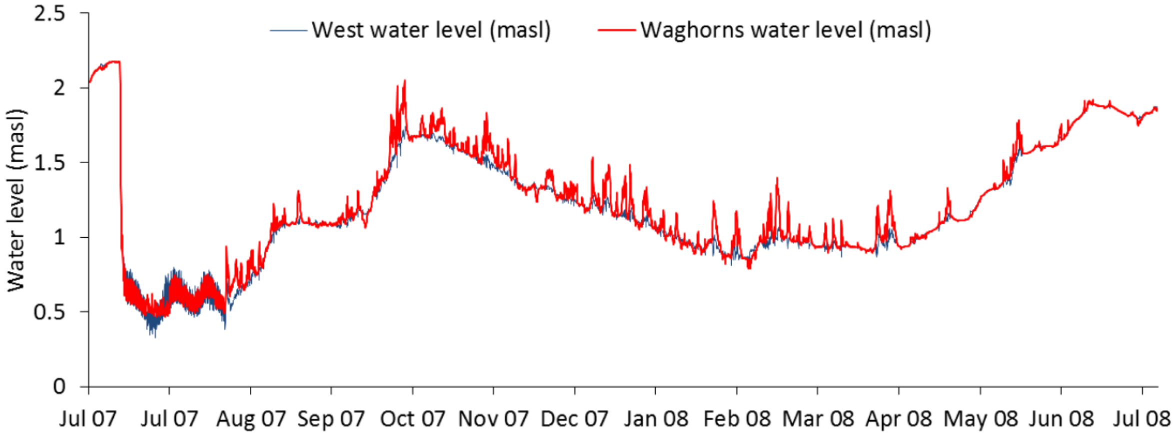


Figure 40: Simulated ELCOM water levels at Waghorn’s Rd (east end of lagoon) and at the West sampling site

Examination of ELCOM model output revealed that salinities may be quite variable across the lagoon, particularly between the main body of the lagoon and the eastern arm, varying with opening location, duration and spring/neap cycle. The simulations indicate that when the lagoon is opened at Walker’s Bay, salinities in the main body of the lagoon reach > 20 within a few days, but it may take a week or more for salinities to reach that level in the eastern arm. After the lagoon is opened salinity stratification is present in simulations in some areas, particularly in the deeper channels in the eastern arm (Figures 42-43). There were no data collected in the eastern arm with which to validate model output for this area of the lagoon during the simulation period. However, high resolution sampling (including sites in the eastern arm) was undertaken during and following an opening event (at Walker’s Bay) in July 2012 (Figure 44). Although the measured data is not directly comparable with ELCOM output, data from the July 2012 opening indicate that salinities in the main body of the lagoon (at the centre site) were between c. 20 and 28 within 3 days of opening, and that it took 2 weeks for salinity in the eastern arm to reach a maximum of c. 21 (although this ranged considerably (7-21) between sites in that part of the lagoon). Based on these data ELCOM may be underestimating salinities immediately following (1 – 2 days) an opening, otherwise the July 2012 data compares well with ELCOM simulations. The simulations indicate that, as for the field data, salinities at the centre site were between 20 and 25 within 2 to 4 days of opening, and that salinities in the eastern arm were c. 20 within 16 days of opening.

Opening location also appears to significantly affect salinity in the lagoon (Figures 45-46). When the lagoon is opened at Charlie’s Bay, salinities in the deep channel in the eastern arm reach > 25 within a few tides, but salinities in the main body of the lagoon remain far lower (< 15) for several weeks. Maximum salinities tend to be higher with a Walker’s Bay opening for the Centre, South and West sampling sites (c. 32) compared with c. 18 for a Charlie’s Bay opening, but for the East sampling sites and in the eastern arm, salinity tends to be higher during a Charlie’s Bay opening (c. 32 compared with 25 for a Walker’s Bay opening).

Current velocities through the lagoon opening also vary with opening location, with velocities c. 1 – 2 m s^{-1} at Walker's Bay compared with 2 – 3 m s^{-1} at Charlie's Bay (Figure 47). Maximum current velocities (which occur immediately following an opening) also tend to be higher at Charlie's Bay (c. 6 m s^{-1}) than at Walker's Bay (c. 4 m s^{-1}).

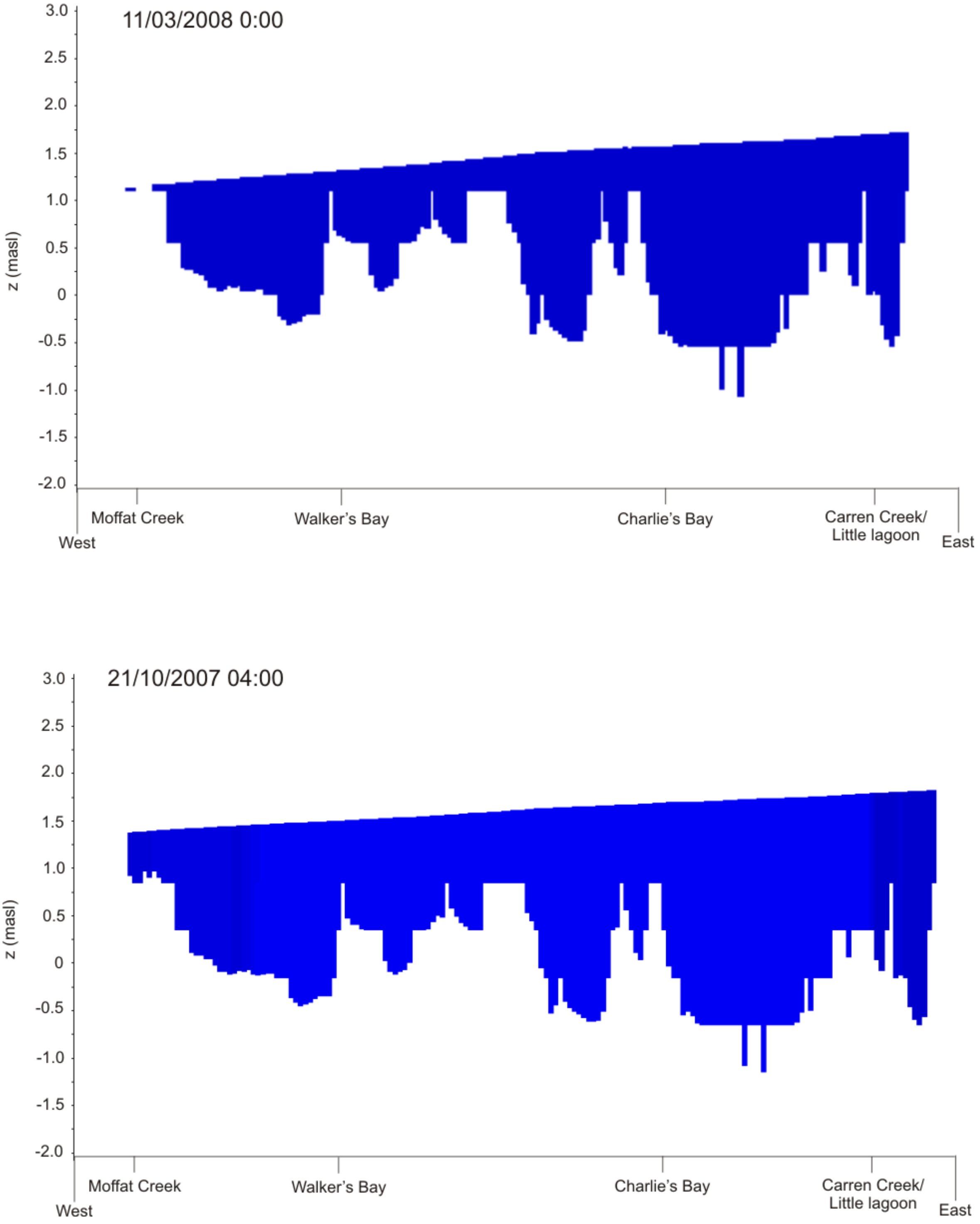


Figure 41: ELCOM-simulated water level (z; masl) in a west-east cross-section for two occasions when wind setup was strongly evident in the lagoon. Surface inflows and opening locations indicated on x axes. N.B. Lagoon was closed on both occasions.

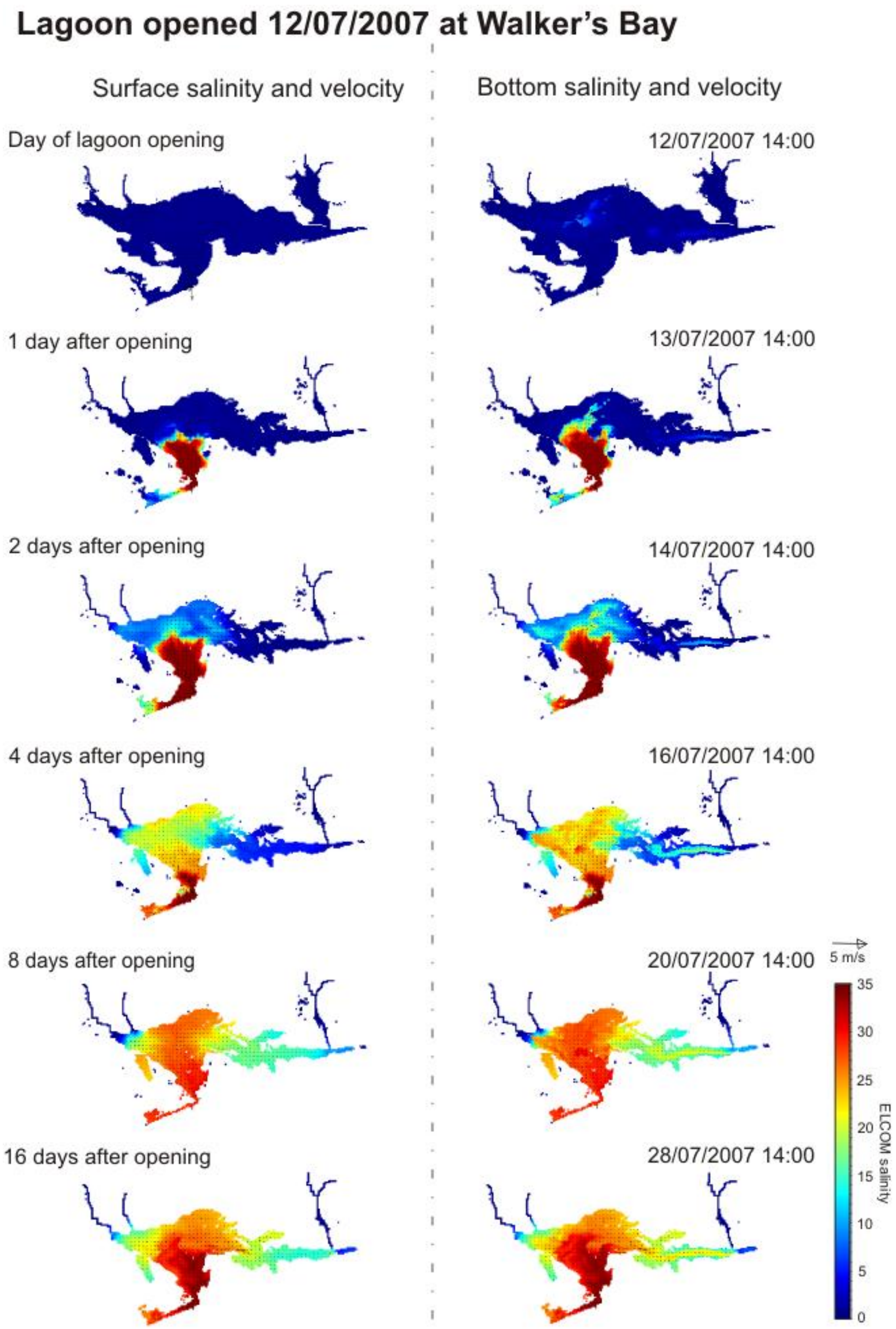
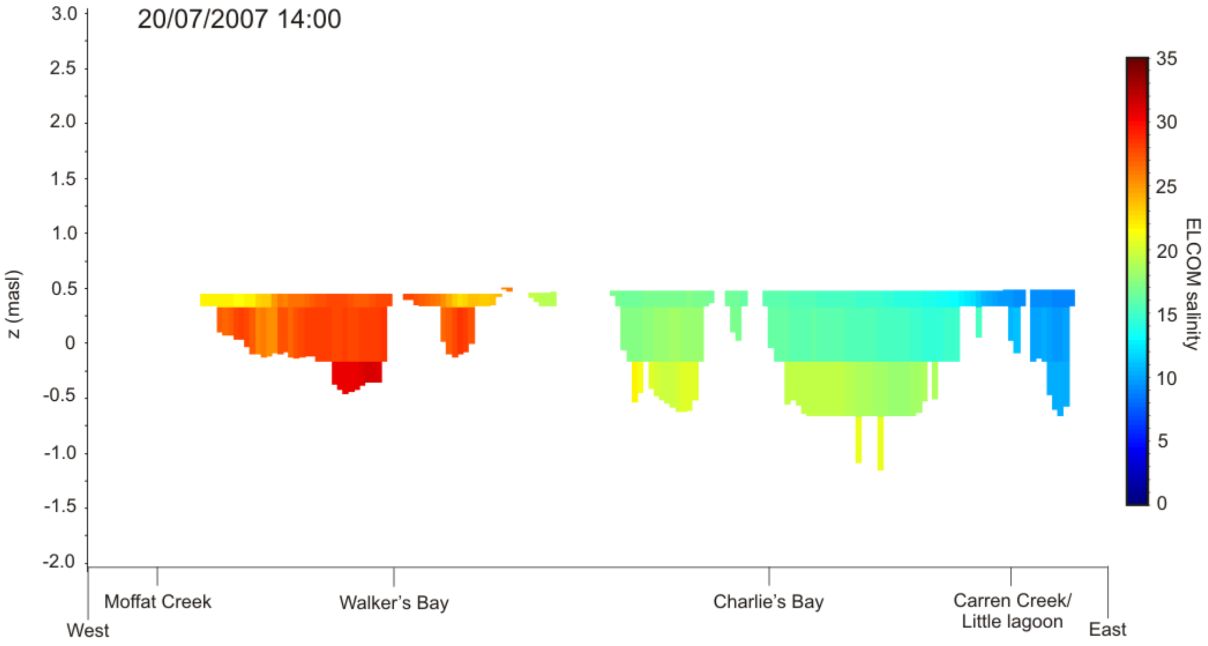


Figure 42: ELCOM-simulated salinity and current velocity on the day of, and in the days following, an opening at Walker's Bay on 12/07/2007. Please see the digital Appendix for 3D model animations.



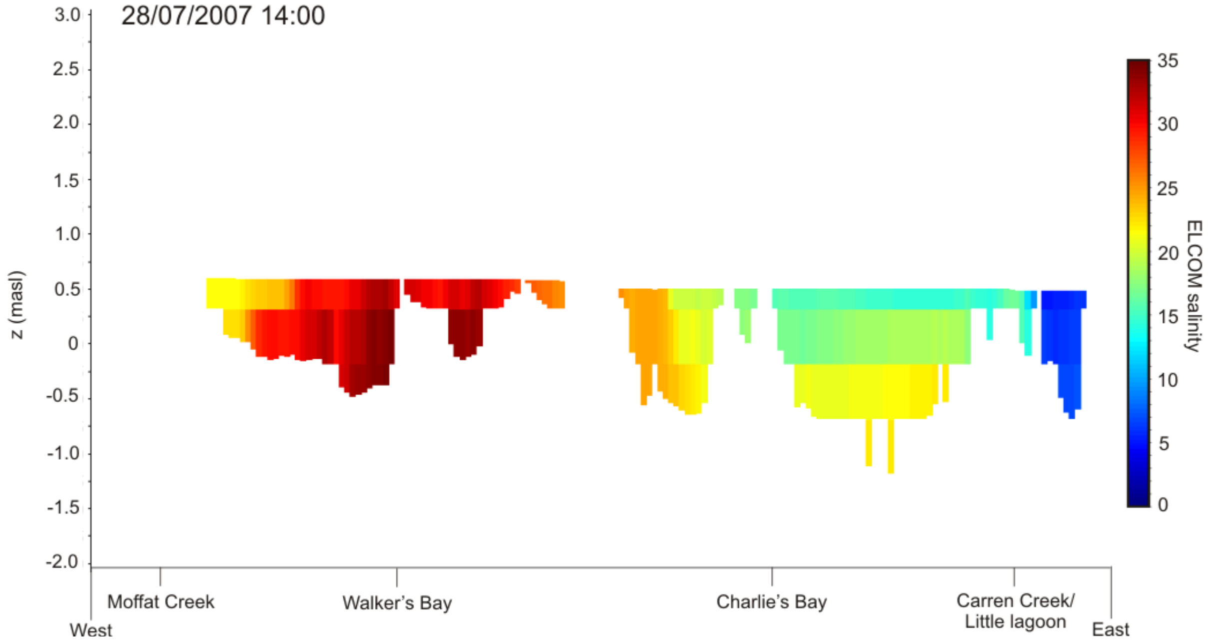


Figure 43: ELCOM-simulated water level (z; masl) and salinity in a west-east cross-section on two occasions following an opening at Walker’s Bay on 12/07/2007. Surface inflows and opening locations are indicated on x axes. N,B. These “curtains” (cross-sections) are discontinuous because areas that are above sea level are exposed when the lagoon is open.

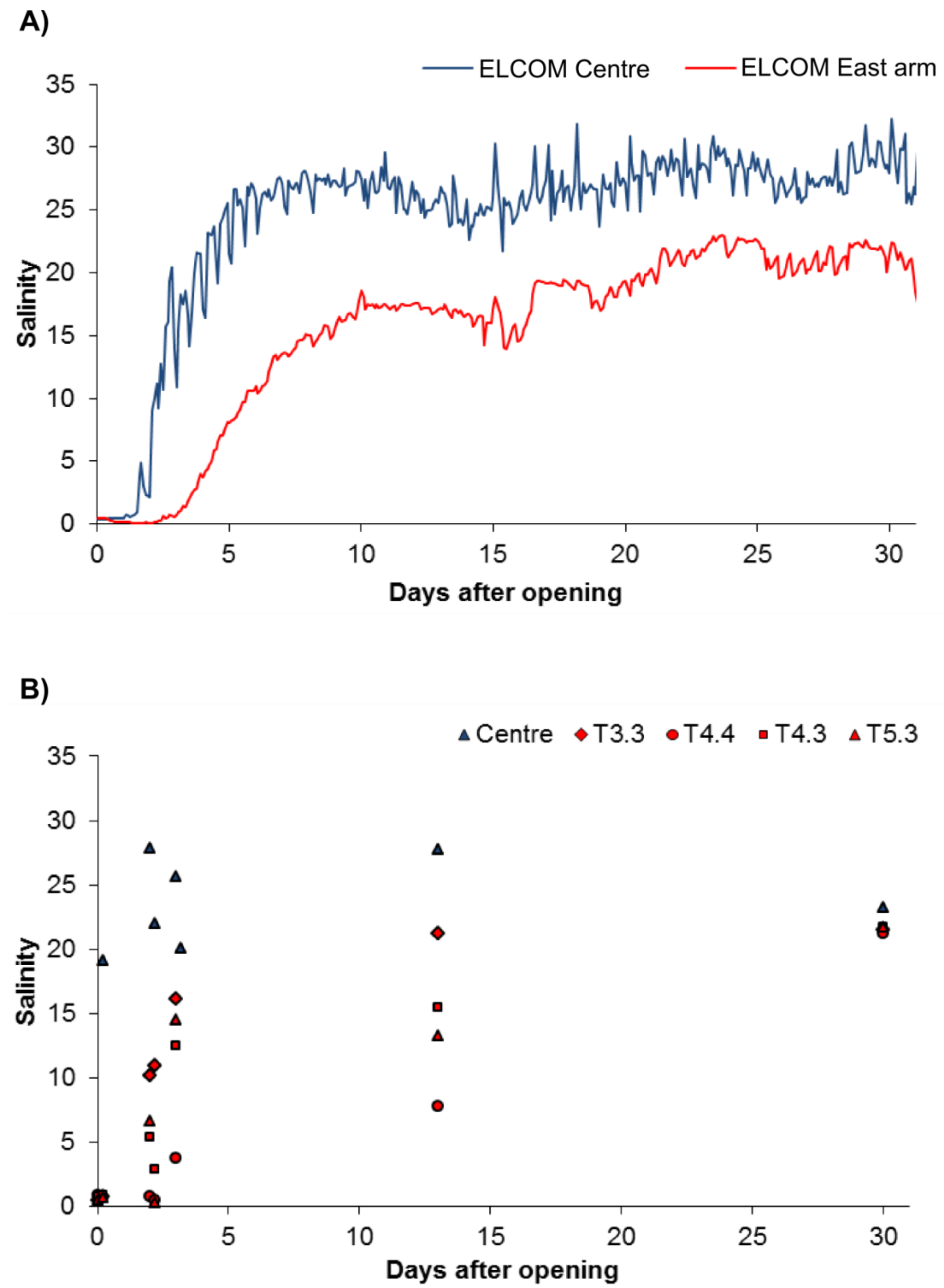


Figure 44: A) ELCOM surface salinities at the centre site and eastern arm (from simulation of July 2007 Walker's Bay opening), and B) measured surface salinity at centre site and 4 sites in the eastern arm (T3.3, T4.4, T4.3 and T5.3) following the July 2012 Walker's Bay opening. (Measured data provided by Andy Hicks, ES). Note that data are not directly comparable due to different time period simulated/sampled.

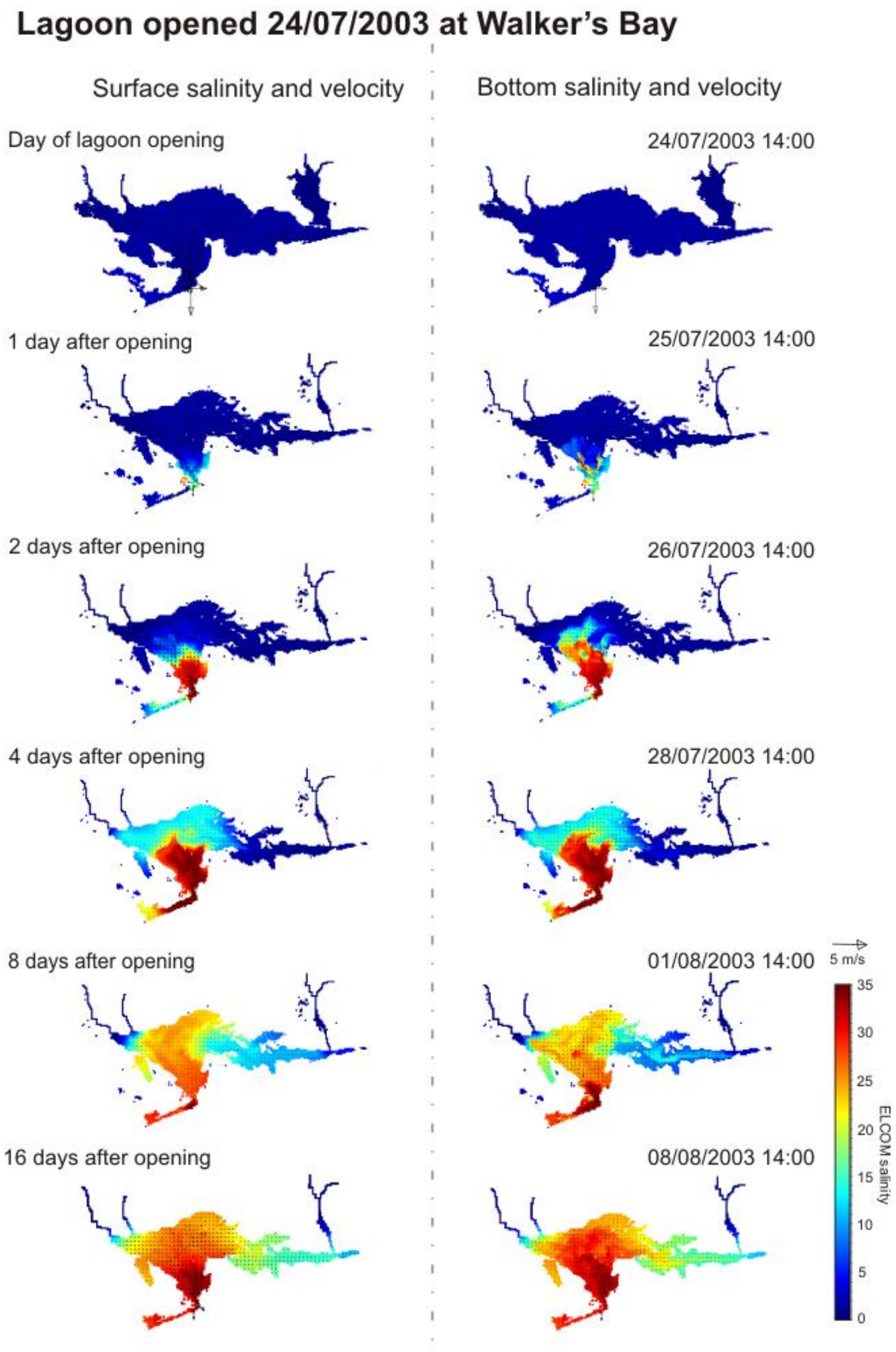


Figure 45: ELCOM-simulated salinity and current velocity on the day of, and in the days following, an opening at Walker's Bay on 24/07/2007. Please see the digital Appendix for 3D model animations.

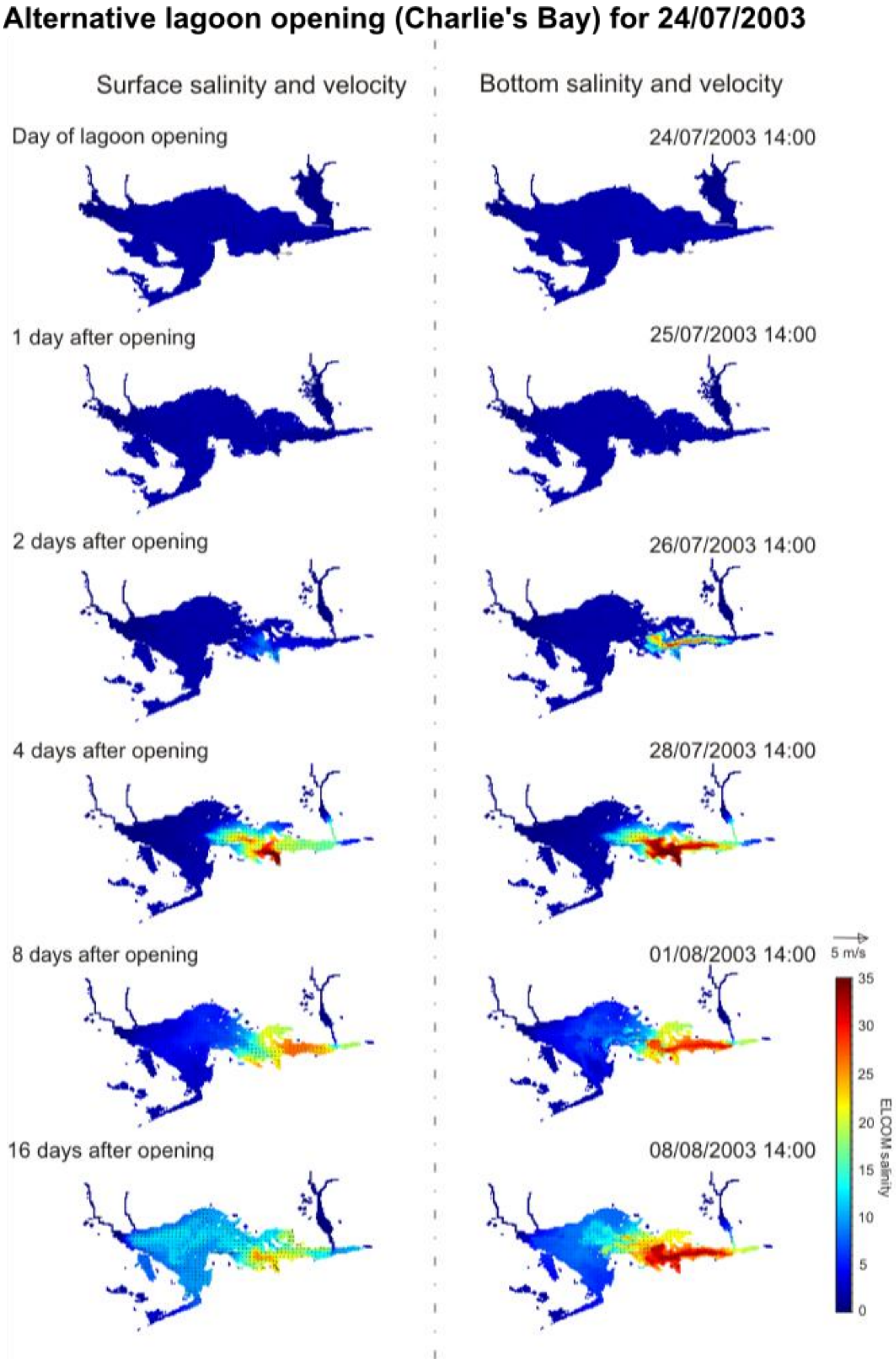


Figure 46: ELCOM simulated salinity and current velocity using the same input data as for Figure 45, but with the opening location changed from Walker's Bay to Charlie's Bay. Please see the digital Appendix for 3D model animations.

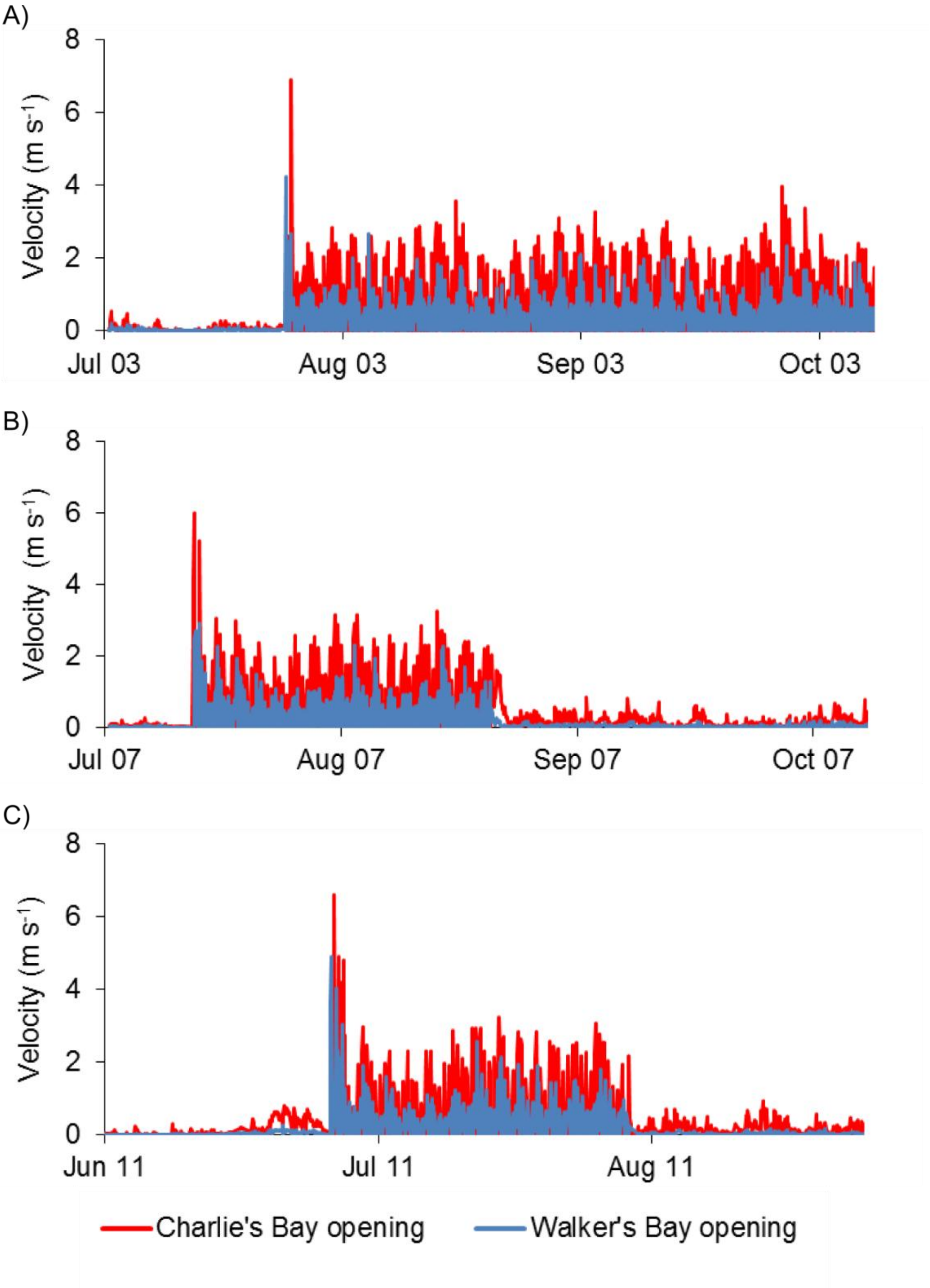


Figure 47: Effect of opening location on ELCOM-simulated velocities during lagoon opening for three separate periods (Jul – Oct 2003, Jul – Oct 2007, and Jun – Sep 2011).

3.6 Remote Sensing

Figure 48 shows a true colour composite of a Landsat 5 image captured on 20 December 2010. At this point in time the lagoon was open to the ocean, and the water clarity enabled observations to include bottom reflectance over the majority of the lagoon. Areas of high reflectance represent areas of shallow water or exposed sediment. Green or brown colours may be representative of macroalgae or macrophytes (e.g. *Ruppia*).

Figure 49 also shows a true colour composite of a Landsat 5 image captured on 21 January 2011, when the lagoon was still open. This image is possibly the last remote sensing image to be captured of Southland with Landsat 5, which soon after failed due to an electrical fault and the end of 28 years of remote sensing images from this satellite. Interpretation of the colours present in this image is similar to Figure 48.

Figure 50 shows the results of an unsupervised K-means classification of the image shown in Figure 48 (i.e., 20 December 2010). Deeper areas with less bottom reflectance are shown as red, with shallow areas appearing as yellow, purple and blue.

Figure 51 shows the results of unsupervised classification of a Landsat 7 image captured on 20 March 2012. On this date water clarity seemed to be much lower, and there was much less contrast and bottom reflection apparent than was observed in the previous Landsat 5 images. Some care needs to be taken when interpreting such an image, as reflectance may be caused by a combination of water column constituents and from the bottom substrate, macrophytes and macroalgae. It is possible that elevated turbidity may be responsible for higher reflectance near the Waituna Creek inflow.

Figure 48: True colour composite of bands 1, 2, 3 (B,G,R) of Landsat on 20 December 2010.

Figure 49: True colour composite of bands 1, 2, 3 (B,G,R) of Landsat on 21 January 2011. Grey boxes are used to mask some small areas of cloud.

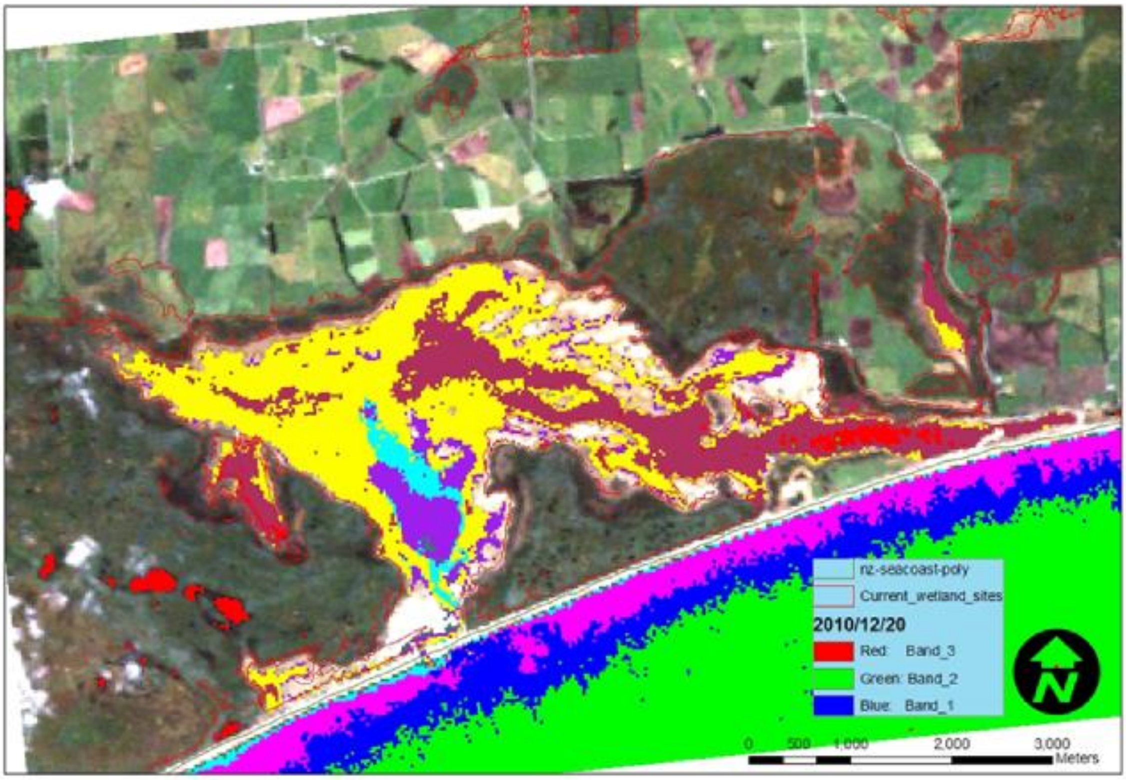


Figure 50: Unsupervised classification of pixels into similar spectral response classes on 20 December 2010.

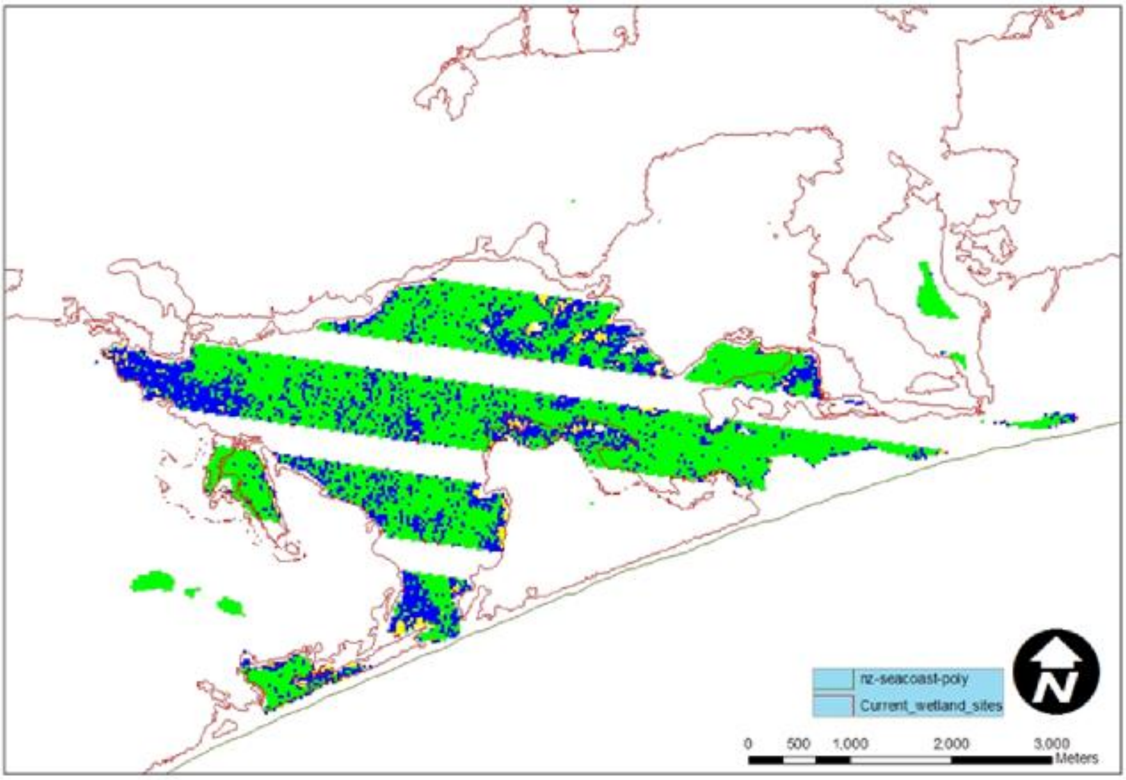


Figure 51: Unsupervised classification of pixels into similar spectral response classes on 20 March 2012. Strips of missing data are caused by instrument failure of the ETM+ sensor on-board Landsat 7.

4 Discussion

The objective of this study was to construct and apply process based models to assess whether various management scenarios could meet goals for sustaining the natural values of the Waituna Lagoon ecosystem. Specifically, an immediate goal of lagoon management is to prevent the decline and potential collapse of *Ruppia* beds. This study has highlighted aspects of physical and ecological interactions which have important implications for the management of Waituna Lagoon, and specifically how these interactions sustain *Ruppia* populations. Results from the 1D hydrodynamic-ecological model (DYRESM-CAEDYM) suggest that resuspension is a critical process affecting phytoplankton and suspended sediment concentrations in this shallow system, and that macroalgae – as well as phytoplankton – are an important influence on *Ruppia* dynamics in the lagoon, and specifically are a key part of any regime shift towards an alternate state whereby *Ruppia* disappears.

Model simulations using current (i.e. 2001–2011) nutrient loads, suggest that raising the opening trigger water level or reverting to a more “natural” opening regime whereby the lagoon opening is not actively managed, would result in a collapse in the *Ruppia* beds, likely due to increased shading as both macroalgae and phytoplankton biomass increase substantially. Water quality in Waituna Lagoon is known to be greatly affected by opening events, which result in a high degree of tidal flushing after the barrier is breached (Schallenberg *et al.* 2010). However, model simulations suggest that it is not possible to maintain an abundant and stable *Ruppia* population in the lagoon with changes to the opening regime alone, i.e., nutrient load reductions are required simultaneously. Comparatively small nutrient load reductions (e.g. 10–25% of current loads) appear to have a limited effect on *Ruppia* or the other primary producer groups, likely due to the very high level of current loading (based on AgResearch estimates c. 99% of total nitrogen and 90% of total phosphorus loading in freshwater inflows may originate from non-natural inputs or the consequence of catchment development). Model simulations indicate that to maintain stable and abundant *Ruppia* under a natural opening regime a substantial reduction in nutrient loads (i.e. 70–90%) is required. However, regular winter openings in combination nutrient load reductions of 50% nitrogen and 25% phosphorus may also achieve this goal.

Three-dimensional hydrodynamic modelling of the lagoon with ELCOM indicates that salinity distribution may be horizontally and vertically variable in the lagoon, and is influenced by opening location. Simulations also showed substantial short-term variations of the water level (by up to 0.5 m) at the eastern end of the lagoon in association with the strength of the prevailing westerly winds and resulted in significant effects on circulation currents in the lagoon hydrodynamics. Anecdotal observations of scouring of accumulated fine sediments during lagoon opening were reinforced in simulations with the three-dimensional model, which gave outputs of very high water velocities for grid cells near the lagoon opening during the initial opening phase. Salinity stratification may sometimes be present in the deep channel of the eastern arm, an area which tends to have *Ruppia* beds in close proximity, but has not been the focus of historical water quality monitoring effort by ES. Stratification has potential to have important implications on dissolved oxygen concentrations in bottom waters, which may be depleted if there is persistent stratification (i.e. greater than a few days or weeks). Stevens and Robertson (2010) noted that surficial sediments in deeper parts of the lagoon not only appear to be less well oxygenated than in shallower parts of the lagoon, but that they may also be undergoing a transition towards reduced oxygen status over the past 3–4 years. Any such transition could have important water quality implications as it may induce higher rates of release of dissolved nutrients, phosphate in particular, as well as potentially affecting the habitat suitable for *Ruppia* colonisation. An on-going sampling programme to monitor any changes in sediment oxidation-reduction status may be useful as bottom sediments provide a valuable means to integrate more rapid transitions in water column variables (Trolle *et al.* 2010).

4.1 Model performance

The performance of the calibrated model was generally highly satisfactory, typically simulating the timing and magnitude of modelled variables. Model statistics suggest that model performance was mostly comparable to, or better than, previously published applications of DYRESM-CAEDYM (e.g. Burger *et al.* 2008; Fragoso *et al.* 2011; Gal *et al.* 2009; Trolle *et al.* 2011b; Özkundakci *et al.* 2011). Where the model did not perform well, field measurements usually had relatively low values, and were often at or below analytical detection limits (Table 7), making for considerable uncertainty in the field data against which the model performance is statistically assessed. Although the model captured the timing and magnitude of elevated phytoplankton biomass and periods of low phytoplankton biomass, it did not capture some of the high chlorophyll *a* field measurements. It is likely that some of these events may have been highly localised, and hence not represented in the 1D horizontally averaged model, and/or possibly driven by extreme (and short-lived) resuspension events. Meteorological input data was obtained from Tiwai which is some distance (c. 15 km) from the lagoon, and thus may have affected the ability of the model to capture wind-driven resuspension events accurately. Sensitivity analysis of meteorological input data revealed that there was high sensitivity of the model output to wind speed. Furthermore, as model input typically consists of interpolated monthly samples for freshwater inflows, it may not be able to reproduce short-lived phytoplankton blooms resulting from periods of high nutrient loading not captured in the monthly monitoring. Proxies such as turbidity, which can be logged continuously *in situ*, and the development of relationships of turbidity to total nutrient concentrations for surface inflows may help to more accurately quantify the effect of storm flows on nutrient loads to the lagoon, which may be particularly relevant to capturing short-term periods of high loading, for example, when drain clearing takes place.

This model application required considerable development work to allow inclusion of *Ruppia* as a state variable, and to provide dynamic feedback between *Ruppia* biomass and bed shear stress, which affects resuspension of suspended sediments, organic matter and phytoplankton. There were limited field data available for calibrating and validating *Ruppia* and macroalgae, and conversion of field survey data (measured as percent cover) for *Ruppia* to a single biomass estimate is confounded by the lack of any biomass measurements in the lagoon and highly variable water levels, which greatly affect lagoon area. However, the model reproduced the trends identified in field surveys for *Ruppia* and macroalgae (Stevens & Robertson 2007; Robertson & Stevens 2009; Stevens & Robertson 2010; Sutherland & Taumoepeau 2011), and simulation of other variables (particularly chlorophyll *a* and phosphate) improved considerably after developing the model to include these two groups.

4.2 Model limitations

A scientific model aims to represent variables and processes in a logical and objective way, but is necessarily a simplified view of a complex reality. Model limitations and assumptions are detailed below and are summarised in Table 9.

4.2.1 Conceptual complexity

Conceptual simplifications of the system being modelled are unavoidable (e.g. Harris 1994), but omission of species and processes may affect model output. In this study, zooplankton and fish were omitted due to a lack of data with which to calibrate and validate the model. Zooplankton and fish assemblages in Waituna Lagoon are variable, and dependent to a substantial degree on whether the lagoon is open or closed (Schallenberg, Hall & Burns 2003; Duggan & White 2010), which would further add to conceptual model complexity. We attempted to compensate for the effect of zooplankton grazing on phytoplankton by slightly elevating values for the phytoplankton respiration rates. It is important to acknowledge that this will not capture the dynamic effect of variable zooplankton biomass on phytoplankton populations, or the effect of phytoplankton succession on zooplankton dynamics. However, freshwater zooplankton communities typically take time to establish once the coastal bar is closed and there is no clear indication of persistent dominance by larger grazers (e.g., *Daphnia* sp.) that are likely to exert direct control on phytoplankton biomass (Ian Duggan, pers. comm.).

As macrophytes (*Ruppia* spp.) are rooted in the sediment, it is assumed that there is no nutrient limitation and nutrient uptake is from the sediments (Section 2.3.1). Nutrient availability is typically much greater in sediments than in the water column; porewater nutrient concentrations (except for nitrate) may be orders of magnitude higher than in overlying water (e.g. Lohrer *et al*., 2010). Given the high levels of current nutrient loading to Waituna Lagoon, the assumption of no nutrient limitation is likely to be appropriate for the current state of the system, and even for management scenarios that reduce nutrient loads quite substantially. However, some macrophyte species may uptake nutrients from the water column (e.g. Dudley *et al.*, 2001), so the assumption of nutrient uptake from the sediment may have implications for determining the potential role of *Ruppia* as a nutrient sink in the lagoon. There does not appear to be a consensus in the literature on the relative importance of water column or sediment nutrient uptake and considerable variation in nutrient uptake and metabolism has been shown by seagrass species (e.g. Burkholder *et al*., 2007) and freshwater species of macrophytes (Mi *et al.*, 2008). Inclusion of more complex nutrient dynamics for *Ruppia* would therefore require more species- and system-specific data to parameterise *Ruppia* nutrient uptake and limitation functions.

Sediment nutrient dynamics implemented in CAEDYM for this study comprise a relatively simplistic process representation, with sediment nutrient release regulated by water column variables only, i.e. not by sediment nutrient content (Hipsey *et al.* 2011), as discussed in other DYRESM-CAEDYM applications (Trolle *et al.* 2011b; Özkundakci *et al.* 2011). However, model-specific output, which gives the contribution of each process represented in the model for each state variable, indicates that total nitrogen and phosphorus in the water column are highly influenced by resuspension, and dissolved sediment nutrient fluxes are comparably low, suggesting that internal loading from releases of dissolved nutrients may be relatively small compared to the external loading from catchment inflows. Unless future scenarios propose to increase external loading substantially, which has been shown to result in an increase in internal loading in shallow systems (Jeppesen *et al.* 2005), the present configuration should be sufficient to adequately simulate nutrient dynamics in the lagoon. It should, however, be informed with any transitions noted in the composition of, and potentially releases from, the bottom sediments (as alluded to at the beginning Section 4).

The results derived from a 1D model as opposed to a more complex 3D model, are related to the morphological complexity of the system being modelled, but must also account for the focus of the study. If the aim is to quantify the effect of management options over multi-year time scales then the computationally less demanding 1D model is a more practical solution. Results from a 3D model (ELCOM) do suggest, however, that salinity distribution is horizontally variable in the lagoon. Thus, the output from the 1D model used in this study (DYRESM-CAEDYM) should be interpreted as an integration of that horizontal variability. However, simplification of the system being modelled is an unavoidable reality even for the 3D model case. For example, grid size in the 3D model is also a balance between morphological complexity and computer run times, and cannot capture certain fine-scale features, such as narrow channels, that are smaller than the grid size (50 x 50 m in the present study). While the 3D model revealed some interesting features in the lagoon hydrodynamics, the ability of the 1D model to capture the temporal dynamics of the simulated variables suggests that the 1D model meets our objective to quantify the long-term effects of management options on lagoon ecology.

4.2.2 Boundary conditions

Ecological models generally resolve processes at a daily or sub daily time step, but input data are almost always available only as a sub sample of the idealised data. Storm flow events, for example, have been shown to have significant impacts on the overall nutrient budget in Lake Rotorua (Hoare 1987). A current study on the Lake Rotorua catchment indicates that a large storm event can transport as much phosphorus to the lake as approximately four weeks at base-flow conditions (Jonathan Abell, pers. comm.). In the present study, daily input data for inflows were mostly derived from monthly sampling for four surface inflows into Waituna Lagoon (but with the addition of one intensely sampled storm flow event in May 2011). This sampling frequency may have resulted in an underestimation of total nutrient and sediment loads, but may be able to be partially addressed by *in situ* loggers of proxies such as turbidity (see discussion in section 4.1)

The location of the water level recorder at the far eastern end of the lagoon is likely to have a substantial effect on the calculated water balance. Three-dimensional modelling of the lagoon with ELCOM indicates that a tilting of the water surface on a west-east axis across the lagoon occurs frequently. The water level is higher, on average, at the eastern end, with a maximum difference of c. 0.5 m, likely due to prevailing westerly winds causing water to accumulate on the opposite side of the lagoon. Furthermore, in a shallow system such as Waituna, evaporation sometimes represents a significant portion of the water balance, but the location of the climate station c. 15 km from the lagoon is likely to affect evaporation estimates, and thus our estimation of lagoon outflow derived from the water balance.

Table 9: Model (DYRESM-CAEDYM) assumptions and associated implications

Assumption	Implications
• 1-dimensionality (i.e. model is vertically resolved and horizontally averaged)	This implies that the model output represents the integration of horizontal variability in the lagoon, and so may not capture any localised horizontal features, such as variations in suspended sediments in surface waters in central parts of the lagoon versus bays. This also has implications for assessing model performance as measured against samples taken from one location in the lagoon may be influenced by localised events and thus may not represent horizontally averaged values.
• Daily values for nutrient and suspended sediment concentrations in inflows derived by linear interpolation between typically monthly samples	This may underestimate the influence of storm flows on nutrient and sediment loads entering the lagoon, and the simulations may not be able to reproduce short-lived responses such as phytoplankton blooms resulting from periods of high nutrient loading not captured in the monthly monitoring.
• Macrophytes (which may include many species in reality, e.g. *Ruppia polycarpa, R. megacarpa, Myriophyllum triphyllum*) are modelled as one group (conceptualised as *Ruppia* spp.)	There are likely to be differences in growth and respiration rates, tolerance to salinity etc. between macrophyte species. The model was parameterised using data from available literature. The model may not capture the rapid growth of an annual plant like *R. polycarpa*, which is likely to be able to quickly establish in areas of the lagoon that have only recently been inundated. Thus, macrophyte biomass in model output may be underestimated at higher elevations that are inundated for short periods only.
• Macroalgae (which may include many species in reality, e.g. *Enteromorpha* sp., *Bachelotia antillarum*, *Cladophora* sp.) are modelled as one group	As with macrophytes, there are likely to be differences in growth and respiration rates, tolerance to salinity etc. between macroalgae species. Model parameters were obtained from available literature, mostly applicable to estuarine species such as *Ulva* sp. and *Enteromorpha* sp., but the model will not capture any interactions between different macroalgae species.
• The macrophyte group is salinity limited (i.e. respiration rate is increased above a certain salinity)	This assumes that the effect of salinity on macrophyte growth is constant throughout the life cycle of the plant. There is some evidence to suggest that *Ruppia* spp. require freshwater for germination but adult plants are tolerant of high salinities (Brock, 1982). The moderate effect of high salinities on macrophyte growth assumed in this study may not capture the importance of low salinities during spring/early summer for annual plants such as *R. polycarpa*. Thus, results from scenarios that include openings during this period of the year should be interpreted with some degree of caution.
• No explicit representation of higher trophic levels, such as fish or zooplankton	The effect of zooplankton grazing on phytoplankton was partly compensated for within the model by parameter settings for elevated phytoplankton respiration rates, but this will not capture the dynamic effect of variable zooplankton biomass on phytoplankton populations. Grazing by higher trophic levels, such as fish and birds, may influence macrophyte biomass although the macrophyte beds are likely to be more important to higher trophic levels as habitat than as a direct food source.
• Sediment nutrient releases regulated by water column variables	Model output indicates that internal loading from releases of dissolved nutrients may be relatively small compared to external loading from catchment inflows. Should there be a transition towards reduced oxygen status in the lagoon sediments then this may induce higher release rates of dissolved nutrients from the sediments, requiring adjustment of model parameters. Changes in sediment composition (e.g. organic content) will not be accounted for in the present model setup, however.

4.3 Sensitivity analyses

In this study, we implemented a sensitivity analysis to quantify the effect of variation of a given parameter on the model output, while other parameters were kept at their calibrated values. By using a one-factor-at-a-time method (e.g. Morris 1991), we showed low to moderately high sensitivity of the model output across the parameters selected for the analysis, although the model was highly sensitive to those parameters that directly altered the growth and mortality rates of *Ruppia* and macroalgae. (N.B. Whilst the sensitivity of the model to parameters affecting salinity limitation of *Ruppia* were not included in this sensitivity analysis, the effect of salinity on *Ruppia* spp. and sensitivity of model output to the degree of salinity limitation imposed on *Ruppia* is discussed in Section 4.4 below). Two important insights into the behaviour of the Waituna Lagoon model were gained during this sensitivity analysis. First, the majority of the model output variables did not show a disproportionate response to given changes in parameter values when using the sensitivity ratio (SR) as an indicator. This result indicates that any potential uncertainty in the model input parameter values would not result in a disproportionate uncertainty in model output. This also has important implications for interpreting those model scenarios which aim to reduce, for example, nutrient loading to the lagoon. The model output for scenarios should therefore reflect the trajectory and the magnitude of change relatively accurately, but should still be viewed with some caution. Second, we identified complex interactive effects on model output of changing species-specific parameters, such as maximum growth rates for *Ruppia* and macroalgae. While the local sensitivity analysis in this study was not designed to detect interactions amongst parameters, the interactions observed were in accordance with the conceptual model (Figure 4). For example, the effects of changing the maximum growth rate of macroalgae would also influence the growth of *Ruppia* due to shading of macroalgae.

A formal uncertainty analysis of the model was outside the scope of the present study but could potentially provide a quantitative estimate of uncertainty in model input parameters. Uncertainty analyses of complex ecosystem models are generally computationally demanding, using techniques such as Monte Carlo simulations. Commonly, an uncertainty analysis using Monte Carlo methods requires a large number of model evaluations for which all model parameters are iteratively adjusted within physically realistic value ranges, which often results in thousands of simulations. At this stage, it is difficult to accurately determine the minimum number of simulations required for a Monte Carlo uncertainty analysis of the Waituna Lagoon model, because this would require a quantitative estimate of the standard deviation of the model output which can only be obtained after the initial few hundred model runs. Based on the comparably limited number of simulations of the sensitivity analysis carried out in this study, an initial conservative estimate of the required number of simulations would be 8,500. At present, a computing time for a single run of the Waituna Lagoon model over the entire simulation period (2001–2011) is c. 15 min, which renders the uncertainty analysis highly impractical for this study. We argue that, given the relatively low sensitivity of the model output to changes in parameter values, it is necessary to adopt the model scenario results as part of the decision support system, in preference to an otherwise more limited information base with which to proceed in implementing potentially costly management actions.

Results from the sensitivity analysis that manipulated model complexity suggested that model performance was typically reduced with decreases in complexity. For example, an observed decline in *Ruppia* biomass from 2007 onwards was simulated by the calibrated model, whereas for the model without representation of macroalgae, *Ruppia* biomass remained relatively stable from 2007 to the end of the simulation period (Figure 16). This goes some way towards justifying the inclusion of macroalgae in the conceptual model, despite the limited availability of field data. Although increasing model complexity by including higher biology (e.g. zooplankton) could have potential to further improve model performance, a balance between model complexity and available field data has been reached for the present study, which attained good measures of model accuracy. Removal of resuspension from the conceptual model drastically reduced model performance, particularly for total suspended solids and chlorophyll *a*, highlighting the critical role of this process in Waituna Lagoon. Resuspension of inorganic sediment, organic matter and phytoplankton is a key physical process that drives ecological dynamics of many shallow lakes and lagoons (Scheffer *et al.* 1993; Hamilton & Mitchell 1996; Schallenberg & Burns 2004; Drake, Kelly & Schallenberg 2010).

Slight changes in nutrient and sediment inputs in inflows (± 10%) did not result in disproportionate changes to model output. There was a minor increase in macroalgae biomass in both cases, likely due to complex interactions and feedbacks between nutrient loading and primary producers. Although a slight increase in nutrient loading might increase macroalgae growth, a slight decrease in nutrient loading may decrease phytoplankton populations, which then reduces light limitation on macroalgae. The influence of physical forcing data (wind speed ± 10% and air temperature ± 0.5 ºC) appears to affect some modelled variables. For example, macroalgae biomass decreases with an increase in wind speed, likely due to an increase in resuspension and consequently phytoplankton, which then shade macroalgae. Temperature will directly affect the rate of many modelled processes, such as organic matter mineralisation and primary producer growth and respiration rates. These results are consistent with those from the parameter uncertainty analysis, whereby kinetic parameters associated with *Ruppia* and macroalgae had an appreciable effect on model output for these variables. Where sensitivities promoted the biomass of *Ruppia*, macroalgae or phytoplankton, this was usually associated with reductions in the biomass amongst one of the other group(s). This reflects the competitive environment for primary producers for nutrients and light in shallow resuspension-affected systems such as Waituna Lagoon (Figure 4).

4.4 *Ruppia* spp. and salinity

Ruppia spp. are recorded as being abundant in a wide variety of habitats (i.e. fresh, brackish and saline waters) throughout temperate and tropical regions in New Zealand and Australia (e.g. Mason, 1967; Brock 1982; Carruthers *et al.*, 2007; Champion & Clayton, 2004). Mason (1967) first described *R. polycarpa* and *R. megacarpa* using plants from Lake Ellesmere, and listed known habitats and distributions for the two species. *R. polycarpa* habitat was described as fresh and brackish water, with specimens from c. 40 locations from the North and South Islands of New Zealand. Mason noted that the species was unusual because of the wide range of habitats in which it was found, i.e. in rivers, coastal lakes and lagoons, but also in inland lakes (up to c. 700 masl*).* *R. megacarpa* habitat was described by Mason as being saline pools and lagoons, based on specimens from c. 20 locations in both the North and South Islands of New Zealand.

In Australia, both *R. polycarpa* and *R. megacarpa* are found in fresh (salinities < 3) to hypersaline (salinities > 35) coastal lakes, lagoons and estuaries in New South Wales, Victoria, South Australia, Western Australia, and Tasmania (Brock, 1982; James & Brock, 1982). *R. megacarpa* tends to be more prevalent in permanently and intermittently open estuaries, whereas *R. polycarpa* tends to be found in ephemeral inland lakes and salt ponds (Snoeijs & van der Ster, 1983; Carruthers *et al.*, 2007). Research in ephemeral saltmarsh ponds in Victoria, Australia (that ranged in salinity from c. 14 to 370) suggested that *R. polycarpa* was an opportunistic species, adapted to germinate even when water was hypersaline (Vollebergh & Congdon, 1986). These species are typically described as being salt tolerant; in an assessment of the relationship between macrophyte occupancy and salinity *R. polycarpa* (and *Lamprothamium macropogon*, another macrophyte species identified in Waituna Lagoon by Sutherland and Taumoepeau (2011)) were described as having an "unequivocally positive relationship with salinity" (Smith et al., 2009). However, high salinities may be detrimental to *R. polycarpa* germination and growth in salinised wetlands in Western Australia (Sim *et al.*, 2006). The upper salinity limit for *R. polycarpa* germination was 40 – 50 and survival of adult plants declined markedly at salinities above 45, although growth was not significantly affected at salinities between 6 and 30. Experimental results by Gerbeaux (1989) using samples from Lake Ellesmere, suggest that *Ruppia* seedling growth was reduced at high salinities, but the experiments used tap water combined with sea salt in some experiments, and water from Lake Ellesmere or seawater in others. The ionic composition of the different solutions was very different, particularly the concentration of sulphates associated with magnesium, and potassium, both of which may influence the growth of macrophytes and potentially confound the results (see pages 158-159 in Gerbeaux's thesis for discussion on this point). It should also be considered that germination experiments under controlled salinity conditions may not consider many other factors, such as what level of germination per unit area is required for macrophyte persistence/abundance, or that synchronised high rates of germination are not effective as a survival strategy in ephemeral or unstable environments. It is likely that enhanced germination at low salinities is a response to inundation after high rainfall/inflow events, rather than a direct salinity limitation at high salinities; i.e., an effective survival strategy in ephemeral habitats of fluctuating salinity (Mary de Winton, NIWA, pers. comm.).

In New Zealand, *R. polycarpa* has been recorded in freshwater lakes (e.g. Coffey & Clayton, 1988; Wells *et al.*, 1997), but has also been recorded in brackish and saline habitats, for example, in the Chatham Islands (Champion & Clayton, 2004), Lake Ellesmere (e.g. Gerbeaux, 1989), and Waituna Lagoon (e.g. Johnson & Partridge, 1998). Excessive growth of *R. megacarpa* in a saline (and sometimes hypersaline) pond in Nelson has been a problem for the pond's users (model boat enthusiasts), necessitating drainage and spraying with chemicals such as glyphosate in an effort to control growth (Wells *et al.*, 2010). These records, and the available literature on the two species, suggest that both *Ruppia* species are tolerant of a range in salinities. *R. megacarpa* likely requires saline habitats, although *R. polycarpa*, whilst salt tolerant, may respond positively to fresh/brackish water for germination (e.g. Brock, 1982; Sim *et al.*, 2006).

Increases in salinity in Waituna Lagoon have been linked with the decline in *Ruppia* spp. observed between 2009 and 2011, as the length of time during which the lagoon was open to the sea increased during the same period (Robertson & Funnell 2012). As a sensitivity analysis we have investigated the response of modelled *Ruppia* biomass to varying degrees of salinity limitation. The calibrated model effectively imposes a mild, rather than severe, effect of salinity on *Ruppia* growth. In contrast, with no salinity limitation at all, modelled *Ruppia* biomass is increased over much of the simulation period, and appears to closely match observed data. However, survey methodologies used in 2007 were different to those used in later years, and so the data may not be directly comparable. Increasing salinity limitation beyond that which is simulated by the calibrated model does not result in a better match with observed data.

The calibrated model salinity function also compares very well with the experimental results of Sim *et al.* (2006), although the influence of mid-range salinities is slightly reduced in the model compared to the experimental results. However, surveys in Waituna Lagoon indicate that *R. megacarpa* abundance was similar to or greater than that of *R. polycarpa* in three surveys out of four, i.e., in 2007, 2010 and 2011 (Stevens & Robertson, 2007; Robertson & Stevens, 2009; Stevens & Robertson, 2010; Sutherland & Taumoepeau, 2011). Our modelled *Ruppia* represents both species, and so the slightly lesser effect of salinity on *Ruppia* growth (compared to the experimental results of Sim *et al.* 2006) is likely justified, given that *R. megacarpa* is a saline species.

The mild salinity limitation imposed by the calibrated model parameters appears to result in the best match between observed and modelled data. Furthermore, it is consistent with literature that indicates that although *R. polycarpa* may require fresh/brackish water at certain times, generally, *Ruppia* spp. are adapted to live in environments with fluctuating salinity.

4.5 Nitrate toxicity

Nitrate concentrations may sometimes reach levels that are toxic to aquatic fauna, such as macroinvertebrates and fish, and limits have been set within water quality guidelines to afford protection to sensitive species (e.g. Hickey & Martin 2009 and references therein). Although these guidelines do not specifically cover the effect of nitrate toxicity on plants, research from overseas has indicated that high concentrations of nitrate (and/or ammonium) may have a direct toxic effect on seagrasses (e.g. Burkholder *et al.* 1992, 1994, van Katwijk *et al.* 1997).

Mesocosm experiments using the eelgrass *Zostera marina* have indicated that increased water column nitrate was associated with a decline in plant biomass, decreased C: N tissue ratios, and possible C-limitation for the plants (Touchette *et al.* 2003). The effects of nitrate toxicity on seagrasses are not consistent amongst the studied systems, however, and are perhaps dependent on, or confounded by, other factors such as temperature (e.g. Nelson 2009). The effect of high nitrate levels may also be highly species-specific. For example, growth of *Zostera marina* was decreased in treatments subject to long-term nitrate enrichments, whereas growth of *Halodule wrightii* and *Ruppia maritima* was stimulated in the same treatments (Burkholder *et al.* 1994).

Nitrate levels in Waituna Lagoon were often high (> 1 mg L^{-1}) between 2007 and 2011, i.e., the period when *Ruppia* spp. abundance was observed to decline (Figures 12-13). It is possible that direct nitrate toxicity is a stressor for *Ruppia* populations in the lagoon (in addition to a decline in light levels caused by algal growth that is promoted by high nitrogen loading). However, given the lack of consensus in the literature, direct nitrate toxicity on *Ruppia* has not been included in the model and controlled experimental studies would be required to investigate the possibility that direct nitrate toxicity is a significant factor contributing to the observed *Ruppia* decline in this system.

4.6 Lagoon management scenarios

The scenarios simulated using the model were broadly categorised into i) hydrological, ii) nutrient and sediment reduction scenarios and iii) combined hydrological and nutrient reduction scenarios. Hydrological scenarios involved alterations to the lagoon opening regime, or altered climate (i.e. rainfall, surface inflows), and nutrient and sediment reduction scenarios involved reductions in nutrient and/or sediment loading. Combined scenarios involved combination of the natural opening regime or winter opening regime scenario with nutrient reduction scenarios. The scenarios were used to address the overall aim of this study:

to assess whether various management scenarios could meet goals that would prevent the decline and potential collapse of *Ruppia* beds, and to provide a pathway to management to sustain an abundant and stable *Ruppia* population in the lagoon. We have defined an abundant and stable *Ruppia* population as having average *Ruppia* biomass similar to that observed in a survey by Stevens and Robertson (2007) in March 2007 (i.e. 30 – 40 g C m^{-2}), and with minimum biomass not less than c. 20 g C m^{-2} (similar to that observed in 2009 when *Ruppia* was observed to be in decline by Robertson and Stevens (2009). We have defined a “regime shift” as a situation in which macroalgae and/or phytoplankton dominate primary producer biomass and *Ruppia* biomass declines to below c. 5 g C m^{-2} and shows no recovery.

4.6.1 Hydrological scenarios

For the scenarios whereby the lagoon opening was either not actively managed, or opened only at a water level “trigger point” of 2.8 masl, *Ruppia* biomass collapsed within four years and did not recover, and macroalgae/phytoplankton were the dominant primary producers, meeting the criteria of a “regime shift” as defined in Section 4.6. The absence of regular tidal flushing of nutrients transported to the lagoon via surface and groundwater inflows, promoted growth of phytoplankton and macroalgae. Due to increased shading, *Ruppia* biomass decreased, enhancing resuspension of phytoplankton and exacerbating light limitation of *Ruppia*. Subsequently, high phytoplankton biomass shaded macroalgae, resulting in a concurrent decrease in macroalgae biomass. This is consistent with research that suggests that increasing eutrophication in coastal systems results in, firstly, the replacement of macrophytes (e.g. seagrasses, *Ruppia* in the present study) with macroalgae, and secondly, the replacement of macroalgae by phytoplankton (Valiela *et al.* 2012). Shading by epiphytic macroalgae has also been identified as a cause of seagrass loss in many Australian estuaries (Walker & McComb 1992). If retention of *Ruppia* is the aim of Waituna Lagoon’s management, then opening regimes that are not actively managed are not sustainable given anthropogenic nutrient loads from the catchment, which could represent a c. 100-fold increase in total nitrogen and c. 10-fold increase in total phosphorus relative to “natural” catchment nutrient loss rates (see section 2.7.2).

Opening for one month every spring resulted in reduced biomass of *Ruppia* relative to the baseline scenario for much of the simulation period, and increased biomass of phytoplankton and macroalgae. Opening for 3 months every winter resulted in *Ruppia* biomass similar to the baseline scenario, which is unsurprising given that the average lagoon opening duration in the baseline scenario was 4 months, with the majority of openings in winter. *Ruppia* biomass was slightly reduced in the scenario with a more variable opening duration compared to the regular 3-month opening, consistent with a reduction in available habitat (due to lower water levels) and slightly elevated salinities. Opening the lagoon for 6 months every summer resulted in *Ruppia* biomass that was similar to the baseline for the first half of the simulation period (when opening duration averaged 5 months), but was reduced for the latter part of the simulation compared to the baseline (where opening duration averaged 3 months). This is consistent with lower water levels limiting available habitat, and elevated salinities imposing mild limitation on growth during prolonged openings.

These hydrological scenarios of lagoon barrier management highlight the importance of regular and sustained periods of opening for maintaining a *Ruppia* population in the lagoon, given present nutrient loadings (i.e. *Ruppia* beds will collapse if there are no or very few openings). However, none of these scenarios resulted in an abundant and stable *Ruppia* population in the lagoon. Despite some salt tolerance of *Ruppia* spp. if high salinities are likely to suppress germination of *R. polycarpa* then it is reasonable to assume that habitats that are permanently saline (or consistently saline during spring/summer) may preclude the long term persistence of this species (e.g. Sim *et al.* (2006) and Section 4.4). Furthermore, opening the lagoon exposes *Ruppia* to desiccation and reduces available habitat (by reducing the lagoon area from c. 10 up to c. 20 km^2). Finally, this management approach represents a shift away from the more “natural” state of the lagoon as a coastal lake, and towards a periodically (annually) estuarine ecosystem, which is different to the natural values recognised in the RAMSAR classification (1976).

4.6.2 Nutrient and sediment reduction scenarios

Current catchment nutrient loads are highly influenced by anthropogenic, or “non-natural”, inputs. The nutrient and sediment reduction scenarios that were relatively low in magnitude (i.e., 10–25% reduction of current loads, when current loads represent c. 100-fold increase in N and 10-fold increase in P relative to “natural” loads), perhaps unsurprisingly resulted in only a small change in modelled variables. “Natural” catchment nutrient loads were derived by extrapolating areal estimates of nutrient loss for current peatland areas to the whole catchment, and could have moderate error, but nevertheless demonstrate the dominance of anthropogenic sources of nutrients to the lagoon.

Nutrient reduction scenarios that included a reduction in nitrogen loading of 50% or more resulted in increased *Ruppia* biomass, decreased macroalgae biomass and chlorophyll *a* concentrations. There was an approximately proportional reduction in modelled chlorophyll *a* with reductions in nutrient concentrations. For example, mean chlorophyll *a* over the simulation period was 2.7 µg L^{-1} for the 25% nutrient reduction scenario, and 0.9 µg L^{-1} for the 90% nutrient reduction scenario, compared to 3.6 µg L^{-1} in the base scenario. The scenario that resulted in the highest *Ruppia* biomass, lowest macroalgae biomass and chlorophyll a concentration was the 90% reduction in both nitrogen and phosphorus. However, a 50% reduction in nitrogen and 25% reduction in phosphorus loading resulted in an abundant and stable *Ruppia* biomass (as defined at the beginning of Section 4.6), and reduced macroalgae and chlorophyll *a* concentrations. Comparison of nutrient loads in New South Wales ICOLLs (Intermittently Closed and Open Lakes and Lagoons) with that of Waituna Lagoon indicated that a 52% reduction in nitrogen loads and 23% reduction in phosphorus loads would be required to maintain moderate environmental quality (Scanes 2012). A recent literature review on eutrophication of coastal lagoons (Schallenberg & Schallenberg 2012) indicated that the current nitrogen loading to Waituna Lagoon exceeds published thresholds for sustained seagrass health (which range from c. 5–30 mg N m^{-2} d^{-1}). Also, guidelines for nutrient load criteria to limit eutrophication in coastal systems suggested that in shallow New Zealand ICOLLs the areal load for nitrogen should not exceed 30 mg N m^{-2} d^{-1} and phosphorus should not exceed 1.5 mg P m^{-2} d^{-1} (Wriggle 2012). Current nutrient loading to Waituna Lagoon (based on average loads for 2001–2011; Table 1) are c. 50 mg N m^{-2} d^{-1} and c. 2 mg P m^{-2} d^{-1}. The results from the model are therefore consistent with other research which indicates at least a 50% reduction in nitrogen loading and a 25% reduction in phosphorus loading are required to sustain an abundant and stable *Ruppia* population in the lagoon.

The results of the 50% reduction in nitrogen and phosphorus (when compared with the 50% reduction in nitrogen and 25% reduction in phosphorus) may not be immediately intuitive but do highlight the complex negative and positive feedbacks between the three primary producer groups. For example, the 50% reduction in nitrogen and phosphorus actually results in lower *Ruppia* biomass than the 50% reduction in nitrogen and 25% reduction in phosphorus. Chlorophyll *a* concentration is greatly reduced with the 50% reduction in both nutrients, but macroalgae biomass is increased, likely because of a reduction in light limitation as phytoplankton biomass is lower. The result is then a decrease in *Ruppia* biomass as the *Ruppia* is shaded by macroalgae. It follows that the effect on *Ruppia*, macroalgae and phytoplankton of reducing nutrient loads to the lagoon will not necessarily be linear in this inherently dynamic system.

Similarly, model simulations indicate that there may be unexpected consequences of reducing only phosphorus loading. Model-specific output, which gives outputs of the level of nitrogen and phosphorus limitation for phytoplankton, indicated that phytoplankton were typically phosphorus limited in the base scenario (Appendix 4). However, under the scenarios that did not reduce nitrogen loading by at least 50%, but did reduce phosphorus loading by at least 25% (i.e. 25% reduction in phosphorus and sediment, and 50% reduction phosphorus with 25% reduction in nitrogen), although chlorophyll *a* decreased slightly, there was a concurrent slight increase in macroalgae biomass. Nutrient limitation is known to be variable and dependent upon, for example, season, salinity of the system and/or phytoplankton species composition, and reducing the input of either nitrogen or phosphorus often has consequences for the other nutrient species (Boesch 2002). Thus management options that aim to reduce only one nutrient should be considered with caution and need to extend beyond solely considerations of limitation of planktonic biomass.

4.6.3 Combined hydrological and nutrient reduction scenarios

If the lagoon is not opened at all (i.e. natural opening regime) then very substantial nutrient load reductions (at least a 70% reduction in nitrogen and phosphorus) are required to obtain a stable and abundant *Ruppia* population. Further reductions (i.e. 90% nitrogen and phosphorus) may also be required to reduce macroalgae biomass to a low level. This is consistent with research by Scanes (2012) that indicated nitrogen loads would need to be reduced by 79% and phosphorus loads by 72% for the lagoon to return to a near reference condition, i.e. clear waters with minimal algal biomass and strong seagrass growth (Scanes, 2012).

When nutrient reductions were combined with the 3-month winter opening scenario the results were consistent with nutrient reduction scenarios and research by Scanes (2012), Schallenberg and Schallenberg (2012) and Wriggle (2012). That is, a 50% reduction in nitrogen loading and 25% reduction in phosphorus loading resulted in an abundant and stable *Ruppia* population, and reduced macroalgae biomass and chlorophyll *a* concentration. These reductions, when combined with the 3-month winter opening regime, resulted in *Ruppia* biomass that was more stable when compared to the scenario that reduced nutrient loading only (i.e. using the more variable 2001–2011 opening regime). This is consistent with lowered water levels associated with some long openings (up to 11 months) during 2001–2011, that would have limited *Ruppia* biomass.

In summary, the scenario results indicate that nutrient load reductions are required to achieve a healthy lagoon with abundant and stable *Ruppia* biomass, i.e., it is not possible to achieve healthy *Ruppia* populations with changes to the opening regime alone. There is little effect of nutrient load reductions less than 50% (particularly for nitrogen) on primary producer groups. Very severe nutrient loads reductions (i.e. 90%) are required to reduce macroalgae and phytoplankton to very low levels with a return to a more natural opening regime. However, under the scenario that involved a 50% reduction in nitrogen and 25% reduction in phosphorus combined with winter openings, both macroalgae biomass and chlorophyll *a* concentration are reduced and *Ruppia* biomass is similar to that observed in 2007, supporting the nutrient load guidelines indicated previously in Scanes (2012), Schallenberg and Schallenberg (2012) and Wriggle (2012).

The scenarios are necessarily hypothetical and in reality there are factors that are likely to affect the predictions made in model simulations. For example, lagoon opening duration is uncertain as the lagoon closes by natural processes, but longer openings are likely to affect *Ruppia* biomass. There will also be variations in nutrient inputs associated with variations in climate and management practices (e.g. drain clearing), that will influence phytoplankton, macroalgae and *Ruppia*. Climate scenarios suggest that years with high or low rainfall may influence *Ruppia* biomass (e.g. by affecting water levels in the lagoon). Furthermore, the scenarios do not account for lag times in the response of surface and groundwater nutrient loads to changes in land management. Thus, the predicted outcomes of scenarios represent the response of the lagoon to hypothetical conditions and consideration should be given to factors which will likely affect the actual outcome were the scenarios to occur or be implemented.

4.7 Recommendations and opportunities for model development

4.7.1 Model input and calibration

Because models are seldom constructed using ideal input datasets, it is possible to improve model accuracy by provision of more detailed data. Daily gauging of inflow discharge rates and increasing the confidence associated with groundwater input estimations have potential to improve the water balance and would help to refine model performance in terms of its predictive capabilities. Likewise, water level recorders at more than one location could improve the water balance. Specifically, a level recorder near the lagoon barrier would improve estimations of water transport volumes due to tidal influence and barrier opening.

Regular collection of phytoplankton samples (weekly to monthly) for analysis of biomass, species composition and succession may enable more accurate simulation of phytoplankton (chl *a*) dynamics in the lagoon. New genetic techniques also have the potential to improve the efficiency of species composition analysis (Rueckert, Wood & Cary 2007). Direct measurement of model parameters such as phytoplankton growth rates and nutrient limitation (by bioassays) could also help to refine model predictions by allowing direct comparisons with model-specific physiological parameters (e.g., rates of production, nutrient limitation, etc.). Likewise, *in situ* and/or *in vitro* experiments could help to quantify rates of sediment nutrient release and resuspension (e.g. Gibbs & Özkundakci 2010; Jones *et al.* 2011). More regular and comprehensive surveys of *Ruppia* and macroalgae (ideally biomass as well as percent cover) could improve the representation of primary producers in the system. In particular, quantitative data on macroalgae species composition and biomass is important, given the likely significant impact on macroalgae on *Ruppia* populations, and the uncertainty over macroalgae species identification in previous surveys as noted by Sutherland and Taumoepeau (2012). In the absence of more detailed surveys, estimation of uncertainty in the current measurements would be useful.

Increasing biological complexity in the model would require the collection of additional input and calibration data. For example, regular biomass estimates for zooplankton and fish would be necessary to be able to include these groups in the model and calibrate their responses. Measurements for dissolved organic carbon could help to quantify seasonal variations or long-term trends in the lagoon carbon pool and may be useful to more accurately quantify its influence on light attenuation.

High-resolution, real-time lagoon monitoring data from automated platforms such as the one recently installed in Waituna Lagoon will provide detailed insights into lagoon dynamics at long-term and sub-daily time scales. As the monitoring record expands, these data could be used to fine-tune model performance and better assess the accuracy of the baseline simulation over short time scales.

4.7.2 Scenario development

The results of simulation scenarios presented in this report provide valuable insight into best estimates of the response of the lagoon ecosystem to management practices. These results reveal the opening regimes and nutrient load reductions that are likely required to maintain a healthy lagoon, with stable and abundant *Ruppia* populations, relatively low phytoplankton and macroalgae biomass, as was the study objective.

The current scenarios typically rely on an assumption that it is possible to control the time of both opening and closing the lagoon barrier. In reality closure of the lagoon occurs due to natural processes, and during 2001–2011 the lagoon remained open for periods between 34 to 330 days. Opening is also dependent on lagoon water levels, climate and ocean conditions but there is lesser variability than for the closure case. It should be noted that the scenario with a variable opening duration indicated that *Ruppia* biomass is likely to be reduced during periods of prolonged opening (as lowered water levels reduce lagoon area and associated *Ruppia* habitat). The list of scenarios simulated was not exhaustive and there may be scope for further scenario development that could further refine understanding of the interactive effects of hydrology (e.g. lagoon openings) and nutrient loads to the lagoon and/or other factors such as climate change or sea level rise.

Furthermore, the model can be used to assess the likely effects of any nutrient load increases, should they occur in the future. Current modelling scenarios do not account for lag times in the response of nutrient loads to land use change. For example, nutrient load reductions were applied instantaneously at the beginning of the simulation period, whereas in practice any change in land management is likely to result in a more gradual decline in surface and (particularly) groundwater nutrient concentrations. Further modelling should utilise an extended simulation period and build in a function to progressively shift nutrient loads from “current” levels to the calculated equilibrium load for a given land use, in order to assess the effects of various lag times. A catchment model may be used to begin to inform what lag times occur in response to changes in land use or practice within the catchment.

The Soil Water Assessment Tool (SWAT) is currently being set up for the Waituna Lagoon catchment. This model is being set up to provide inflow volumes and nutrient concentrations on a daily time step as an input to the Waituna Lagoon lake model. However, Lidar data, which are an essential component of the catchment model, became available after the lake model was already set up. The output of the SWAT model will be used to assess the assumptions made during the construction of the lake model with regards to the method used to derive inflows from field observations.

A summary of recommendations for future monitoring and modelling of the lagoon is provided in Table 10.

4.8 Remote sensing

Remote sensing in Waituna Lagoon may have limited potential as a tool to monitor water quality, due to the shallowness and variable bottom reflectance of the lagoon. While correction techniques in the literature can partially correct for bottom reflectance (Ohde & Siegel 2001), there are uncertainties in this correction process, particularly where reflectance may be contributed by bottom sediments, which are variable across the lagoon, as well as macrophytes and macroalgae. Additionally, the Landsat satellite has low spectral sensitivity and low signal ratios, which limit the applicability of images generated to fairly simple applications. During less turbid conditions, remote sensing has the potential to allow observations of macrophyte distributions and possibly macroalgae also, however imagery would have to be corrected for depth variation. The Landsat Data Continuity Mission (LDCM), which carries the Operational Land Imager (OLI), promises improved quantization and spectral resolution over Landsat series satellites, and this data may prove useful in potential future application.

4.9 Conclusions

The model applications to Waituna Lagoon have highlighted complex interactions and processes that have a significant effect on the lagoon ecology. Results from this study are consistent with previous research that suggests that increasing eutrophication in this type of system tends to result in, firstly, dominance of macroalgae over *Ruppia* beds, with subsequent dominance of phytoplankton over macroalgae. This type of “regime-shift” may be difficult to reverse as the loss of macrophytes destabilises bed sediments, enhancing resuspension of sediment and phytoplankton, and further reducing water clarity.

Scenario testing of various management options suggests that opening of the barrier and subsequent flushing of nutrients are crucial to maintaining some *Ruppia* in the lagoon given the current catchment nutrient loads. While this suggests barrier management is currently a necessary management tool, opening the barrier regularly for what can be extended periods (up to 330 days in any one sustained period during 2001–2011) exposes macrophyte beds to desiccation and high salinities, and reduces available habitat. It also represents a shift away from Waituna Lagoon’s “natural” state as a coastal lake, and towards a more estuarine ecosystem for a significant portion of time. However, model simulations using current (i.e. 2001–2011) nutrient loads suggest that raising the opening trigger water level or reverting to a more “natural” opening regime whereby the lagoon opening is not actively managed, would result in a collapse in the *Ruppia* beds, and subsequent “regime shift” to an algae dominated state.

Results indicate that given current catchment nutrient loads it is not possible to maintain a “healthy” *Ruppia* population in the lagoon with changes to the opening regime alone. Nutrient load reductions are required to sustain persistent and productive *Ruppia* beds. In order to revert to a more natural opening regime nutrient load reductions of 70–90% are likely required to sustain an abundant and stable *Ruppia* population, with low macroalgae and phytoplankton biomass. However, opening the lagoon for 3 months each winter combined with a 50% reduction in nitrogen loading and a 25% reduction in phosphorus loading may represent a balance between catchment nutrient loads and barrier management that sustains healthy *Ruppia* in the lagoon whilst minimising the “estuarine” period for the lagoon. These nutrient load reductions (50% of current nitrogen loads and 25% of current phosphorus loads) are consistent with other research on nutrient loading thresholds for reducing the effects of eutrophication and sustaining healthy macrophyte populations in coastal ecosystems (e.g. Scanes 2012; Schallenberg & Schallenberg 2012; Wriggle 2012).

Table 10: Summary of recommendations

1. Develop a set of monitoring data to support additional comparisons of key variables in the lagoon. The critical part of this programme would be to provide quantitative measurements of macrophyte and macroalgae biomass across the lagoon for comparisons with model output and to validate trends observed in the model simulations. Consideration could also be given to a long-term monitoring programme of sediment oxidation status at a small number of stations in the lagoon. Recommendations for monitoring of higher trophic levels such as zooplankton and fish are beyond the scope of this report but if undertaken as part of a regular monitoring programme, could be aligned with developing model capacity for these variables. High-frequency monitoring of physical variables (temperature, water level, salinity, light) and chlorophyll fluorescence from a fixed remote monitoring station installed by ES should be used to refine model simulations to ensure they are consistent in reproducing both short-term and long-term variations in these variables. Other “useful to have” data could include high-frequency measurements of currents in the lagoon during an opening phase and high-resolution acoustic surveys of bed sediments to examine extent of scouring in the vicinity of the opening.
2. Consideration could be given to specific ecological research and experimental work which would allow for greater confidence in model predictions. For example, salinity tolerance of *Ruppia* spp., nitrate toxicity on *Ruppia* spp., nutrient bioassays to assess phytoplankton nutrient limitation (specifically N vs. P), a monitoring programme to support assessments of phytoplankton species composition and succession, quantitative surveys of macroalgae species composition and biomass, salinity tolerance of macroalgae species etc.
3. Prioritise model boundary condition measurements to refine model simulations. ES currently has a number of studies to better quantify groundwater discharge and composition to the lagoon and within the catchment, as well as the effect of storms and drain clearing on variability of nutrient loads. These studies will be valuable in providing more accurate assessments of variations in discharge and nutrient concentrations to the models than the current method for surface inflow nutrient concentrations, which involves interpolation between monthly measurements. Similarly, more proximal meteorological data to Waituna Lagoon than the present station used for model input, 15 km from the lagoon, could be incorporated into model input as data became available. In terms of input data it is important to ensure coastal monitoring is used to inform the coastal boundary inputs to the models, and refinements to the coastal monitoring programme to assist with this goal may be worth considering.
4. Further scenarios could be used to refine understanding of the interactive effects of opening regimes, nutrient load reductions or increases, climate changes effects, sea level rise, etc. The model outputs could be used as a tool for ES to engage its ratepayers, farmers in the Waituna catchment, other stakeholders (e.g. DOC, Fish & Game) and the community in both development of scenarios and evaluation of their potential outcomes.
5. Further modelling could utilise an extended simulation period and build in a function to progressively shift nutrient loads from “current” levels to the calculated equilibrium load for a given land use, in order to assess the effects of various lag times. A catchment model may be used to begin to inform what lag times occur in response to changes in land use or practice within the catchment.
6. Consider an application of ELCOM-CAEDYM in order to provide a three-dimensional representation of the lagoon dynamics, including the dynamics of distributions of macroalgae and macrophytes.

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Appendices

Appendix 1

PCLake Memo

This PCLake memo was sent to ES in May 2012 to detail the issues we identified with this model application, and the macrophyte (*Ruppia*) model we proposed as an alternative.

Please note that the limitations associated with using a separate macrophyte model or with CAEDYM (which did not originally include dynamic feedback between macrophyte biomass and resuspension) outlined in this latter part of this memo have since been resolved. That is, we have developed CAEDYM to include a *Ruppia* specific macrophyte group and dynamic feedback between *Ruppia* biomass and resuspension (see Section 2.3.1 in the main body of this report).

Greg Larkin and Andy Hicks
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Invercargill

Friday 18th May 2012

RE: PCLake applied to Waituna Lagoon

MEMO

It is the opinion of the scientists involved in the Waituna Lagoon project at the University of Waikato that PCLake is unsuitable for Waituna Lagoon (Appendix A). The *Ruppia* model is purpose built for Waituna Lagoon to predict changes in *Ruppia* biomass as a function of light climate, temperature, salinity and water level (which are all output from DYRESM-CAEDYM). Thus, we should be able to predict the effect of management options (nutrient load reductions, opening timings/duration) on *Ruppia* growth. We realise that the *Ruppia* model operating independently may not provide dynamic feedbacks between *Ruppia* and water column turbidity. We hope to address this by en explicit representation of *Ruppia* in our ecological models (DYRESM-CAEDYM and ELCOM-CAEDYM) and will provide progress on this aspect in the upcoming 1-2 weeks.

With respect to the model scenarios to be provided through the lagoon Technical Advisory Group, we thought it would be useful for you to know exactly what the model can and can't do:

1. We can simulate openings to happen at specific times at specific water levels, for the openings to last a specific period of time, or at specific chlorophyll *a* thresholds (by iteration).). We can also simulate for the lagoon to never be opened and for water loss to occur through barrier seepage and evaporation only.
2. We can simulate nutrient load reductions/increases (likely associated with land use change/management options). We can further improve on this by linking with SWAT when it is up and running (i.e., when the processed LIDAR data is available). However, it should be possible to run some scenarios without linking to SWAT. Perhaps, some extreme cases (e.g. all native, all dairy) would be good to determine the sensitivity of the model/system.
3. We can simulate changes in climate. If you want this included, we require climate change projections for Southland. It is also possible for us to generate those changes in association with a

climate change specialist, but this is not possible within the time constraints of the current project. This climate change scenario would involve alterations of air temperature, rainfall and humidity (but not wind since climate change models do not explicitly output this variable).

We can simulate any combination of the above, but setting up the input data for this is not trivial as we have several levels of complexity - nutrient loading, climate, and lagoon opening timing and duration. The timeframe for running scenarios depends on the number and intricacy of scenarios requested and will be evaluated once requested.

Over the coming week (21-26th May) we will run at least one example scenario to illustrate the output from the DYRESM-CAEDYM and *Ruppia* models.

Kind regards,

David Hamilton

APPENDIX A

PCLake applied to Waituna Lagoon

A PCLake simulation was undertaken for Waituna Lagoon for the period 1st January 2002 – 31st December 2010. PCLake allows for inclusion of three phytoplankton groups (defined as blue-green algae, green algae, and diatoms) and one macrophyte group. Input into the model included inflow volume and temperature, and suspended sediment, NH_4-N, NO_3-N, organic N, PO_4-P, and organic P loads (derived from data provided by ES), as well as solar radiation and wind speed (derived from climate data from Tiwai Point climate station), and evaporation and outflow volume (calculated as for the DYRESM-CAEDYM model).

As part of the calibration process, the model simulation was compared with field data from 2002- 2007, (leaving 2008-2010 for validation). After extensive effort at calibrating PCLake for Waituna Lagoon, the model was able to simulate water level, dissolved oxygen and nitrogen concentrations fairly well, but suspended sediment, phosphate and total phosphorus concentrations and total chlorophyll *a* compared poorly with field data (Table A1; Figure A1). Furthermore, comparison of PCLake output with output from the 1D coupled hydrodynamic-ecological model DYRESM-CAEDYM suggests that the latter is performing better at simulating most calibrated variables (in addition to salinity, which is not included in PCLake). Obviously, the advantage of PCLake (and one of the reasons it was included in the original proposal) is that it allows simulation of a freshwater macrophyte group, but we have identified several reasons why PCLake might not be an ideal model for simulating macrophyte biomass in Waituna Lagoon (in addition to the overall poor model performance when compared with DYRESM-CAEDYM).

PCLake is a zero-dimensional model which assumes an average lagoon area with varying depth. Conceptually, this is a problem for a system such as Waituna that experiences such vast changes in water level and surface area. Specifically, this is a problem for simulating the effect of water level changes on macrophytes (*Ruppia*) as there is no way to force the model to “dry out” areas where the *Ruppia* is likely to be (i.e. those areas that are above mean sea level (Johnson & Partridge 1998; Stevens & Robertson 2007). Furthermore, currently, there is no option to include salinity as a state variable in the model (though model creator/developer Jan Janse has indicated that this is in development). For modelling phytoplankton and *Ruppia* dynamics in Waituna (both of which are likely to be significantly affected by changes in salinity) this is less than ideal. Rather than ignoring these issues and continuing on with PCLake regardless, we have developed an alternative: a macrophyte model (currently parameterised for *Ruppia*, but we will consider the possibility of including other macrophyte species) that uses daily output from DYRESM-CAEDYM to simulate macrophyte vegetation and seed biomass. We may also be able to couple this alternative model to 3D output from ELCOM-CAEDYM (although, it should be recognised that for long term scenario exploration we will be using the 1D model due to the prohibitively long model run times for the 3D model).

Alternative *Ruppia* model

In PCLake changes in macrophyte biomass are calculated as a function of water temperature, nutrient, and light availability. In our macrophyte model, by using output from DYRESM-CAEDYM of water level, water temperature, salinity, nutrient concentrations, light extinction (derived from phytoplankton biomass, suspended sediment and DOC concentrations), we can apply similar limitation functions (i.e. temperature, light, nutrients) plus account for limitation in vegetative growth due to salinity and desiccation (as our model resolves in the vertical dimension, whereas PCLake does not). Furthermore, inclusion of a seed biomass pool in our model allows us to include a limitation function based on seed availability.

Possible limitations

The alternative macrophyte model will respond to the conditions simulated by CAEDYM, but there will be no feedback back into CAEDYM from the macrophyte model (i.e. the effect of increased macrophyte biomass on phytoplankton growth and sediment resuspension). We recognise this may be a problem and will work on a solution. For example, we will explore options to quantify nutrient uptake by macrophytes, and the effect of macrophyte biomass on sediment resuspension, which may then be fed into long term simulations in CAEDYM.

References

Johnson P.N. & Partridge T.R. (1998) Vegetation and water level regime at Waituna Lagoon , Southland. Science for Conservation Series; 98, Department of Conservation, Wellington, NZ.

Stevens L. & Robertson B. (2007) Waituna Lagoon 2007: Broad scale habitat mapping and historical sediment coring. Prepared by Wriggle Ltd for Environment Southland.

Table A1: Comparison of statistical measures of model performance, (Pearson R and normalised root mean square error), for PCLake and DYRESM-CAEDYM calibrations. Highest R, or lowest NRMSE, indicating better model performance, are highlighted in bold for each variable.

Variable	PCLake		DYRESM-CAEDYM	
	R	NRMSE	R	NRMSE
Water level (m)	**0.99**	0.23	0.98	**0.12**
Temperature (ºC)	Not provided as output in PCLake		**0.94**	**0.13**
Salinity	Not included in PCLake		**0.97**	**0.20**
Dissolved oxygen (mg L^{-1})	**0.66**	0.18	0.52	**0.13**
Phosphate (mg P L^{-1})	0.02	1.55	**0.11**	**1.28**
Total phosphorus (mg L^{-1})	0.07	0.77	**0.29**	**0.73**
Nitrate (mg N L^{-1})	0.84	0.81	**0.89**	**0.67**
Ammonium (mg N L^{-1})	0.28	**1.00**	**0.50**	1.80
Total nitrogen (mg L^{-1})	0.84	0.62	**0.85**	**0.39**
Chlorophyll *a* (µg L^{-1})	0.18	**1.29**	**0.36**	1.35
Total suspended solids (mg L^{-1})	0.15	1.07	**0.56**	**0.82**

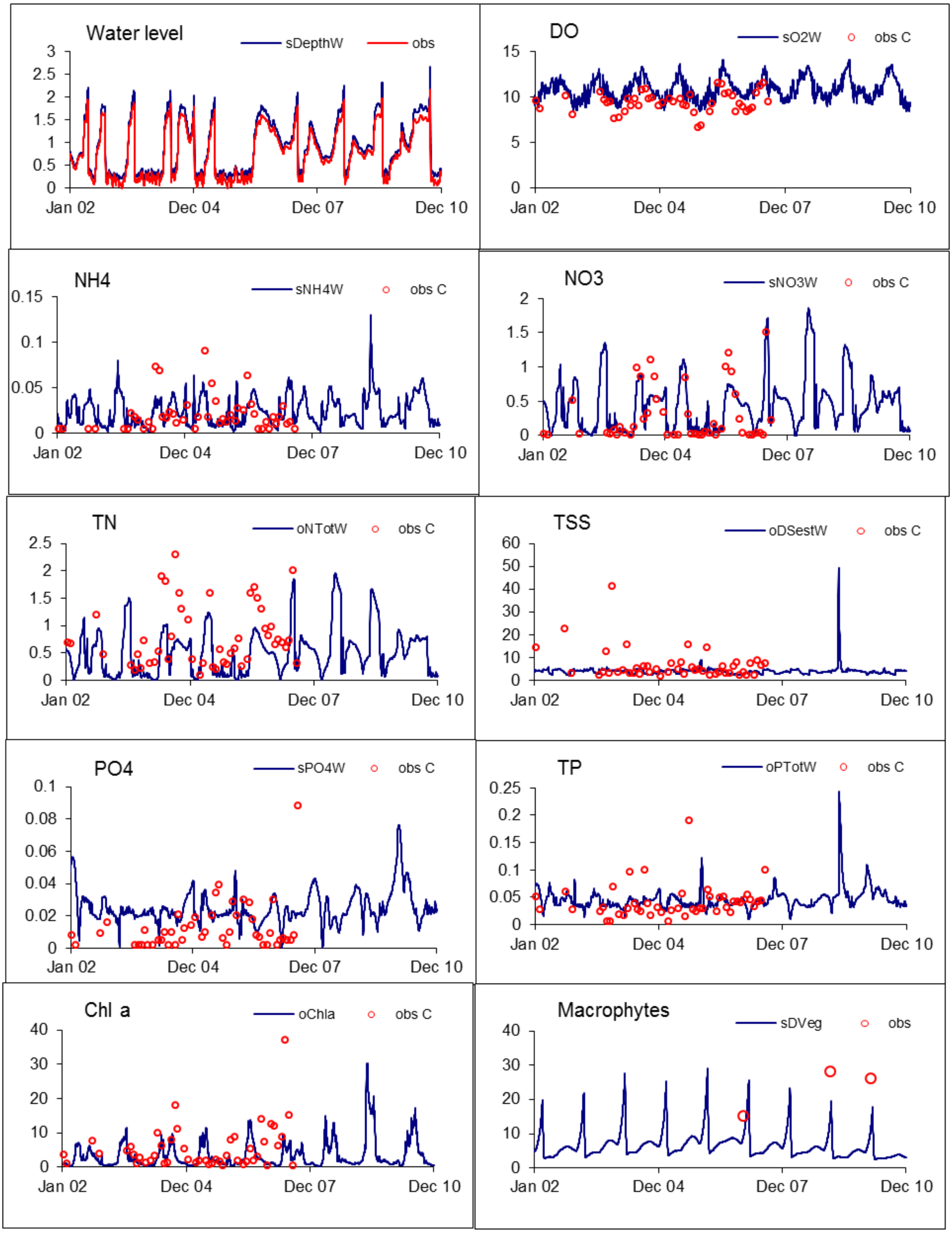

Figure A1: Comparison of PCLake simulation results (solid blue line) with field data (red circles) for the calibration period (2002-2007, except for macrophytes for which only one survey was undertaken between 2002 and 2007, so surveys in 2009 and 2010 are also plotted). N.B. Field data for other variables from 2008 – 2010 were reserved for model validation.

Appendix 2

Parameters used in DYRESM-CAEDYM for Waituna Lagoon

Table A2.1: Parameters used in DYRESM for Waituna Lagoon

Parameter	Value	Unit	Reference/notes
Critical wind speed	3.0	$m\ s^{-1}$	
Emissivity of water surface	0.96	-	
Mean albedo of water	0.1	-	
Potential energy mixing efficiency	0.2	-	
Shear production efficiency	0.30	-	
Vertical mixing coefficient	200	-	
Wind stirring efficiency	0.5	-	
Effective surface area coefficient	1.3×10^{7}	m^{-2}	

Table A2.2: Light parameters used in CAEDYM for Waituna Lagoon

Parameter	Description	Value	Units	Reference/notes
Extinction coefficients				
KdNIR	Near infrared extinction coefficient	1	m^{-1}	
BaseKd	PAR extinction coefficient	0.4	m^{-1}	
KdUVA	Ultra Violet A extinction coefficient	1	m^{-1}	
KdUVB	Ultra Violet B extinction coefficient	2.5	m^{-1}	
Fraction of incident short wave radiation				
%NIR	NIR fraction of short wave radiation	0.510		
%PAR	PAR fraction of short wave radiation	0.450		
%UVA	UVA fraction of short wave radiation	0.035		
%UVB	UVB fraction of short wave radiation	0.005		

Table A2.3: Sediment parameters used in CAEDYM for Waituna Lagoon

Parameter	Description	Value	Units	Reference/notes
Inorganic particle constants				
deSS	Density of suspended solid particles	2650	kg m^{-3}	
diaSS	Diameter of suspended solid particles	3	µm	
KeSS	Specific attenuation coefficient of suspended solids	0.05	mg L^{-1} m^{-1}	
tcSS	Critical shear stress of suspended solids	0.05	N m^{-2}	
Static sediment constants				
vSed	Temperature multiplier of sediment fluxes	1.06		
Sediment oxygen demand				
rSOs	Static sediment exchange rate	1.30	g m^{-2} day^{-1}	
KSOs	Half sat constant for DO sediment flux	0.50	mg L^{-1}	
Nutrient fluxes				
SmpPO4	Release rate of PO_4	0.0002	g m^{-2} day^{-1}	(Fredericks & Heggie 1999)
KOxS-PO4	Half sat constant for PO_4 sediment flux	0.0200	mg L^{-1}	
SmpNH4	Release rate of NH_4	0.0010	g m^{-2} day^{-1}	(Jones *et al.* 2011)
KDOS-NH4	Half sat constant for NH_4 sediment flux	0.0500	mg L^{-1}	
SmpNO3	Release rate of NO_3	-0.0200	g m^{-2} day^{-1}	(Jones *et al.* 2011)
KDOS-NO3	Half sat constant for NO_3 sediment flux	0.5000	mg L^{-1}	
SmpSi	Release rate of Si	0.0385	g m^{-2} day^{-1}	
KDOS-Si	Half sat constant for Si sediment flux	0.5000	mg L^{-1}	
Sediment composition				
sedOrganicFrac	Fraction of sediment that is organics	0.1		
SedPorosity	Sediment porosity	0.1		
resusRate	Composite resuspension rate	0.095	g m^{-2} day^{-1}	
resusKT	Resuspension rate half sat constant	1E+08		

Table A2.4: Nutrient cycling parameters used in CAEDYM for Waituna Lagoon

Parameter	Description	Value	Units	Reference/notes
Organic particles (POM)				
POC1max	Max transfer of POCL-DOCL	0.010	day^{-1}	
POC2max	Max transfer of POCR-DOCR	0.005	day^{-1}	
POP1max	Max transfer of POPL-DOPL	0.005	day^{-1}	
POP2max	Max transfer of POPR-DOPR	0.005	day^{-1}	
PON1max	Max transfer of PONL-DONL	0.004	day^{-1}	
PON2max	Max transfer of PONR-DONR	0.005	day^{-1}	
POMDia1	Diameter of POM particles (labile)	9	µm	
POMDia2	Diameter of POM particles (refractory)	50	µm	
POMDensity1	Density of POM particles (labile)	1100	kg m^{-3}	
POMDensity2	Density of POM particles (refractory)	1100	kg m^{-3}	
tcPOM1	Critical shear stress for POM (labile)	0.07	N m^{-2}	
tcPOM2	Critical shear stress for POM (refractory)	0.07	N m^{-2}	
KePOC1	Specific attenuation coefficient of POM (labile)	0.01	mg L^{-1} m^{-1}	(Hanson *et al.* 2011)
KePOC2	Specific attenuation coefficient of POM (refractory)	0.02	mg L^{-1} m^{-1}	(Hanson *et al.* 2011)
Dissolved organics				
DOC1max	Max mineralisaton of DOCL-DIC	0.002	day^{-1}	
DOC2max	Mac mineralisation of DOCR-DIC	0.001	day^{-1}	
DOD1max	Max mineralisaton of DOPL-PO_4	0.050	day^{-1}	
DOD2max	Max mineralisation of DOPR-PO_4	0.001	day^{-1}	
DON1max	Max mineralisation of DONL-NH_4	0.018	day^{-1}	
DON2max	Max mineralistion of DONR-NH_4	0.001	day^{-1}	
KeDOC1	Specific attenuation coefficient of DOC (labile)	0.01	mg L^{-1} m^{-1}	(Hanson *et al.* 2011)
KeDOC2	Specific attenuation coefficient of DOC (refractory)	0.01	mg L^{-1} m^{-1}	(Hanson *et al.* 2011)
SminDOC	Min salinity for DOC flocculation	1.00		
SmaxDOC	Min salinity for DOC flocculation	10.00		
kfloc	Flocculation rate constant	0.50		
kSWNP	Rate of DOCR photolytic decay	0.001		
Dissolved inorganics				
Kadd1PO4	PO_4 adsorption constant	0.50		
Kadd2PO4	PO_4 desorption constant	0.50		
vN2	Temperature multiplier for denitrification	1.07		
KoN2	Denitrification rate coefficient	0.08	day^{-1}	
KN2	Half sat constant for denitrification	0.50	mg L^{-1}	
vON	Temperature multiplier for nitrification	1.07		
KoNH	Nitrification rate coefficient	0.10	day^{-1}	
KON	Half sat constant for nitrification	0.50	mg L^{-1}	
YNH	Ratio of O_2 to N for nitrification	3.4286	mg N (mg O)$^{-1}$	

Table A2.5: Phytoplankton parameters used in CAEDYM for Waituna Lagoon (cyanophytes, cryptophytes, diatoms)

Parameter	Description	Value	Units	Reference/notes
Pmax	Maximum growth rate	0.7, 1.1, 1.5	day^{-1}	(Reynolds 2006)
Ycc	Ratio of C to chl *a*	45, 50, 30	mg C (mg chl *a*)$^{-1}$	(Geider 1987; Gallegos & Vant 1996; Geider, MacIntyre & Kana 1997)
IK	Parameter for initial slope of P/I curve	115, 60, 20	µmol m^{-2} s^{-1}	
Kep	Specific attenuation coefficient of phytoplankton	0.025, 0.025, 0.025	µg chl a L^{-1} m^{-1}	
Nutrient parameters				
KP	Half saturation constant for phosphorus	0.0075, 0.003, 0.0025	mg L^{-1}	
KN	Half saturation constant for nitrogen	0.05, 0.021, 0.018	mg L^{-1}	
KSi	Half saturation constant for silica	0.20	mg L^{-1}	Diatoms only
UNmax	Maximum rate of phytoplankton nitrogen uptake	1.125, 1.5, 1.05	mg N (mg chl *a*)$^{-1}$ day^{-1}	
UPmax	Maximum rate of phytoplankton phosphorus uptake	0.16, 0.22, 0.15	mg P (mg chl *a*)$^{-1}$ day^{-1}	
Temperature limitation				
vT	Temperature multiplier for phytoplankton growth	1.08, 1.07, 1.06		
Tsta	Standard temperature	22, 18, 16	º C	
Topt	Optimum temperature	28, 24, 22	º C	
Tmax	Maximum temperature	35, 32, 30	º C	
Respiration, mortality and excretion				
kr	Respiration rate coefficient	0.08, 0.12, 0.16	day^{-1}	
vR	Temperature multiplier for phytoplankton respiration	1.07, 1.07, 1.07		
fres	Fraction of respiration relative to total metabolic loss rate	0.7, 0.7, 0.7		
fdom	Fraction of metabolic loss rate that goes to DOM	0.5, 0.5, 0.5		
Salinity limitation				
maxSP	Maximum salinity	36, 36, 36		
phsal	Type of water environment	0, 0, 2		0 = freshwater, 2 = estuarine
Sopt	Optimum salinity/minimum bound of salinity tolerance	3, 10, 20		
Bep	Salinity limitation vale at S=0 and S=max SP	8, 5, 1		
Settling and resuspension				
ws	Constant settling velocity	-0.6E-6, -0.23E-5, -0.2E-4	m s^{-1}	
tcpy	Critical shear stress for phytoplankton	0.05, 0.05, 0.05	N m^{-2}	(Beaulieu 2003)
alpPy	Resuspension rate constant	0.000033	mg chl *a* m^{-2} s^{-1}	
Dtphy	Phytoplankton sediment survival time	1, 1, 10	days	

Table A2.6: Macrophyte (*Ruppia*) parameters used in CAEDYM for Waituna Lagoon

Parameter	Description	Value	Units	Reference/notes
Vmax	Maximum growth rate	0.05	day^{-1}	(Baird *et al.* 2003; Everett, Baird & Suthers 2007; Haese & Pronk 2011)
IK	Parameter for initial slope of P/I curve	200	µmol m^{-2} s^{-1}	(Qu *et al.* 2004; Haese & Pronk 2011)
kB	Respiration rate coefficient	0.0055	day^{-1}	(Haese & Pronk 2011)
vB	Temperature multiplier for macrophyte respiration	1.05		
RV_{max}	Maximum *Ruppia* biomass (veg)	120	g C m^{-2}	(Carruthers *et al.* 1999; Ierodiaconou & Laurenson 2002; Haese & Pronk 2011)
Temperature limitation				
vT	Temperature multiplier for macrophyte growth	1.09		
Tsta	Standard temperature	20	ºC	
Topt	Optimum temperature	26	ºC	
Tmax	Maximum temperature	32	ºC	
Salinity limitation				
Smax	Maximum potential salinity	36		
Sopt	Optimum salinity/minimum bound of salinity tolerance	8		(Gerbeaux 1989)
Bep	Salinity limitation value at Smax	1.6		

Table A2.7: Macroalgae parameters used in CAEDYM for Waituna Lagoon

Parameter	Description	Value	Units	Reference/notes
Vmax	Maximum growth rate	0.20	day^{-1}	(Giusti & Marsili-Libelli 2005)
Ycc	Ratio of C to chl *a*	50	mg C (mg chl *a*)$^{-1}$	
IK	Parameter for initial slope of P/I curve (no photoinhibition)	70	µE m^{-2} s^{-1}	
Hmac	Conversion macroalgae biomass to height	0.01	m (g C m^{-2})$^{-1}$	
Kmac	Specific attenuation coefficient of macroalgae	0.125	g C m^{-2} m^{-1}	(Krause-Jensen & Sand-Jensen 1998)
Nutrient parameters				
KP	Half saturation constant for phosphorus	0.035	mg L^{-1}	
KN	Half saturation constant for nitrogen	0.25	mg L^{-1}	(Giusti & Marsili-Libelli 2005)
INmin	Minimum internal N concentration	0.003	g N (mg chl *a*)$^{-1}$	
INmax	Maximum internal N concentration	0.02	g N (mg chl *a*)$^{-1}$	
UNmax	Maximum rate of macroalgae nitrogen uptake	0.001	g N (mg chl *a*)$^{-1}$ day^{-1}	(Giusti & Marsili-Libelli 2005)
IPmin	Minimum internal P concentration	0.00045	g P (mg chl *a*)$^{-1}$	
IPmax	Maximum internal P concentration	0.0002	g P (mg chl *a*)$^{-1}$	
UPmax	Maximum rate of macroalgae phosphorus uptake	0.00014	g P (mg chl *a*)$^{-1}$ day^{-1}	
Temperature limitation				
vT	Temperature multiplier for macroalgae growth			
Tsta	Standard temperature	1.08	ºC	
Topt	Optimum temperature	20	ºC	
Tmax	Maximum temperature	33	ºC	
Respiration, mortality and excretion		39		
kB	Respiration rate coefficient	0.0268	day^{-1}	
vB	Temperature multiplier for macroalgae respiration	1.07		
fres	Fraction of respiration relative to total metabolic loss rate	0.75		
Salinity limitation				
maxSP	Maximum potential salinity	26		
Sopt	Optimum salinity/minimum bound of salinity tolerance	20		
Bep	Salinity limitation vale at S=0 and S=max SP	1		No salinity limitation

Appendix 3

Table A3.1: Selected parameters for the sensitivity analysis

Parameter	Abbreviation	Species	Calibrated value	+10%	-10%
Maximum potential growth rate of phytoplankton	Pmax	CYANO	0.7	0.77	0.63
Maximum potential growth rate of phytoplankton	Pmax	CRYPT	1.1	1.21	0.99
Maximum potential growth rate of phytoplankton	Pmax	MDIAT	1.5	1.65	1.35
Maximum rate of phytoplankton nitrogen uptake	UNmax	CYANO	1.125	1.2375	1.0125
Maximum rate of phytoplankton nitrogen uptake	UNmax	CRYPT	1.5	1.65	1.35
Maximum rate of phytoplankton nitrogen uptake	UNmax	MDIAT	1.05	1.155	0.945
Maximum rate of phytoplankton phosphorus uptake	UPmax	CYANO	0.16	0.176	0.144
Maximum rate of phytoplankton phosphorus uptake	UPmax	CRYPT	0.22	0.242	0.198
Maximum rate of phytoplankton phosphorus uptake	UPmax	MDIAT	0.15	0.165	0.135
Respiration rate coefficient phytoplankton	kr	CYANO	0.08	0.088	0.072
Respiration rate coefficient phytoplankton	kr	CRYPT	0.12	0.132	0.108
Respiration rate coefficient phytoplankton	kr	MDIAT	0.16	0.176	0.144
Constant settling velocity phytoplankton	ws	CYANO	-6.00E-07	-6.6E-07	-5.4E-07
Constant settling velocity phytoplankton	ws	CRYPT	-2.30E-06	-2.53E-06	-2.07E-06
Constant settling velocity phytoplankton	ws	MDIAT	-2.00E-05	-0.000022	-0.000018
Resuspension rate constant (phytoplankton)	alpPy	-	0.000033	0.0000363	0.0000297
Maximum growth rate of *Ruppia*	Vmax	*Ruppia*	0.05	0.055	0.045
Respiration rate coefficient *Ruppia*	kb	*Ruppia*	0.0055	0.00605	0.00495
Maximum growth rate of macroalgae	Vmax	Macroalgae	0.2	0.22	0.18
Maximum rate of macroalgae nitrogen uptake	UNmax	Macroalgae	0.001	0.0011	0.0009
Maximum rate of macroalgae phosphorus uptake	UPmax	Macroalgae	0.00014	0.000154	0.000126
Respiration rate coefficient macroalgae	kb	Macroalgae	0.0268	0.02948	0.02412
Composite resuspension rate	resusRate	-	0.095	0.1045	0.0855

Appendix 4

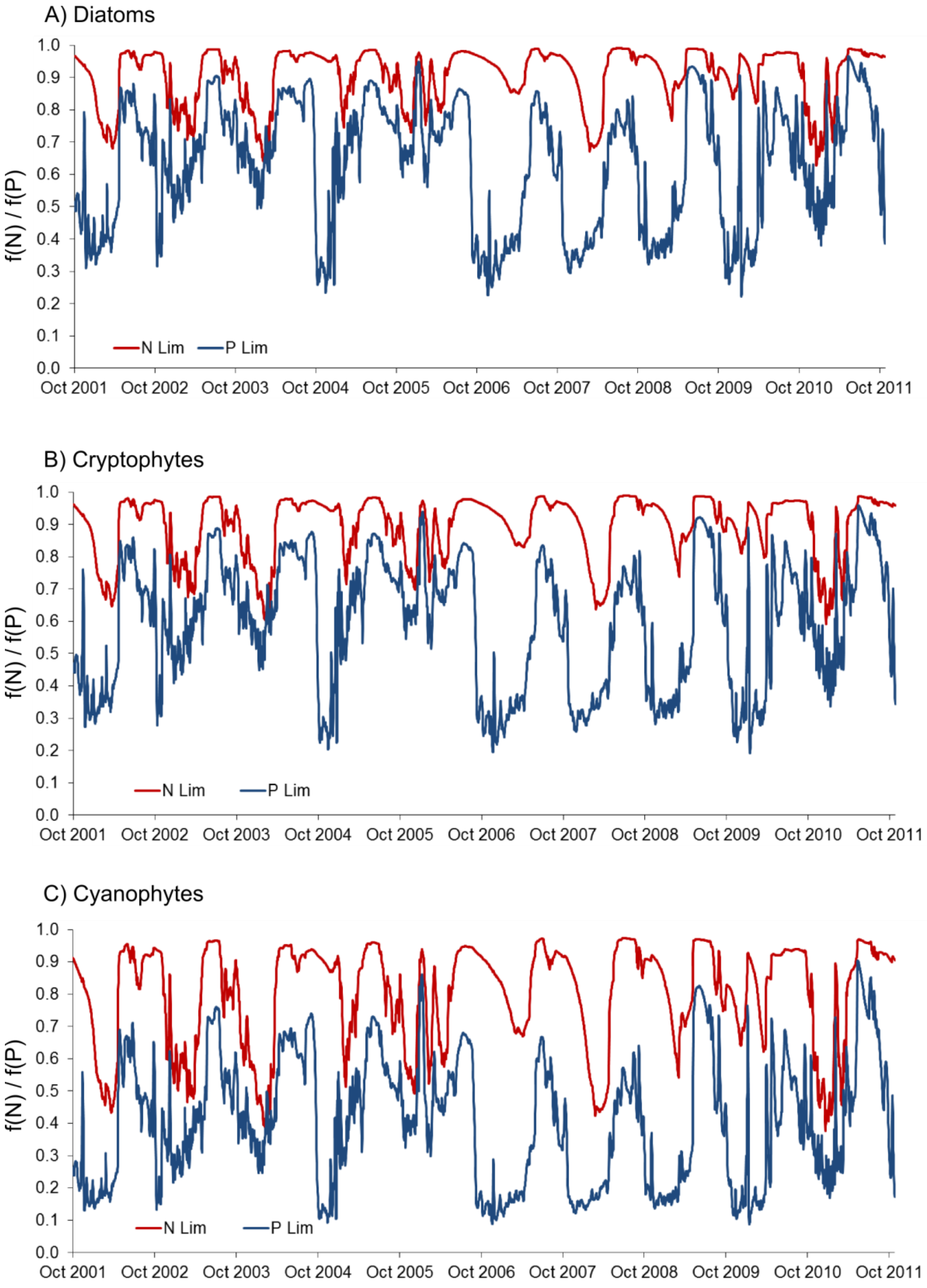

Figure A4.1: DYRESM-CAEDYM output - Nitrogen and phosphorus limitation for simulated phytoplankton groups, (A) diatoms, B) cryptophytes and C) cyanophytes), for the base scenario (i.e. calibration/validation period).

Appendix 5

Waituna Lagoon Modelling Timeline

Approved by Jane Kitson (30/11/11)
- 5th December: Project commencement;
- 5th December - 20th January: Data collation (will continue until the end of the project) and preliminary report by 20th January, to indicate data quality, quantity, suitability and timeliness of provision for each model.
- 20th January to 20th April: preparation and formatting of input data for models: ECOPATH, ELCOM and DYRESM, PCLake and SWAT. Preliminary model runs of each.
- Around late April or early May: progress report with meeting of modellers with relevant review panels (e.g., might include Kitson, Robertson and Scanes); selection of model scenarios.
- 20th April to 22nd June: model calibration, running scenarios.
- 30th June: Final report.

Task	Contract Date	Delivery Date + comments
Data collation (will continue until the end of the project) and preliminary report by 20th January, to indicate data quality, quantity, suitability and timeliness of provision for each model	5th December 2011 – 20th January 2012	16th January 2012 Memo - Catchment modelling for Waituna Lagoon – developed in response to a justification for the specific catchment modelling approach that has been adopted for Waituna Lagoon. The memo addressed a general concern of the Waituna catchment group about the use of the Surface Water Assessment Tool (SWAT) to support the catchment modelling which in turn supports the lagoon modelling. 20th January 2012 Preliminary report on the Waituna Lagoon Modelling indicating data quality, quantity, suitability and timeliness of provision for each model. 2nd April 2012 Update on the data timeline for the Waituna Lagoon Modelling project.
Preparation and formatting of input data for models: ECOPATH, ELCOM and DYRESM, PCLake and SWAT. Preliminary model runs of each.	20th January 2012 to 20th April 2012	No deliverable due

Progress report with meeting of modellers with relevant review panels (e.g., might include Kitson, Robertson and Scanes); selection of model scenarios.	Late April or early May 2012	Friday 4th May 2012 Conference call between UoW and Environment Southland Hannah Jones: 1D model 3D model *Ruppia* model Deniz Ozkundakci: Catchment model Mat Allan: Remote sensing Discussion: Model scenarios Monday 7th May 2012 Memo regarding PCLake applied to Waituna Lagoon Friday 18th May 2012 Further memo regarding PCLake applied to Waituna Lagoon
Model calibration, running scenarios	20th April to 22nd June 2012	Monday 16 July 2012 Lagoon Technical Group update with scenarios sent to UoW from Greg Larkin Sunday 22 July 2012 Waituna Lagoon DYRESM-CAEDYM model application
Report due	30th June 2012	10th August
Revised report (including further scenarios) LTG meeting (20th Sept)	By 24th September	Meeting with UoW, Karen Wilson, Greg Ryder and Barry Robertson on 21st August 2012 to discuss model output and further scenarios. Revised report: 17th September