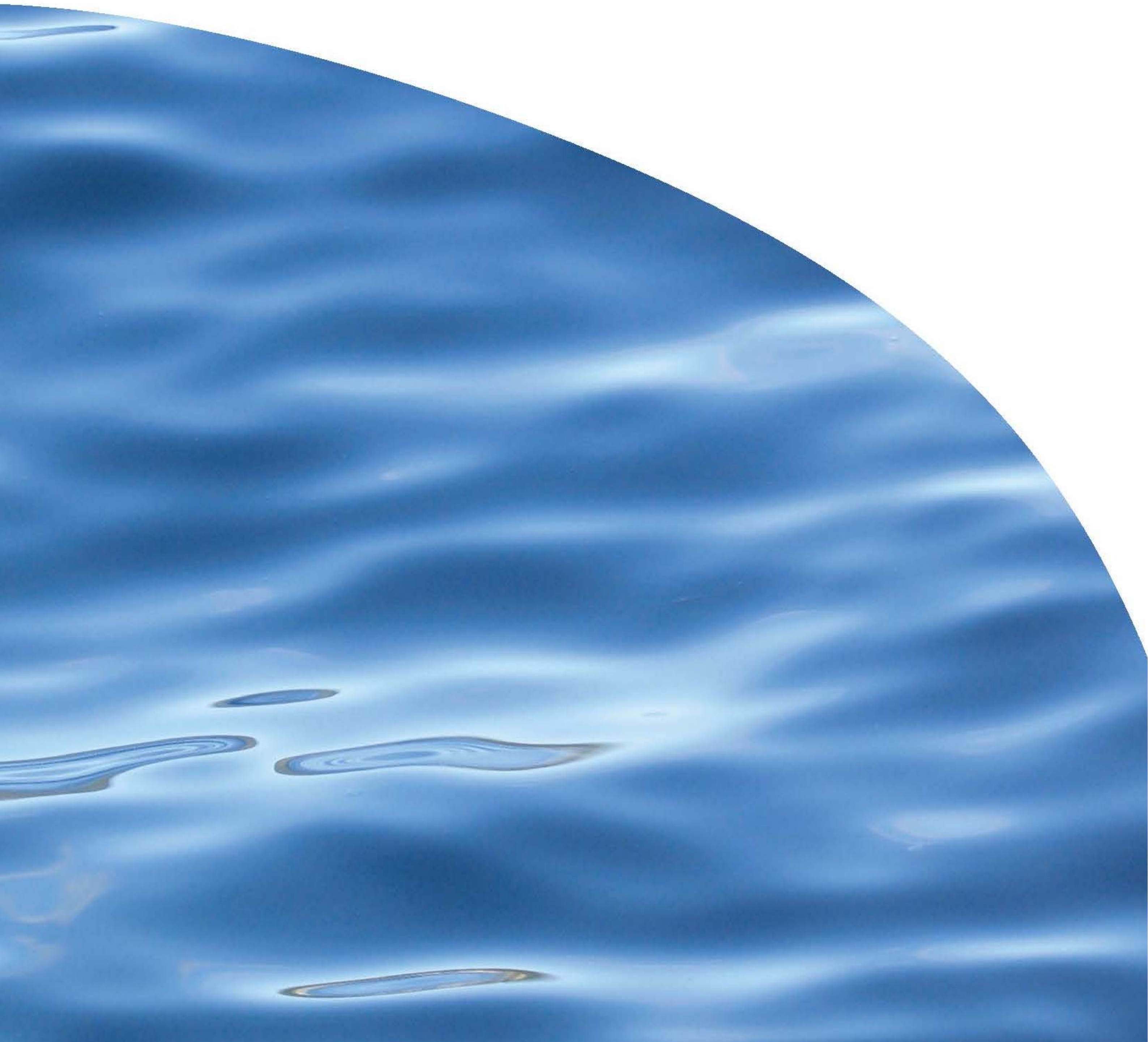
REPORT NO. 2423

AQUACULTURE EFFECTS MODELLING IN THE HAURAKI GULF



AQUACULTURE EFFECTS MODELLING IN THE HAURAKI GULF

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Prepared for Waikato Regional Council

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ISSUE DATE: 9 January 2014

RECOMMENDED CITATION: Knight BR, Keeley NB, Taylor D 2014. Aquaculture effects modelling in the Hauraki Gulf. Prepared for Waikato Regional Council . Cawthron Report No. 2423. 36 p. plus appendices.

EXECUTIVE SUMMARY

Waikato Regional Council (WRC) is undergoing development of a Marine Management Model (MMM) for the eastern Waikato coastal marine area (covering the Firth of Thames and wider Hauraki Gulf) that will assist in addressing a range of resource management issues, such as aquaculture development, biosecurity risks, and oil spill response. The first stage of the MMM has included the construction and calibration of an underlying 3-D hydrodynamic model and production of 2-year hindcast datasets on hydrodynamic conditions. This report represents the completion of stage one and focuses on the application of models that utilise the hindcast datasets to in turn forecast potential seabed and water-column effects of finfish aquaculture in the Wilson Bay and Coromandel Marine Farming Zones. The models and outputs described in this report have been made accessible to WRC staff and are intended to assist in the guidance in the consenting and monitoring of aquaculture developments as well as wider state of the environment monitoring in the Waikato coastal marine area.

Finfish aquaculture (*e.g.* kingfish, hapuka) is likely to be introduced in the near future in the Wilson Bay Marine Farming Zone within the Firth of Thames, and more extensively in the designated Coromandel Marine Farming Zone (CMFZ) in the Hauraki Gulf. The potential extent of effects that may arise from finfish aquaculture and the addition of artificial feed (nutrients) to the environment are relatively unknown for the region and hence is the focus of the initial modelling efforts. The following objectives are addressed:

- Provide a brief background on finfish aquaculture effects and approaches to forecasting effects using modelling tools;
- Describe how the modelling tools can be used in the assessment process for proposed developments;
- Describe the seabed and water-column effect modelling tools, their configuration, strengths and limitations;
- Configure and run the models based on several scenarios and present example results.

There are many different types of potential ecological effects associated with aquaculture, ranging from direct effects on the seabed to biosecurity risks to interactions with marine mammals and wild fish. The types of effects that can be quantified are conducive to modelling applications aimed at forecasting potential effects under a range of development scenarios. In the case of finfish aquaculture, these include effects of organic deposition on the seabed beneath and around the farms and the transport of nutrients in the water column and wider environment.

Seabed effects immediately beneath and in close proximity to finfish farm structures can be reliably quantified using a variety of established indicators (sediment chemistry and biotic) which can then be used to determine an overall enrichment stage (ES). The size and

intensity of organic enrichment gradients around farms (commonly referred to as a depositional footprint) is strongly influenced by water depth and current speeds, which together constitute the dispersive properties of a site.

Simple methods and tools are provided to extract bathymetric and hydrodynamic data from the hindcast datasets, and together with information on cage configuration and feed inputs, use a 2-D depositional model (DEPOMOD) to predict depositional flux to the seabed. DEPOMOD outputs are then used to compare differences in the level of solids deposition or flux among modelled locations. These outputs can be used to estimate ES and determine whether initial feed levels fall within pre-determined acceptable limits.

In addition to enrichment of the seabed, feed-added aquaculture results in the addition of dissolved nutrients into the water column, which can be transported long distances from farms and influence important biological processes such as phytoplankton production. As a first step in modelling water-column effects, the transport of nutrients was modelled by releasing passive tracers during the 2-year hydrodynamic model runs. An important aspect of the tracer modelling is that each passive tracer group is modelled individually. This means that when the passive tracer modelling runs are complete, different feed loading scenarios can be quickly combined to investigate cumulative effects from several aquaculture sites and varying river (land-use) inputs. Tracer concentration results from the model can be scaled into real-world units; in the example scenarios, the tracers were scaled to total nitrogen concentration units that are based on inputs from both finfish aquaculture (feed) and rivers.

A range of feed and weather scenarios were carried out to provide example outputs for both the seabed and water-column modelling. The model results showed that the deep, high-flow waters of the CMFZ will assist in mitigating both seabed and water-column effects. In the case of seabed effects, results indicate that the maximum feed levels used in the scenarios for the CMFZ would likely result in effects that fall below the maximum enrichment stage limit used in the Marlborough Sounds.

Model outputs utilising passive tracers assisted in visualising differences in potential nitrogen loading between finfish farms, major rivers, and the ocean. Following runs of two weeks, the surface concentrations of total nitrogen downstream of simulated fish farms were an order of magnitude lower than the surface concentrations associated with the region's major rivers. The initial results for the model also show that the Wilson's Bay Marine Farming Zone inputs have the potential to interact with riverine-sourced nitrogen inputs in the Firth of Thames. WRC's decision to limit nitrogen inputs to the region appear to be sensible given the potential for cumulative effects.

Modelling is by nature an iterative process, whereby models are continually improved and developed. The models described in this report are considered 'first stage' tools for gauging aquaculture effects associated with a range of development scenarios. It is envisioned that the hydrodynamic and aquaculture effects models will continue to be improved, and that over time, additional models contributing to the overall MMM will be further developed and expanded to encompass more complex processes (*e.g.* chemical and biotic processes, cumulative effects) and issues (biosecurity risk, coastal hazards, oil spills).

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1. INTRODUCTION

Waikato Regional Council (WRC) is undergoing development of a Marine Management Model (MMM) for the eastern Waikato coastal marine area (covering the Firth of Thames and Hauraki Gulf) that will assist in addressing a range of resource management issues, such as aquaculture development, biosecurity risks, and oil spill response. Stage one of the MMM includes several steps, including:

1. Construction and calibration of an underlying 3-D hydrodynamic model
2. Production of 2-year hindcast datasets using the hydrodynamic model
3. Use of the hindcast datasets to drive models for forecasting effects of aquaculture developments in the Wilson Bay and Coromandel Marine Farming Zones.

This report focuses on Step 3 and describes the approach and methods used to assess potential aquaculture effects utilising hindcast datasets generated from the hydrodynamic model (see Knight & Beamsley 2013).

There are additional initiatives and projects underway that align with the modelling described in this report. These projects include an ongoing initiative by Ministry for Primary Industries (MPI) to develop guidance and tools for the aquaculture industry, consenting authorities (councils) and wider stakeholders (see MPI 2013). In addition, WRC is in the process of developing a regional monitoring framework, and guidance and standards for the aquaculture industry in the Waikato region with support from MPI's Aquaculture Planning Fund. The modelling carried out as part of this report is intended to inform the development of monitoring programmes and assist in the future consenting and management of aquaculture in the Waikato coastal marine area.

1.1. Report scope and objectives

This report covers the approach and methods used to construct and apply models for assessing ecological effects of aquaculture in the Firth of Thames. The modelling described focuses on finfish (feed-added) aquaculture. In future, the models can also be applied to assess effects of shellfish aquaculture. Commercial farming of shellfish such as Greenshell™ Mussels is well established in the Waikato coastal marine area and further development of mussel farming is expected in the Wilson Bay Marine Farming Zone. Farming of fish (*e.g.* kingfish, hapuka) is likely to be introduced in the near future within the Wilson Bay farming zone that lies in the Firth of Thames and the designated Coromandel Marine Farming Zone (CMFZ) located futher seaward in the Hauraki Gulf. Fish farming involves addition of artificial feed (nutrients) to the environment. The potential extent of effects that may arise from finfish aquaculture is relatively unknown for the region and hence is the focus of the initial modelling efforts described in this report.

The methodologies, analyses and results described in this report provide an example of how the potential effects of finfish aquaculture on the seabed and water column may be assessed using the tools developed. The following objectives are addressed:

- Provide a brief background on finfish aquaculture effects and approaches to forecasting effects using modelling tools;
- Describe how the modelling tools can be used in the assessment process for proposed developments;
- Describe the seabed and water-column effect modelling tools, their configuration, strengths and limitations;
- Configure and run the models based on several scenarios and present example results.

Meeting the above objectives completes the first stage of the MMM. The performance of the tools presented in this report is reliant on the quality of the underlying hindcast datasets provided by the hydrodynamic model described in Knight and Beamsley (2013). Modelling is by nature an iterative process, whereby models are continually improved and developed. The models being applied are considered 'first stage' tools for gauging aquaculture effects associated with a range of development scenarios. It is envisioned that the hydrodynamic and aquaculture effects models will continue to be improved, and that over time, additional models (or 'modules') contributing to the overall MMM will be further developed and expanded to encompass more complex processes (*e.g.* food webs, cumulative stressors) and issues (biosecurity risk, coastal hazards, oil spills).

A key to successful implementation of the MMM and its various components is the provision of widely accessible outputs. Open accessibility to transparent modelling tools maximises their usefulness in building trust with stakeholders, guiding decision making and developing resource management and monitoring frameworks. Wherever possible, the methods and tools presented in this report have been made fully accessible to WRC through the development of simple interfaces for accessing models and their outputs.

2. BACKGROUND

There are many different types of potential ecological effects associated with aquaculture farms in the marine environment, ranging from direct effects on the seabed to biosecurity risks to interactions with marine mammals and wild fish (Figure 1). A recent, highly comprehensive review of the ecological effects of aquaculture in New Zealand has been compiled by MPI and is available as an on-line resource (MPI 2013). The wide range of effects shown if Figure 1 are all important to consider when developing aquaculture and their importance will depend on a number of factors including biological and physical characteristics of the site (*e.g.* proximity depth, currents) and the level of other activities affecting in the area. The types of effects that can be quantified are conducive to modelling applications aimed at forecasting potential effects under a range of development scenarios. In the case of finfish aquaculture, these include effects associated with the deposition of organic matter on the seabed beneath and around farm structures and the release of nutrients in the water column and wider environment.

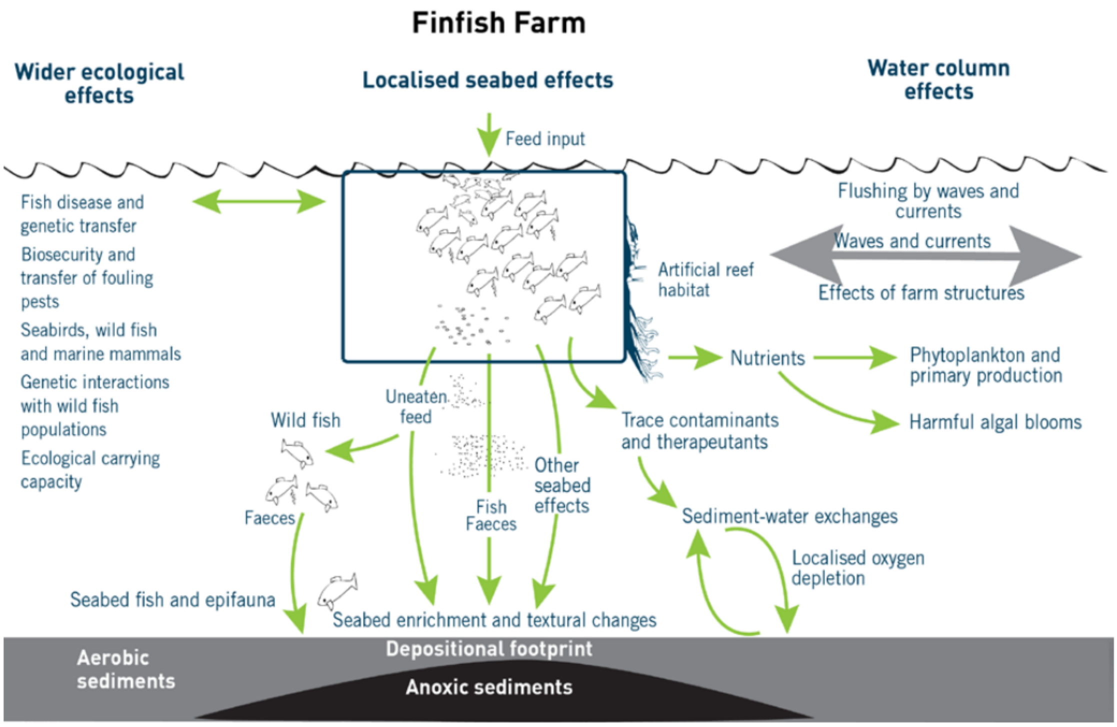

Figure 1. Schematic of ecological effects associated with finfish farms. Figure from MPI (2013).

Effects from finfish aquaculture on both the underlying seabed and the water column are heavily influenced by the extent to which the site is exposed to currents and mixing of the water column. In order to characterise hydrodynamics under a range of climate and weather conditions, a previously developed hydrodynamic model (Knight & Beamsley 2013) was used to generate high resolution, 3-D hindcast datasets of the

Hauraki Gulf over a 2-year period. The 2-year hindcast dataset is now available to drive models for assessing seabed and water-column effects under a range of environmental conditions and aquaculture scenarios (*i.e.* varying feed levels, differing locations). Ultimately, the integration of hydrodynamic data within aquaculture effects models facilitates visual forecasts of where and to what extent aquaculture wastes will be distributed in the surrounding environment. This information can then inform sustainable development and management of aquaculture at the farm scale, while also guiding the management and monitoring of wider effects (including cumulative) at the regional scale.

2.1.1. Hydrodynamic modelling

The hydrodynamic regime, including mixing of the water column and currents, heavily influences the extent of seabed and water-column effects from local to regional scales. Hydrodynamic modelling was undertaken using the Semi-implicit Eulerian-Lagrangian Finite-Element model (SELFE). The SELFE model is described in detail elsewhere (see Zhang & Baptista 2008; Knight & Beamsley 2013; or the SELFE website[1]); a brief description is provided here as background to its use in aquaculture effects models. The model domain covers the entire inner area of the Hauraki Gulf that includes the Firth of Thames (Figure 2).

The SELFE model domain has been configured with triangular elements varying in size depending on the local water depth and topographical complexity. Unstructured triangular elements make up the model domain and contain information on the average properties of the water (*e.g.* temperature, salinity) in a given volume and point in time. Subsequent physical modelling of water elevations, current speeds and directions and the resulting transport and mixing of physical properties such as temperature and salinity can be tracked within the model domain. The extent and spatial resolution of the model domain has been developed to facilitate relatively quick turn-around of scenerio modelling[2]. The model mesh consists of approximately 130,000 horizontal elements and 40 vertical layers[3]. The horizontal mesh is refined in specific areas (*e.g.* existing or potential aquaculture areas) to provide high resolution hydrodynamic model data (Figure 2).

Comparisons of model ouputs with field measurements (Dunmore *et al.* 2012) showed the hydrodynamic model performs well (Knight & Beamsley 2013). It is important to note that as the hydrodynamic model underlies any subsequent effects analysis, hydrodynamic modelling uncertainties have the potential to propogate through to effect assessments. Consequently, uncertainities idenified by Knight and Beamsley

[1] http://www.stccmop.org/knowledge_transfer/software/selfe
[2] The specified model is able to carry out 1-month simulations in ~ three days using existing high-end multi-core desktop computer technology.
[3] Vertical layers are not evenly distributed in the water column, but are configured to have higher resolution at the surface and seabed (Knight & Beamsley 2013).

(2013) and limitations presented here, should be acknowledged alongside any aquaculture effect model results. On the basis of the locations for which the hydrodynamic model was validated, we consider the model results to be acceptably accurate for consent planning and management.

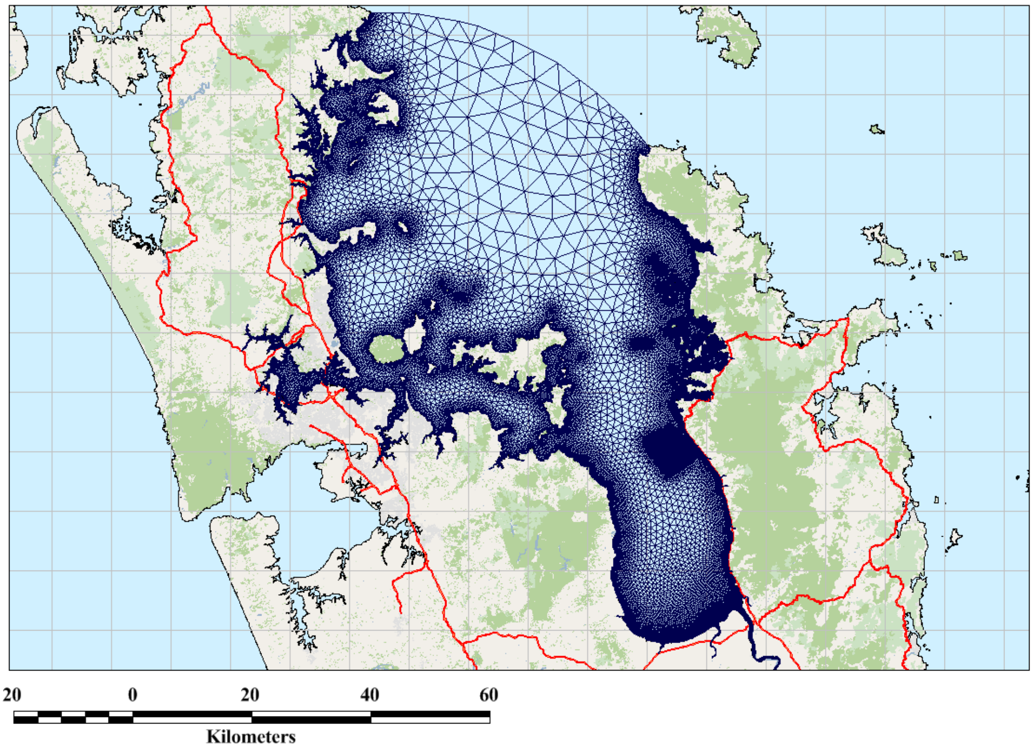

Figure 2. Unstructured finite element grid used for the SELFE model of the Hauraki Gulf and Firth of Thames. There are 129,864 elements used to generate the representation of the model domain. Densely coloured areas are those with the finest resolution.

2.1.2. Modelling seabed effects

The primary cause of seabed effects associated with finfish farms is the deposition of fish faeces and to a lesser extent, uneaten food and biofouling drop-off, which leads to over-enrichment on the seabed due to the high organic content of the deposited material. The level of seabed effects at a given site is directly related to farming intensity and, in particular, the amount of feed that has been used in the previous 6–12 months. If not managed appropriately, bio-deposition from finfish farms can change well-aerated and species-rich soft-sediments into anoxic (oxygen-depleted) zones. Anoxic zones are dominated by only a few hardy enrichment-tolerant sediment-dwelling species and under worst-case conditions the seabed may become azoic (devoid of animal life).

Seabed effects are most evident directly beneath the sea pens where highly enriched conditions are often evident, and reduce with increasing distance from the cage boundaries. At sites with relatively weak currents (*i.e.* low flow sites) natural / background seabed conditions are usually observed within a distance of about 150 m from the sea pens (Brooks *et al.* 2002, Giles 2008, Keeley *et al.* 2013b). Hence, seabed effects are considered to be reasonably localised, with measurable effects limited to local to bay-wide scale of tens to hundreds of metres.

Seabed effects in close proximity to finfish farm structures can be reliably quantified using a variety of established benthic indicators (sediment chemistry and biotic). As a result, finfish farms can be managed within consented limits set in terms of maximum allowable seafloor effects (*i.e.* ecological quality standards, EQS) that are based on quantifiable indicators and an associated overall enrichment stage (ES). The ES gradient is important because it provides a framework for categorising enrichment effects, and a common scale against which a range of environmental indicators / variables can be quantified (Keeley *et al.* 2012). The resulting empirical relationships between the environmental variables and ES can be used to reliably evaluate seabed conditions by placing them on the continuous scale from 1.0 (good) to 7.0 (bad) (*i.e.* using a continuous but bounded variable).

The size and intensity of a farm's depositional footprint is strongly influenced by a site's water depth and current speeds, which together constitute the dispersive properties of a site. Depth is important because it provides a period of time over which sinking particles may be diluted and dispersed by currents in the water column. Similarly, the strength of the currents is also important, as they can affect the transport particles further during their sinking phase. After particles have settled on the seabed, resuspension under high-flow conditions can also further disperse settled material. As a result, deep, high-flow sites are likely to have larger, but more diffuse depositional footprints than low-flow sites when farmed at a comparable intensity.

Once an acceptable ES magnitude and area of effect has been determined for an aquaculture development area, the 2-D depositional model (DEPOMOD) can be used to predict the extent of seabed enrichment under a range of feed scenarios (see Box 1). The DEPOMOD model is a proven tool for evaluating cage configurations and feeding rates and avoiding the exceedance of unacceptable ES levels. As described In Keeley *et al.* (2013a), modelled flux as estimated by DEPOMOD can be related to the ES gradient, and therefore linked to post-development monitoring for compliance.

More complex, 3-D models can be used to incorporate spatial forcing parameters and predict horizontal movements of currents to refine the depositional footprint. These models are particularly useful in more complex environments where seabed features are likely influence far-field dispersal. An increase in model complexity, however, often coincides with additional uncertainty, and the cost and extra effort involved in developing and validating these models to address this uncertainty can be significant. DEPOMOD enables a simpler approach, and one that has undergone an extensive validation process over the last 10 years. Moreover, the fundamental input parameters and processes that are used in DEPOMOD can be inbedded in spatially explicit models at a later stage if required.

Box 1: DEPOMOD

DEPOMOD (version 2.2) is a widely used and published software program designed specifically for managing fish farm wastes (Cromey & Black 2005; Magill *et al.* 2006). DEPOMOD is routinely used by the Scottish Environmental Protection Agency to set discharge consents of in-feed chemotherapeutants and in setting biomass (and thereby feed use) limits (SEPA 2005).

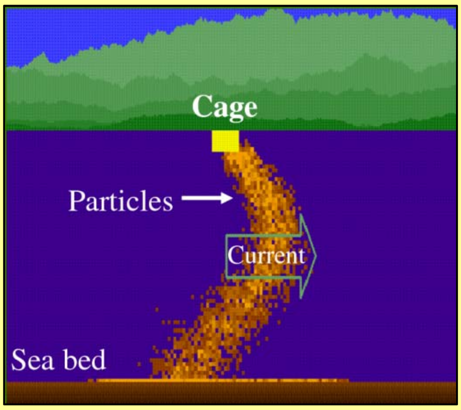

Image: Chris Cromey, SAMS

A number of processes that DEPOMOD simulates have been validated against field measurements (Cromey *et al.* 2002; Chamberlain & Stucchi 2007). Similar modelling approaches have been used in France, Norway, Ireland, Canada, Australia, Chile and South Korea. The model has also recently been adapted for use with Atlantic Cod farming in the United Kingdom. The relatively featureless, flat seabed around the Coromandel Marine Farming Zone is perfectly suited to the application of DEPOMOD. Keeley *et al.* (2013a) have demonstrated DEPOMOD to accurately predict the likely degree and spatial extent of deposition of biodeposits for salmon farms in the Marlborough Sounds, New Zealand. DEPOMOD can be applied to a variety of fish and shellfish provided estimates of appropriate species specific parameters are available. In the case of shellfish which are not fed directly, faeces and pseudo faeces production may need to be approximated by configuring appropriate parameters.

2.1.3. Modelling water-column effects

Finfish aquaculture results in the addition of dissolved nutrients into the water column, which can be transported long distances from farms and influence important biological processes such as phytoplankton production (*e.g.* Buschmann *et al.* 2007). In the Waikato coastal marine area, finfish farms would represent one of many sources of nutrient inputs to the marine environment; hence assessment of the effects of finfish farms must take into account multiple sources of nutrient inputs, including those from land-based activities. The modelling approach and methods described in this report enables the temporal and spatial patterns in nutrient transport and dispersal that may arise from various finfish farm developments to be placed within a cumulative context (*e.g.* with multiple finfish farms and river inputs combined).

As is the case for most temperature coastal environments, nitrogen (N) is the primary nutrient of concern for the Waikato Region in relation to finfish aquaculture. Availability of dissolved forms of inorganic nitrogen (DIN; nitrate [NO_3-N] and ammonium [NH_4-N]) is more likely to limit growth of primary producers such as benthic macroalgae and phytoplankton than other macronutrients such as phosphorus. When other factors such as light are not limiting, increases in DIN concentrations could therefore lead to increased growth of primary producers and carry-on effects to the ecosystem. Over time, the cumulative nutrient loading from aquaculture and other sources (*e.g.* land-based agriculture) can lead to symptoms of eutrophication (*e.g.* reduced water clarity, nuisance algal blooms, and low dissolved oxygen).

Numerical modelling assists in understanding potential water-column effects by forecasting the potential spatial extent of nutrient transport, mixing and dilution, which in turn assists in visualising areas that may be under risk of adverse effects. For this project, cumulative distribution (spread) of dissolved nutrients from potential finfish farms is simulated through release of passive tracers imbedded and released within the hydrodynamic model at locations throughout the model domain. The approach using passive tracers is considered a ‘first stage’ approach, as it is based on physical processes alone and simulates the potential dilution and flushing effects of nutrient and waste-product additions associated with finfish farming.

The ‘passive tracer’ approach has previously been applied to New Zealand King Salmon Company Ltd’s (NZKS) Environmental Protection Agency (EPA) Board of Inquiry hearing to estimate the potential water column nitrogen concentration changes from the addition of finfish farms in the Marlborough Sounds (Gillespie *et al.* 2011). Not all experts were in agreement on the complexity of modelling required to assess water-column effects, and some highlighted that, among other concerns, a passive tracer approach has a limited ability to predict likely changes to algal uptake of nutrients and subsequent nutrient distribution and food web effects. Nevertheless, a passive tracer approach provides initial transport information on two of the major physical processes (*i.e.* advection and dilution) that act to remove and redistribute wastes from aquaculture activities. The approach is able to look at the combined nutrient ‘pressure’ effects from multiple sources; therefore, the model outputs are useful for determining whether any areas of overlapping pressures may arise. Modules for full biogeochemical modelling have been developed for integration with the SELFE hydrodynamic model. It is anticipated that more complex, biogeochemical modelling can be carried out in future, as aquaculture and knowledge of the environment develops. Availability of time-series data for validating models from consent-based and wider state of the environment monitoring will be important in determining actual effects and validating both simple and complex models for improved planning in future.

3. MODELLING METHODS

The following sections describe the methods used in setting up and carrying out modelling for assessing seabed and water-column effects. The number of potential scenarios for assessing effects associated with aquaculture are numerous due to the many possible combinations of feed levels and cage configurations as well as a wide range of seasonal and weather conditions that influence hydrodynamics (currents, mixing of the water column). The scenarios selected for this report are intented only to demonstrate the utility of the models and provide examples of their outputs.

Seabed effects at the farm scale are particularly sensitive to feed levels as well as hydrodynamics of the site. For this reason, we ran three different feed level scenarios over the same time period to give an example of the process required to estimate a ‘maximum’ initial sustainable loading. An alternative scenario for the highest feed level was also run under different season and weather conditions (Table 1). Seabed modelling was carried out for a configuration of eight Polarcirkel sea pens arranged within the CMFZ.

Table 1. Scenarios modelled and their respective periods. Seabed scenarios are based on different mean annual feed levels for eight Polarcirkel sea pens in the Coromandel Marine Farming Zone (CMFZ). Water-column scenarios were based on the release of nutrients over five locations across the CMFZ and the Wilson Bay Marine Farming Zone and were multiplied by 1.5 to simulate 50% higher seasonal feeding rates.

Scenario		**Feed rate ($kT\ yr^{-1}$)**	**Description**	**Start date**	**End date**
Seabed (Farm)	1	1.2	Summer storm period	24 Feb 2009	11 Mar 2009
	2	1.8	Summer storm period	24 Feb 2009	11 Mar 2009
	3	3.1	Summer storm period	24 Feb 2009	11 Mar 2009
	4	3.1	Calm winter period	13 June 2009	28 June 2009
Water column (CMFZ)	1	14.5	‘Typical’ winter period	15 May 2012	26 July 2012
	2	14.5	Calm summer period	1 Dec2008	1 Jan2009
	3	14.5	Summer storm period	24 Feb 2009	11 Mar 2009
	4	14.5	Calm winter period	13 June 2009	28 June 2009
	5	14.5	Winter storm period	1 July 2009	15 July 2009

For modelling water-column effects, we chose to model the current recommended maximum feed inputs (N loading) in order to demonstrate the potential for investigating large-scale cumulative effects. Feed inputs typically increase above mean annual feed levels in the summer months by up to about 50%. Hence in order to capture ‘worst-case-scenarios’ over a given time period in the water column, we modelled the release of nitrogen based on the annual feed loading rate and a multiplier of 1.5[4]. Scenarios covered periods of two to ten weeks in etiher summer or winter and under calm and stormy conditions (Table 1).

The modelling methods for assessing seabed and water-column effects utilise hindcast datasets from the hydrodynamic model that is described previously in Knight and Beamsley (2013). Modelled meterological data used to drive the hydrodynamic models are shown are in Appendix 1.

3.1. Seabed effects modelling

As described in Section 2.1.2, finfish farms can be managed within consented limits set in terms of maximum allowable seafloor effects (*i.e.* ecological quality standards, EQS) that are based on quantifiable benthic indicators and the associated overall enrichment stage (ES). Here we describe how the DEPOMOD model (see Box 1) can be used to forecast levels of enrichment based on the characterstics of a site (currents, water depth) and various combinations of feed inputs and cage configurations.

3.1.1. Initial considerations

Before seabed effects can be modelled using DEPOMOD, some broader decisions need to be made regarding the species to be farmed, location and likely farm configuration. It is critical to initially select sites where the seabed is less likely to be overly impacted by finfish farming (*e.g*. deeper, high-flow sites versus shallow, low-flow sites) and that are removed from potentially sensitive and/ or highly valued marine habitats. Other important decisions concern the cage type (*e.g*. individual plastic circular pens versus concentrated blocks of square pens) and how they are to be arranged across the site. Cage type and arrangement decisions are likely to be influenced by how much space is available and the type of environment[5].

The potential intensity of the aquaculture operation may also be determined by regional operational constraints such as, nutrient or feed discharge limits[6]. In most instances, the species to be farmed will be predetermined; however, this may not be

[4] Note that this seasonal 1.5 multiplier has not been used for the seabed scenarios which use mean annual feed loadings. Mean annual feed loadings are used because annual benthic monitoring data used to relate the modelled seabed flux to benthic enrichment stage (ES) scores has been obtained from commercial farming operations which incorporate seasonal variations. Subsequently a higher level of confidence in annual seabed effects is possible based on the large quantities of data used to relate benthic modelling to effects (see e.g. Keeley *et al.* 2013).

[5] For example, square cage configurations may not be suitable in regions exposed to large waves.

[6] For example, limits of "… total net discharge of nitrogen from fed aquaculture to a maximum of 800 tonnes per year and an associated maximum of 13,600 tonnes of feed discharged per year…" are set for the Coromandel Marine Farming Zone http://www.waikatoregion.govt.nz/Council/Policy-and-plans/Rules-and-regulation/Regional-Coastal-Plan/Regional-Coastal-Plan/6-Marine-Farming/Development-of-Marine-Farming/611B/

the case for more speculative developments. Nevertheless, it is critical to select a likely candidate species in order to determine feed to solids conversion ratios, as well as other model input parameters (*e.g.* particle sinking velocities and waste feed, etc.). In the case of the examples provided here, a site has been selected in the CMFZ and the configuration is similar to a 4 x 2 grid arrangement of eight 38.2 m diameter, 20 m deep circular sea pens (as described in Zeldis *et al.* 2010a) . A total nitrogen discharge limit of 800 tonne N per year has been set for finfish farms in the CMFZ.

3.1.2. Modelling procedures

The hindcast dataset from the hydrodynamic model contains current data for 40 depth layers and any location within the model domain. These data can in turn be used to drive DEPOMOD and assess deposition for a given feed input. A graphical user interface (GUI)[7] has been developed in MATLAB to facilitate extraction of current and bathymetry data from the hydrodynamic model outputs into a format suitable for use by DEPOMOD (Figure 3). DEPOMOD is then used to generate 2-D estimates of the level of solids deposition or flux among locations, and in turn estimate the level of ecological effects for setting initial feed levels within pre-determined acceptable limits.

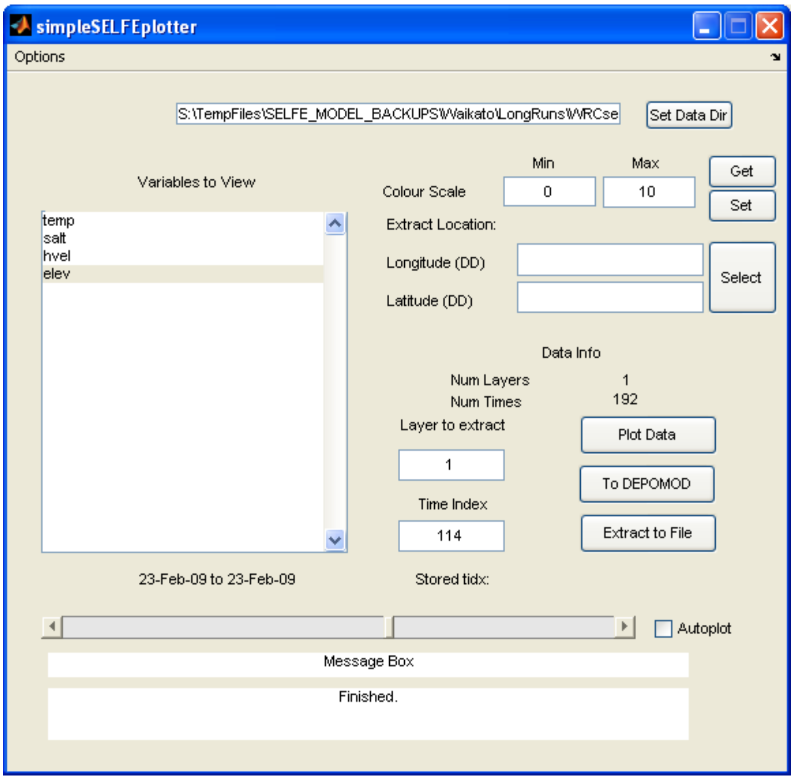

Figure 3. The ‘Simple SELFE Plotter’ assists in visualising raw model outputs and extracting outputs into an appropriate form for investigating seabed effects using DEPOMOD and for visualising data outputs from the hydrodynamic model.

[7] The use of front-end tools provided to WRC staff will be detailed in a separate user manual. The user manual will be a ‘living’ document that will be updated as improvements are made to the models and tools.

Forty vertical layers and two years of data are available from the hydrodynamic model dataset; however, DEPOMOD (version 2.2) is restricted in both the number of layers and number of time periods it is able to process. Consequently, the length of time that is able to be modelled by DEPOMOD is limited to about one month at an hourly resolution[8] and five depth layers[9]. Extracted model data can be repeated to synthesise a modelled footprint for a longer period.

The application of DEPOMOD to determine an appropriate feed loading for a given depositional flux involves many iterations of the model over a range of feeding levels. In order to assist new DEPOMOD users with determining suitable initial feed loadings for different sites and cage configurations, a simple spreadsheet calculator, 'QuickFeed', has been developed. QuickFeed utilises pertinent site characteristics (*i.e.* flow, cage dimensions and depth) to provide preliminary estimates of feed loadings for a given depositional flux. Additional details on QuickFeed are available in Appendix 4 of this report.

The QuickFeed calculator has been designed to be used to help determine an initial feed estimate before beginning an iterative optimisation process in DEPOMOD[10]. The final feed estimates determined from DEPOMOD results, can then more accurately assess the magnitude and spatial extent of depositional flux. As discussed in Keeley *et al.* (2013a), the likely ecological effects (*e.g.* enrichment stage) can be related back to the depositional flux estimated by DEPOMOD.

The steps involved in carrying out an estimation of seabed flux from DEPOMOD to undertake a basic initial evaluation of a proposed site include the following:

1. Acquire current (velocity) and bathymetry data from hindcast datasets and the SELFE model for desired location.
2. Check mean current speeds to classify site as either low- or high-flow. For example, if current speeds at 2 m above seabed are less than 10 $cm.s^{-1}$, then the site is considered a 'low-flow' site.
3. Use proposed feed loading to predict size, shape and intensity of footprints using DEPOMOD.
4. Refer to relevant Depositional flux – Enrichment stage relationships described in Keeley *et al.* (2013a) to estimate approximate size intensity of effects[11].

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[8] This is dependent on the time step used, this is recommended to be 60 minutes to ensure that currents are suitably resolved whilst allowing a sufficient period of time to cover a representative range conditions.
[9] Whilst the depths are preferably distributed evenly from the surface to the seabed through the water column, the near-seabed layer should be located as close to the seabed as possible without being on the seabed. In the SELFE S or sigma vertical grids, this will usually be the 2nd depth cell.
[10] The QuickFeed calculator has not had extensive calibration and should not be used as an assessment tool in isolation.
[11] As a reference, typically seabed fluxes of 05. – 1 kg m^2 yr^{-1} (depending on flow) is an approximate threshold for inducing measurable effects.

A similar process to that above can be used to gauge an appropriate initial feed level for a given farm if there are pre-specified benthic environmental quality standards, *i.e.* a maximum permitted level of effect beneath the sea pens. This would include the following steps:

1. Follow Steps 1 and 2 as above.
2. Use Keeley *et al.* (2013a) to determine a low flux of solid organic matter to the seabed (*i.e.* 95%ile less than required ES) and extreme flux of solids to the seabed (5% less than accepted ES upper limit) for high/low flow site using 6-month no-resuspension curves (see Box 2 and Figure 4 for further details).
3. Use QuickFeed flux calculator to approximate safe and upper extreme feed loading range for subsequent modelling in DEPOMOD.
4. Use initial feed loading bands to predict size, shape and intensity of footprints using DEPOMOD for the two feed scenarios.
5. Use these initial results to guide further scenarios — run and refine until objectives are met with respect to ES magnitude and size of effect.

As an illustrative example for this report, the process has been applied to a site in the CMFZ which contains eight sea sea penpens. The example has been designed to illustrate the process for determining an initial maximum sustainable feed loading level that will not exceed a maximum under pen enrichment score of 5.0.

Box 2 Notes on resuspension in dispersive environments.

In high current environments organic matter can be resuspended, allowing it to be spread over a wider area of the seabed and reducing the worst-case impacts in the immediate vicinity of the seapens. Resuspension can either be included or omitted during DEPOMOD runs. When resuspension is included at dispersive sites, the model may predict no net downward flux and hence no effects, whereas experience to date from high-flow sites has demonstrated that benthic effects still occur (Keeley *et al.* 2013a,b). Hence, the resuspension scenario cannot be meaningfully used to predict the spatial extent of the effects at dispersive sites.

Although unrealistic at dispersive sites, using no-resuspension in DEPOMOD runs can still be used to predict a 'primary footprint' (or downward flux, exclusive of any subsequent resuspension). The spatial extent of the footprint may be underestimated when resuspension is ignored, however results to date from high-flow farms suggest the variation may not be large. Keeley *et al.* (2013a) provide a useful relationship between predicted flux with no resuspension and observed ecological effects in the immediate vicinity of the farm (Figure 3), but note that this still remains an area of active research. Whilst we recommend that worst-case enrichment levels are used as a threshold due to the relative ease of monitoring and interpreting effects of farming operations, ultimately the choice of modelling will depend on the determination of acceptable limits.

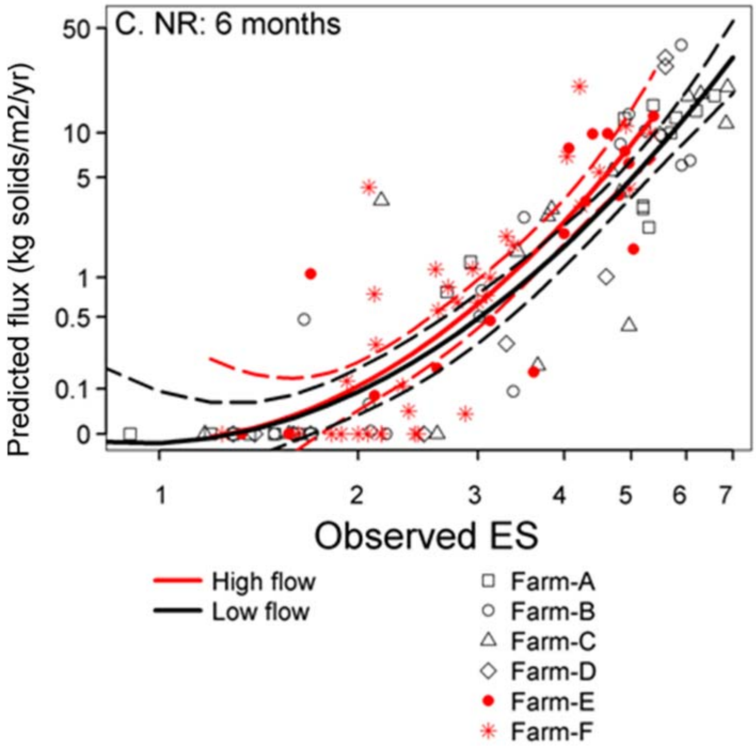

Figure 4. An example of the predicted relationship of modelled flux (y-axis) to enrichment score (x-axis), modelled over six months with no resuspension (from Keeley *et al.* 2013a).

3.2. Water-column modelling

The distribution (spread) of dissolved compounds (nutrients) has been simulated through release of passive tracers during the generation of the 2-year hydrodynamic hindcast dataset. Passive tracers were released at locations likely to be developed for finfish aquaculture and within an approximated volume of the water column that reproduces the likely horizontal and vertical footprints of potential farm sites. Tracers were also released at three river locations and the ocean boundary. Passive tracers used to assess cumulative effects for this assessment are modelled individually using Eulerian time-adaptive 1st order upwind transport routines within the SELFE model. In this approach, the tracers are modelled as concentrations of dissolved materials which are advected within the model in a similar manner to salinity and temperature.[12]

The use of tracers modelled within SELFE is considered a 'coupled' or 'on-line' modelling approach. An on-line approach is computationally intensive, as all of the hydrodynamic processes must be run at the same time as the transport processes for the tracers.[13] The internal time steps within SELFE are small; transport is therefore more accurate within the on-line approach used than other approaches which use

[12] The difference between the tracers and slatinity/temperature is that they are not directly subject to density effects *i.e.* they do not influence density calculations directly, although temperature and salinity induced density effects may affect the vertical transport and mixing of the tracers.
[13] The alternative approach is an uncoupled or off-line model, an example of this is DEPOMOD.

longer time steps.[14] The issue with using a coupled (or on-line) approach is that it requires many more computations and is slower to run. If faster iterations of water-column processes are required in future, consideration could be given to the application of off-line approaches (*e.g.* Knight *et al.* 2009). Another important aspect of the tracer modelling is that each passive tracer group[15] is modelled individually. This means that when the passive tracer modelling runs are complete, different feed loading scenarios can be quickly combined to investigate cumulative effects from several aquaculture sites, or future land-use scenarios.[16]

In order for the tracers to be relevant for calculating the concentrations of nitrogen from aquaculture, the raw tracer concentration results from the model must be scaled into real-world units. Scaling from tracer to nitrogen concentration units requires estimates of the conversion of feed to nitrogen. This in turn depends on the type of feed being used[17], estimates of wastage and the species specific feeding efficiency defined by a ‘feed conversion ratio’ (FCR). The FCR is the ratio of wet weight fish produced for a given dry weight of feed provided. This can be defined as a biological feed conversion ratio (BFCR) which is equivalent to the measured physiological performance of the fish for a given feed input, or an economic feed conversion ratio (EFCR) which is a measure of the operational performance of a farm. The EFCR ignores fish mortality and feed wastage in its calculation and will tend to be higher. The BFCR range is 1.22–2.17 for kingfish (*Seriola lalandi*) and EFCR is 1.30–2.62 (Zeldis *et al.* 2010a). An FCR of 1.7 has been deemed an approproptiate limit for kingfish (*Seriola lalandi*) (Zeldis *et al.* 2010a). Assuming standard New Zealand salmon feeds are fed to kingfish[18], we estimate this gives a total nitrogen (TN) loss of about 56 kg TN per tonne of feed or 95 kg TN per tonne of fish produced (Table 2).

[14] The timestep for the transport of the tracers is based on an adaptive scheme to ensure Courant-Friedrichs-Lewy conditions for numerical stability are achieved (see Zhang and Baptista, 2008 for details). In the model presented here, this means the transport timesteps are at most 60 seconds, but will generally be much smaller than this.

[15] Where a ‘group’ is an individual cage or farm in the case of aquaculture tracers, or a river in the case of land or riverine simulated tracers.

[16] Land-use scenarios will require results for rivers, if available.

[17] The percentage protein content in the feed is particularly important, as this will be proportional to the nitrogen content.

[18] With about 45% dry weight protein content.

Table 2. Estimates of total nitrogen (TN) and dissolved inorganic nitrogen (DIN) lost per tonne of kingfish produced and per tonne of feed.

Description	Result
Feed conversion ratio (FCR)	1.7
Percentage protein in feed	45%
Feed N (kg N/ton of feed, 16% N in protein[1])	72
Fish N (kg retained N/tonne of fish[2])	27.20
Feed N (kg N/tonne of fish produced)	122.40
Faeces production (kg Solids/tonne fish, 26%[3])	442
N % in faeces (salmon)	4%
Faeces N lost (kg per tonne of fish[4])	17.68
DIN excretion (kg DIN per tonne of fish produced)	77.52
DIN excretion (kg DIN per tonne of feed)	45.60
Lost TN (kg TN per tonne fish)	95.20
Lost TN (kg TN per tonne feed)	56.00

[1]Stead and Laird 2002, [2]Bromley and Smart 1981 as referenced by Gowen and Badbury (1987), [3]Butz and Vens-Cappell 1982 as referenced by Gowen and Badbury (1987) [4]Penczak *et al.* 1982 as referenced by Gowen and Badbury (1987).

3.2.1. Scaling of model output into real-world units

Once the feed to nitrogen estimates have been calculated and the tracer releases are complete, the tracer outputs from the model are scaled into real-world units. This is achieved by ensuring the modelled tracer loading rates match the estimated constant nutrient loading rates expected to be released under various feeding regimes for a given finfish species (Table 2)[19]. In order to convert the model concentrations (MC) into real concentration (RC) units (*e.g.* mg N.m^{-3}), the model concentrations need to be scaled by some factor (a) to match real units, so that:

$$RC = a \times MC \qquad (1)$$

Calculation of the scaling factor (a) can be undertaken in a spreadsheet and requires three key pieces of information:

1. The modelled passive tracer release rate (TRR; tracer units/m^3/s)
2. The model release volume (MV) – the volume enclosed by the element geometry of the model that the modelled tracers are released into[20].
3. The real loading rate (RLR) that the passive tracers will be simulating (*e.g.* tonnes of nitrogen/year).

[19] This report focuses only on kingfish; however, a range of species have been proposed for the Waikato region.
[20] The modelled volume should be the volume at mean sea-level.

From this information a modelled loading rate (MLR) can be calculated as:

MLR = TRR x MV (2)

And the associated scaling factor (a) to be used in equation 1 can then be calculated as:

a = RLR/MLR (3)

For the model calculations undertaken for this report, the TRR has been set to a constant 0.01 tracer units m^{-3} s^{-1}. The release volumes and feeding loads are likely to be different for each site; consequently the scaling factor (a) for each tracer run will be different. The scaling factor calculations are undertaken in a spreadsheet, and are provided in a tabular form to WRC as shown in Appendix 3.

In the case of tracers released at open boundaries (*i.e*. rivers or ocean boundaries), the concentration may be directly specified in the boundary conditions file. This means a pre-calculated concentration could be used in real world units (*e.g* . mean annual nitrogen concentrations) or as a generic tracer unit for scaling after the model runs are complete. In the examples used in this report a generic unit of 0.01 tracer units m^{-3} s^{-1} was applied to the river and ocean boundaries.

3.2.2. Calculation of cumulative water-column effects

To calculate cumulative concentration results (CCR) at a given time (t), a linear combination of the modelled tracer results for each site at the same time is undertaken. For example the cumulative concentration from two or more sources could be estimated as:

$$CCR(t) = a_1 \times MC_1(t) + a_2 \times MC_2(t)\ldots + a_i \times MC_i(t) \qquad (4)$$

A schematic of this process for two sources is shown in Figure 5.

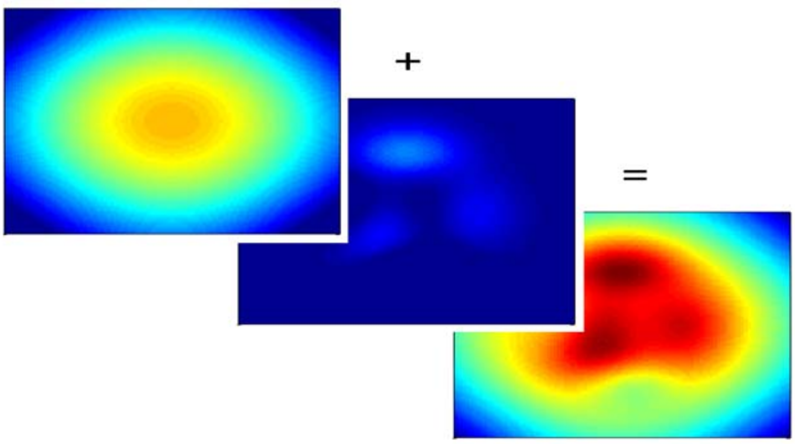

Figure 5. Two example tracer fields (red = high, blue is low concentration) being combined to show a cumulative effect (bottom right plot). This example demonstrates the process that is

undertaken at each time step of the model to generate a cumulative effects layer for each vertical layer of the model.

Because the scale factors (a_i) are proportional to the desired release rates (*e.g.* feeding rates for aquaculture). Different feed loading scenarios can therefore be quickly modelled by calculating appropriate scale factors and adding the results of the relevant tracers together. This requires the calculation shown in equation 4 to be repeated over all cells and times to be investigated. In order to simplify the analysis, the results of aquaculture can be observed in isolation, or for more complex analyses they can be combined with other inputs from rivers or ocean boundaries.

3.2.3. Parameterisation of scenarios

In the main scenario presented in this report, five idealised ‘farm block’ release volumes were parameterised with 56 m radii and a depth of 20 m within the CMFZ. Within one of the large farm blocks, five smaller ‘farm cage’ tracers of 26 m radius were also run as a single combined tracer release[21]. Other sites incorporating main rivers and an oceanic input were also setup and run in the model; therefore the total tracer runs included:

1. Five large ‘farm blocks’ within the CMFZ run as individual tracers.
2. Five indicative ‘farm cage’ releases with a combined area equal to the larger ‘farm block’ within the CMFZ run as a single combined tracer.
3. Two ‘farm block’ sites within the Wilson’s Bay Area C Marine Farming Zone run as individual tracers.
4. Kauaeranga River
5. Waihou River
6. Piako River
7. Ocean boundary.

Details on the exact release locations and radii of the release sites are detailed in Appendix 2 of this report. Nitrogen loadings for example finfish farm scenarios were calculated using a maximum feed loading scenario which was determined by the nutrient and feed limits specified in WRC policy for each region[22]. In the case of the smaller ‘farm cage’ regions, this was parameterised at a hypothetically higher loading rate based on the results of benthic DEPOMOD modelling. All of the aquaculture release sites locations and volumes and feed and TN loading rates are displayed in Table 3. This information is used to estimate scaling factors required to convert model

[21] Approximately matching the surface area of the large 56 m radius ‘farm blocks’.
[22] *i.e.* 800 tonne N/yr in the Coromandel region and 300 tonne N/yr in the Firth of Thames region.

tracer concentrations into feeding scenario specific nitrogen concentrations. The calculated scaling factors are shown in Appendix 3 of this report.

Table 3. Total nitrogen loadings and calculated release volumes or flow rates from SELFE used to convert tracer concentrations to estimate TN concentrations.

Location	Species	Release area (m^2)	Average depth (m)	Volume (m^3)	Feed (kt yr^{-1})	TN load[1] (tonne $TNyr^{-1}$)
CMFZ1	Kingfish	10123	18.54	187642	2.85	159.6
CMFZ2	Kingfish	10117	18.69	189146	2.85	159.6
CMFZ3	Kingfish	9817	19.05	187056	2.85	159.6
CMFZ4	Kingfish	10439	19.32	201741	2.85	159.6
CMFZ5	Kingfish	10167	19.61	199416	2.85	159.6
CMFZ1(1 to 5-small)	Kingfish	10964	19.12	209618	3.63	203.3
Wilson 1	Kingfish	9444	19.10	180342	2.67	149.5
Wilson 2	Kingfish	10170	19.18	195065	2.67	149.5

1. Total N per year calculated based on TN to feed ratio information provided in Table 2 (*i.e*. assuming 72kg TN/tonne of feed).

Nitrogen loadings for rivers and the oceanic boundary were also calculated from available information in Zeldis (2008; Table 4). As these tracers were associated with boundary inputs, the calculation for the scaling factors is slightly different to the aquaculture tracers. With the boundary tracers, the concentration of the incoming water were set to ensure that the model flux approximately matches the total load that should arrive into the model in a year long period. Final scaling factors for the oceanic and riverine tracers are shown in Appendix 3 of this report.

Table 4. Riverine and ocean loadings derived from mean flow and total Firth of Thames nitrogen loading estimates provided by Waikato Regional Council and Zeldis (2005)[1].

Description	Flow (m^3/s)	TN load (tonne $TN.yr^{-1}$)
Firth of Thames	111.06	7,000
Kauaeranga River	7.23	100
Waihou River	67.35	3000
Piako River	22.26	2000
Oceanic inputs (net) [1]	15,030	3600

1. Oceanic flow estimate sourced from Zeldis (2005).

An example water-column scenario was processed for this report and assumes kingfish are farmed with a feed conversion ratio of 1.7 (56 kg TN/tonne of feed; Table 2) under a 'full development' discharge scenario in the Coromandel region only. The full development scenario used an idealised 'farm block' units with all blocks operating at feasible maxima based on council policy limits.

Waikato Regional Council policy has stated yearly rather than sub-yearly feed limits will be used, which means that feed and waste nutrient loading rates may vary considerably over the year without breaching a yearly total limit. Typically, we have observed, during summer months, an approximate increase of 50% above the mean annual feeding rate for salmon farms in the Marlborough Sounds where similar consent requirements exist. The feeding trends also appear to be common in summer months for other farmed salmon species (*e.g. Salmo salar*; Buschmann *et al.* 2007). As a result of this feeding variation, conservative feeding rates applied in the models presented in this report are actually 50% higher than the mean annual rates shown in Table 3. This has been undertaken for our example kingfish farming scenario and models a feasible worst-case impact from the scenario.

3.2.4. Operational tools

The cumulative effects processing is fundamental to the MMM, with tools developed which allow WRC staff to quickly investigate water-column effects from various feed loading scenarios. Before cumulative effects can be assessed the extraction of the raw SELFE data is undertaken using a customised Graphical User Interface (GUI; Figure 3).

After the raw data has been extracted from a given layer from the model, combining water column layers is undertaken in another Matlab GUI entitled the "Cumulative Effects Investigator" (Figure 6). The outputs of this tool are mapped estimates of total nitrogen or any other scaled concentration for which source concentrations can be estimated (*e.g* . larvae, bacteria or pollutants). The outputs of the "Cumulative Effects Investigator" tool highlight areas of waste plume overlaps and retention due to hydrodynamic transport and dilution. This approach is simplistic, but can provide useful insights into areas where nutrient pressures may be high from aquaculture activities before investigating whether more complex biotic modelling approaches should be considered.

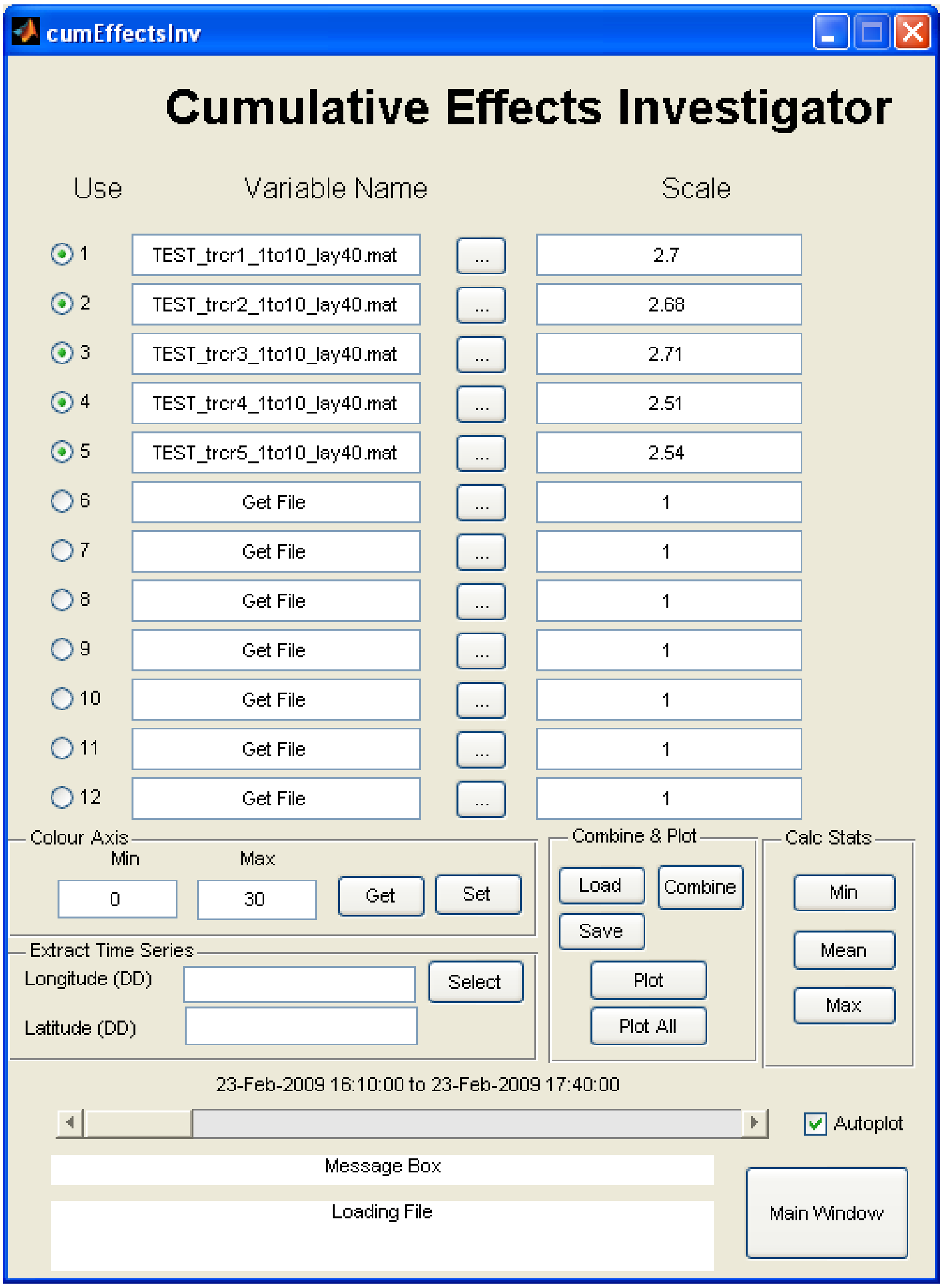

Figure 6. Graphical user interface developed for the Marine Management Model (MMM) to assist in combining model outputs to visualize cumulative effects. The example shows setup for the combination of five different tracer sources from within the Coromandel Marine Farming Zone (CMFZ) to present nitrogen (N) concentration estimates in units of mg/m^3.

4. EXAMPLE MODEL OUTPUTS

4.1. Physical site characterisation

As described in the methods, for the purposes of this report the CMFZ site in the outer Firth of Thames has been selected as an example to demonstrate the modelling steps. Figure 7 provides a spatial interpolated image of the 1 km^2 bathymetric grid that is extracted from the broader bathymetric grid utilised by the SELFE model. This becomes the major grid that is used in the 'GridGen' module in DEPOMOD. The site has a relatively flat bathymetry (sloping SW to NE) in 32–34 m water depth (at chart datum). Earlier site investigations had confirmed that the substrate was comprised almost entirely of soft sediments with relatively low epifaunal diversity (Grange *et al.* 2011).

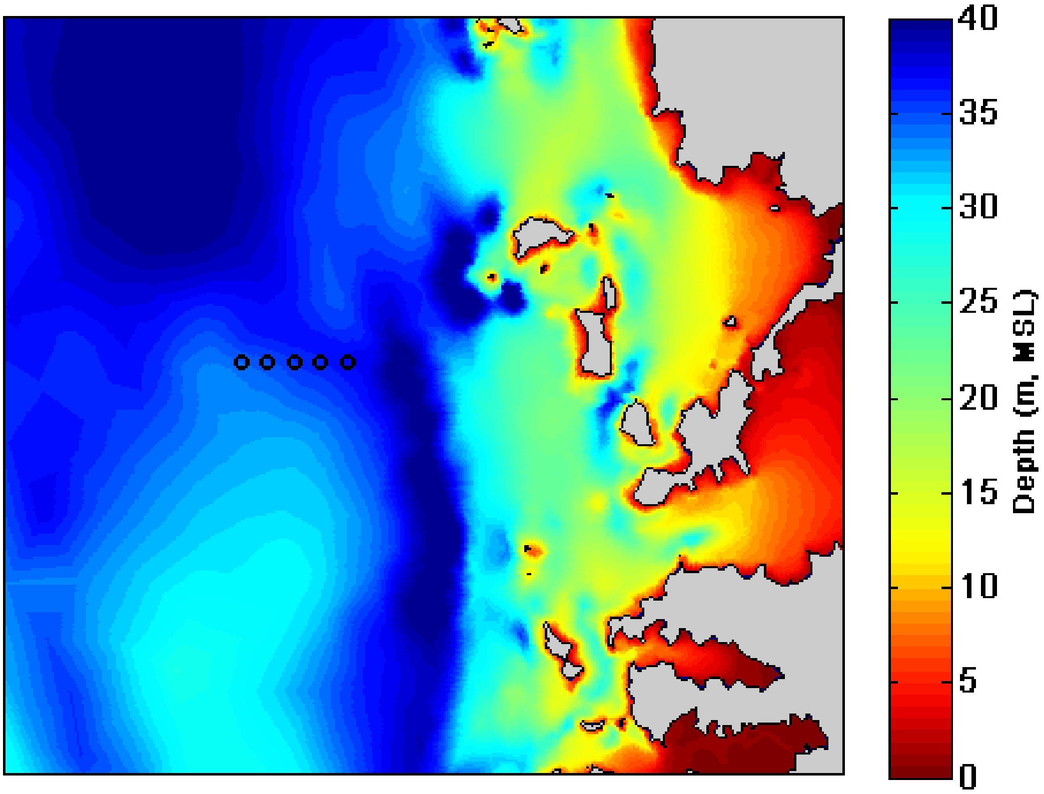

Figure 7. Bathymetry used to generate the major grid for GridGen module in DEPOMOD model. Indicative cage locations are shown for the Coromandel Marine Farming Zone (CMFZ). As can be seen in this image the bathymetry is very flat in this area.

Current data from five depths spanning the water column is also extracted from the SELFE model for use in DEPOMOD. The current speed and directions for three of these depths (surface, mid-water and bottom) are summarised using rose plots in Figure 8. These reveal a tidally oscillating flow between NNW (~350°) and SSE (~170°) with maximum velocities in the range of 40–50 cm s^{-1} near the surface and

20–30 cm s^{-1} near the seabed. Average velocities near the seabed were 12.1 cm $^{s-1}$, which means the currents are faster than the default critical velocity threshold in DEPOMOD (9.5 cm s^{-1}) most of the time; as such it may be considered a high-flow or 'dispersive' site.

The modelled current speed and direction outputs were in very good agreement with independently surveyed currents that have been described for the site (Zeldis *et al.* 2010b). A comprehensive characterisation of the average physical and chemical properties of the water column can also be found in the report of Zeldis *et al.* (2010b).

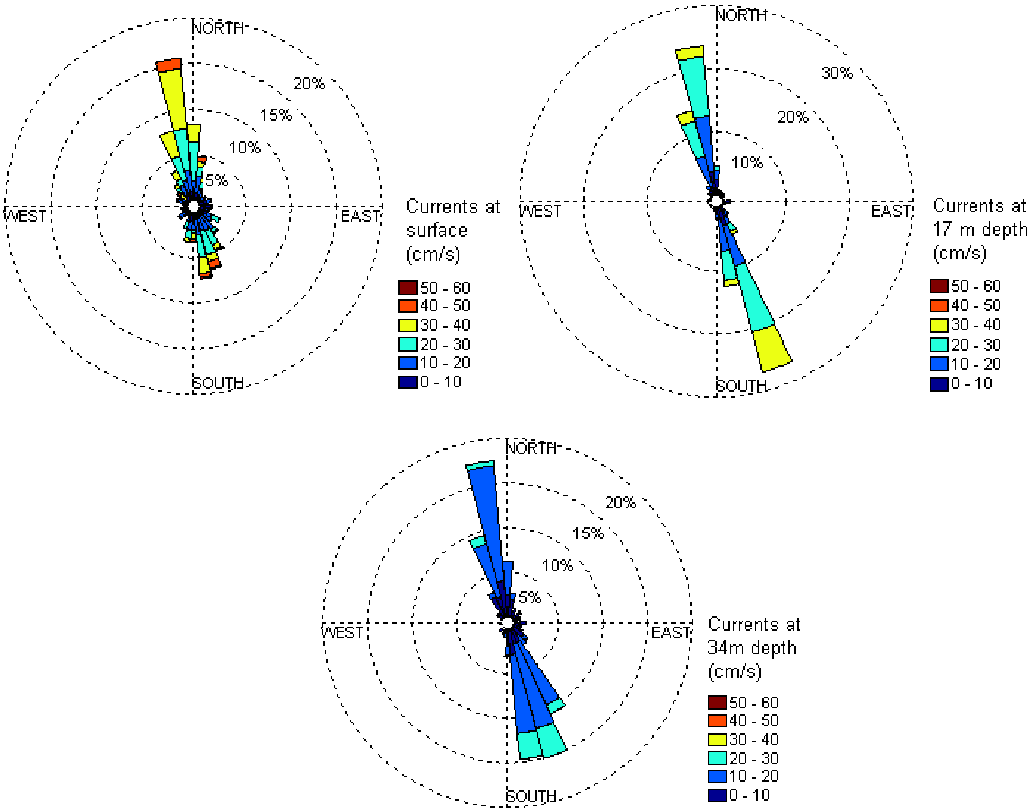

Figure 8. Modelled water currents and directions (going to) for three different depths in model extracted from the model (NZTM 1809376E, 5928040N).

4.2. Seabed

4.2.1. DEPOMOD modelling

DEPOMOD outputs for three feed loading scenarios at the outer CMFZ in the Firth of Thames (Figure 1) show a dispersion of biodeposits by the predominant currents in a NNW and SSE direction from eight modelled farm sea pens. A decreasing gradient of deposition extends approximately 400 m in each direction away from the sea pens. The off-set layout of the seapens was setup in the model to ensure that the depositional footprints did not overlap; this offsetting is beneficial as any overlap would have increased the deposition flux in any areas of overlap. This is an example of the refinements that can be made as a result of some preliminary modelling. Further refinements could be tested by contrasting cage sizes, spacing and other possible arrangements.

In this example scenario, a maximum ES limits of 5 under the farms have been assumed to be acceptable. Model comparisons presented in Keeley *et al.* (2013a; Figure 4) estimate that ES 5 conditions are achieved on average for high-flow sites when depositional flux levels reach ~ 12 kg m^2 yr^{-1}, with a more conservative lower 95% confidence interval of ~ 5 kg m^2 yr^{-1}. The DEPOMOD modelling results show that a 3.1 kt of feed /year feeding limit is possible within the CMFZ site under the ES5 limit assumption (*i.e*. flux < 5 kg m^2 yr^{-1}). If this result could be repeated over five farm blocks, this is higher than the policy limits which would limit feed inputs to 2.6 kt of feed per year for block. This suggests that the farm performance at lower feeding levels should be well within a maximum ES 5 limit and that development of the area would not be limited by seabed effects if an ES5 level and associated areas of effect are determined to be acceptable.

The estimated total size of the primary effects footprint (using a minimum flux threshold of 0.5 kg m^2 yr^{-1} capable of inducing measurable ecological effects, Keeley *et al.* 2013a) at a feed level of 3.1 kt of feed per year is approximately 22 ha (ignoring resuspension). These dimensions can be used to guide the establishment of initial benthic monitoring zones and compliance boundaries.

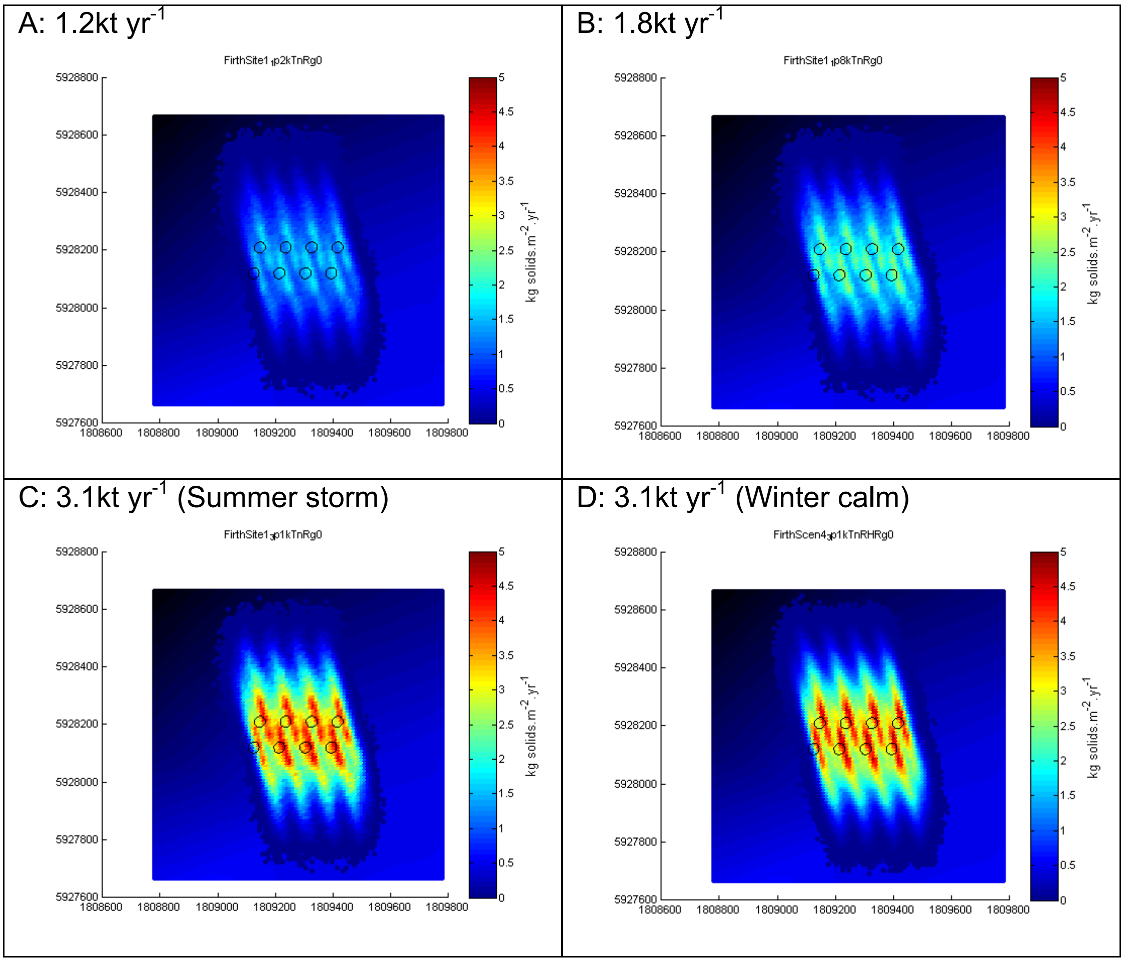

Figure 9. Predicted depositional area (ha) and magnitude (kg m^{-2} yr^{-1}) of bio-deposition for three feed levels (1.2, 1.8 and 3.1 kilotonnes (kT) yr^{-1}) without resuspension in DEPOMOD, and in 'summer storm' and 'winter calm' current scenarios (from the SELFE model outputs) for the highest modelled feed level (3.1 kT per year).

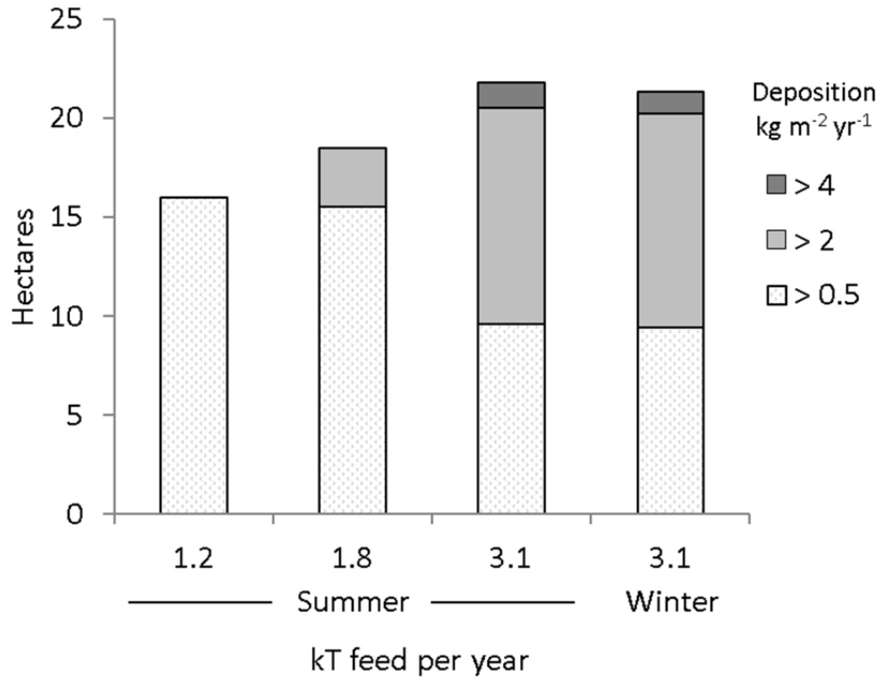

Figure 10. Predicted depositional area (ha) and magnitude (kg m^{-2} yr^{-1}) of bio-deposition for three annual feed levels (kilotonnes (kT) yr^{-1}) without resuspension in DEPOMOD, and in ‘summer storm’ (Firth 3) and ‘winter calm’ (Firth 4) current scenarios (from the SELFE model outputs) for the highest feed level (3.1 kT per year).

Table 5. Predicted depositional area (ha) and magnitude (kg m^{-2} yr^{-1}) of bio-deposition for three feed levels (1.2, 1.8 and 3.1 kilotonnes (kT) yr^{-1}) without resuspension in DEPOMOD and in 'summer storm' (Firth 3) and 'winter calm' (Firth 4) current scenarios (from the SELFE model outputs) for the highest feed level (3.1 kT per year).

Deposition range (kg m^{-2} yr^{-1})	**Feed level** 1.2 kt yr^{-1}	1.8 kt yr^{-1}	(Summer storm) 3.1 kt yr^{-1}	(Winter calm) 3.1 kt yr^{-1}
< 0.5	16.07	15.65	9.6	9.48
0.5–2	0	2.85	10.93	10.82
2–4	0	0	1.25	1.09
4–6	0	0	0	0
6–8	0	0	0	0
8–10	0	0	0	0
Total (ha)	**16.07**	**18.50**	**21.78**	**21.39**

4.3. Water column

Water column results presented as an example focus on the 'summer storm period'[23] with one month of tracer releases modelled from all proposed sites under a full production scenario in the CMFZ. Mean tracer results from a 4-day period after 26 days of continuous release at each of the five farm blocks have been scaled according to an assumed feed loading per block of 2,850 tonne feed per year (Table 3). The final results were then scaled up by an additional 50%, to account for the possibility of increased summer loadings. The results from the individual tracers were combined to generate estimates of potential mean surface and seabed total nitrogen increases for the region, which are presented in Figure 11 and Figure 12.

The results of the CMFZ analysis show the potential for increases in total nitrogen concentrations at the surface of about 15 mg TN.m^{-3} within about 2 km of aquaculture activities after one month of inputs. Whilst a month-long period is useful for displaying the net-transport of material released from the farm, with an approximate two week residence time, the region is unlikely to be near equilibrium; hence concentrations presented in this example would continue to increase with a longer model simulation. Higher concentrations for the same period modelled here would be unlikely sampled in the real system due to un-modelled nitrogen loss processes; consequently it is difficult to ascertain what a possible long-term increase may look like in the region.

[23] Scenario 3 - Table 1.

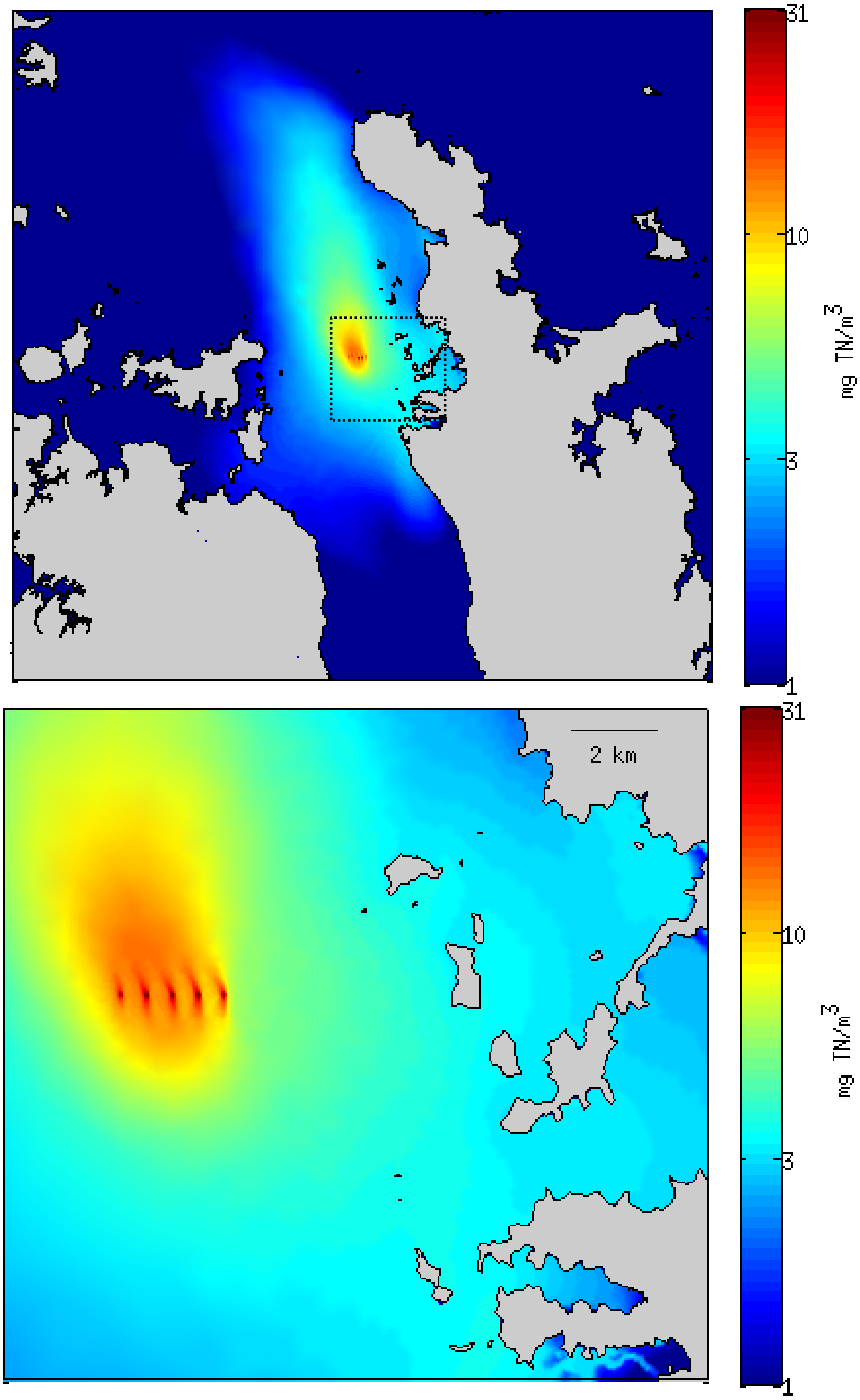

Figure 11. Mean surface total nitrogen concentrations increases from five fully developed sites (14,250 tonne feed/year) over 20–24 March 2009 (mean of 26 to 30 days of release from the 23 February 2009). These initial results suggest a high level of dispersion at the sites with maximum concentrations close to the sea pens less than 30 mg TN m^{-3} (see lower map, zoomed to show release site detail). Note that a logarithmic scale (log_{10}) has been used to present these results.

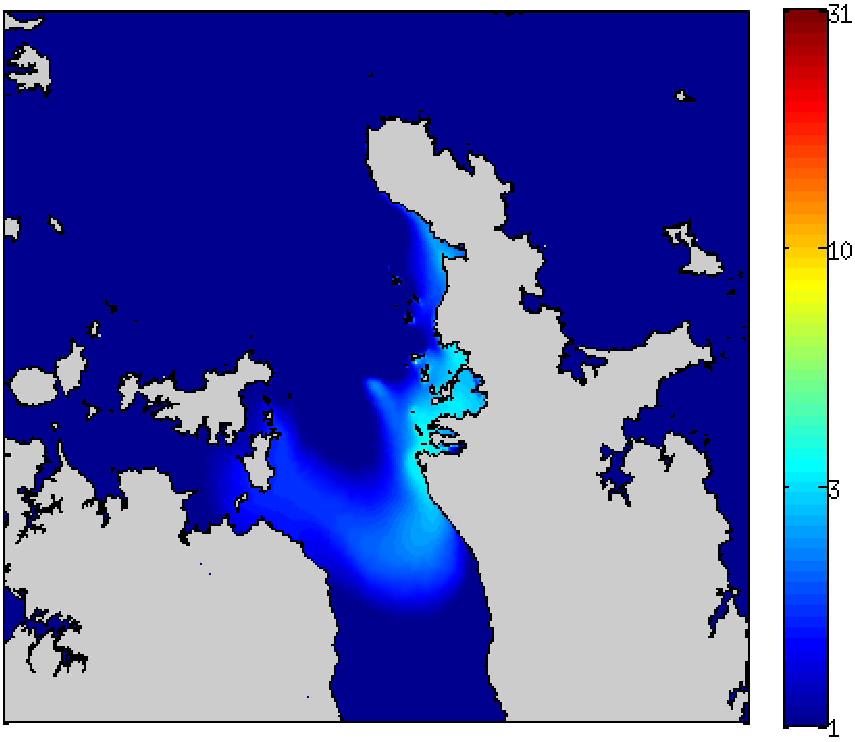

Figure 12. Mean near-seabed total nitrogen concentrations increases from five fully developed sites (14,250 tonne feed/year) over 20–24 March 2009 (mean of 26 to 30 days of release from the 23 February 2009). Note that a logarithmic scale (log_{10}) has been used to present these results and because the seabed varies in depth throughout the region it is not representative of the total nitrogen in the water column.

In order to place the results of the farms into context with other inputs in the region, additional tracer release examples were undertaken to visualise inputs of oceanic and terrestrial sources of nitrogen. Following runs of two weeks and based solely on physical processes, the surface concentrations of total nitrogen downstream of simulated fish farms were an order of magnitude lower than the surface concentrations associated with the region's major rivers (*e.g.* Figure 13). The initial results for the model also show that the Wilson's Bay Marine Farming Zone inputs have the potential to interact with riverine-sourced nitrogen inputs in the Firth of Thames. Additional results for all scenarios are displayed in Appendix 5.

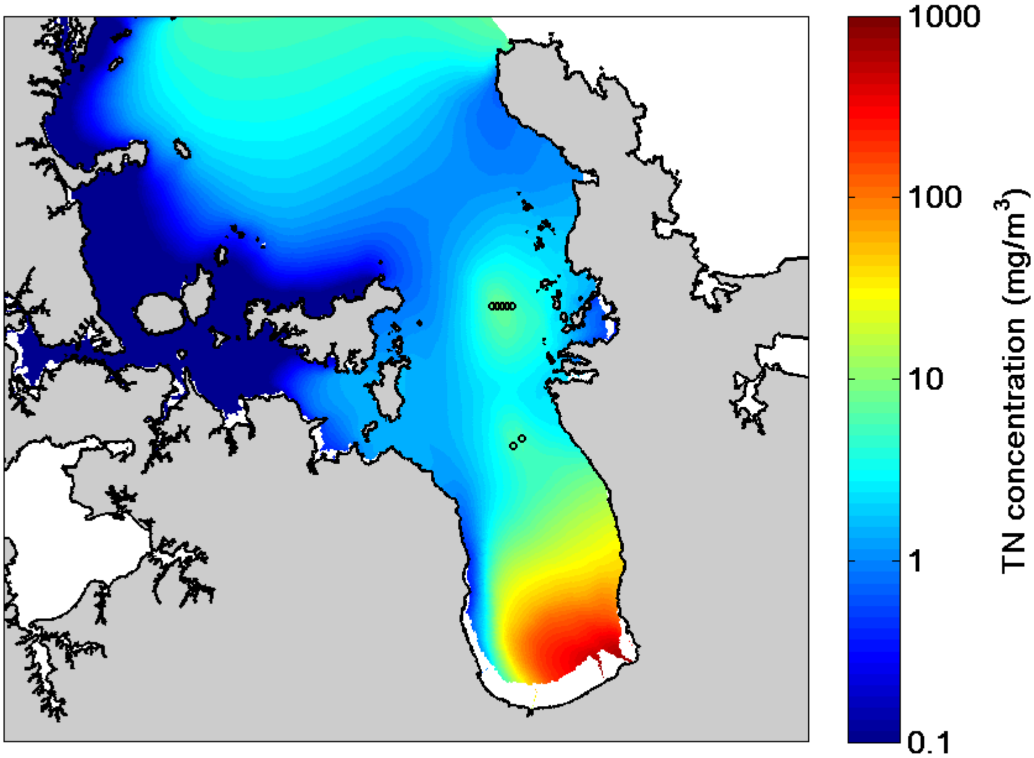

Figure 13. Surface mean cumulative effects over a 2-day period (model days 14 to 16) of all riverine, ocean and aquaculture nitrogen inputs after two weeks of tracer inputs during periods associated with Scenario 4 (13–29 June 2009). Locations of the five Coromandel Marine Farming Zone (CMFZ) sites are indicated by the circles in the middle of the map, with the two southern circles indicating the Wilson's Bay Marine Farming Zone release sites. Note that a logarithmic scale has been used to present these results.

5. CONCLUSIONS

The modelling methods presented in this report are intented to provide first stage assessments of the potential effects of finfish aquaculture on seabed and water-column effects. Initial model results for the water column were based on a maximum allowable nitrogen load scenario and confirm that the CMFZ is a suitable location for considering finfish aquaculture. The model results showed that the deep, high-flow waters present in the CMFZ will assist in mitigating both seabed and water-column effects.

In the case of seabed effects. Initial results indicate that the maximum feed levels could possibly be limited by policy-determined limits in the CMFZ, as higher modelled feed loadings ofor a single block imply that effects would fall below the maximum enrichment stage (ES) limit *(i.e.* ES 5) used in the Marlborough Sounds.

Model outputs for the water column, generated using the cumulative effects investigator tool, assisted in visualising cumulative potential nitrogen loading effects from several finfish farms, major rivers, and the ocean. An initial aquaculture scenario investigated the cumulative effects of five full production blocks in the CMFZ and showed surface nitrogen wastes were well dispersed in the environment after one month of inputs. Additional results for two weeks of inputs also showed that the surface concentrations of total nitrogen downstream of simulated fish farms were an order of magnitude lower than the surface concentrations associated with the region's major rivers. The initial results for the model also show that the Wilson's Bay Marine Farming Zone inputs have the potential to interact with riverine-sourced nitrogen inputs in the Firth of Thames. WRC's decision to limit nitrogen inputs to the region appear to be sensible given the potential for these cummulative effects.

This report illustrated results from a limited number of example scenarios of finfish aquaculture development. Further modelling can now be carried out by WRC staff through the use of the models and front-end GUIs that faciliate extraction and visualision of data from the hydrodynamic hindcast dataset. Modelling is by nature an iterative process, whereby models are continually improved and developed. It is envisioned that the hydrodynamic and aquaculture effects models will continue to be improved. Over time, additional models (or 'modules') contributing to the overall Marine Management Model will be further developed and expanded to encompass more complex processes (*e.g.* chemical and biotic processes, cumulative effects) and issues (biosecurity risk, coastal hazards, oil spills).

6. ACKNOWLEDGEMENTS

Thanks to the following people for their valuable input:

- Hilke Giles and Vernon Pickett — for their thoughts and input into this report
- MetOcean Solutions Ltd. staff, including Brett Beamsley, Greg Pearson, Frank Sonntag and David Johnson — for their huge efforts to prepare the model boundary conditions
- Cawthron Insitute colleagues:
    - Robyn Dunmore, Ross Sneddon and Weimin Jiang — for the data collection and analysis used to develop the model
    - Eric Goodwin — who undertook the DEPOMOD modelling.

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8. APPENDICES

Appendix 1. Weather conditions for scenarios in the middle of the Firth of Thames (37.024ºS, 175.378ºW).

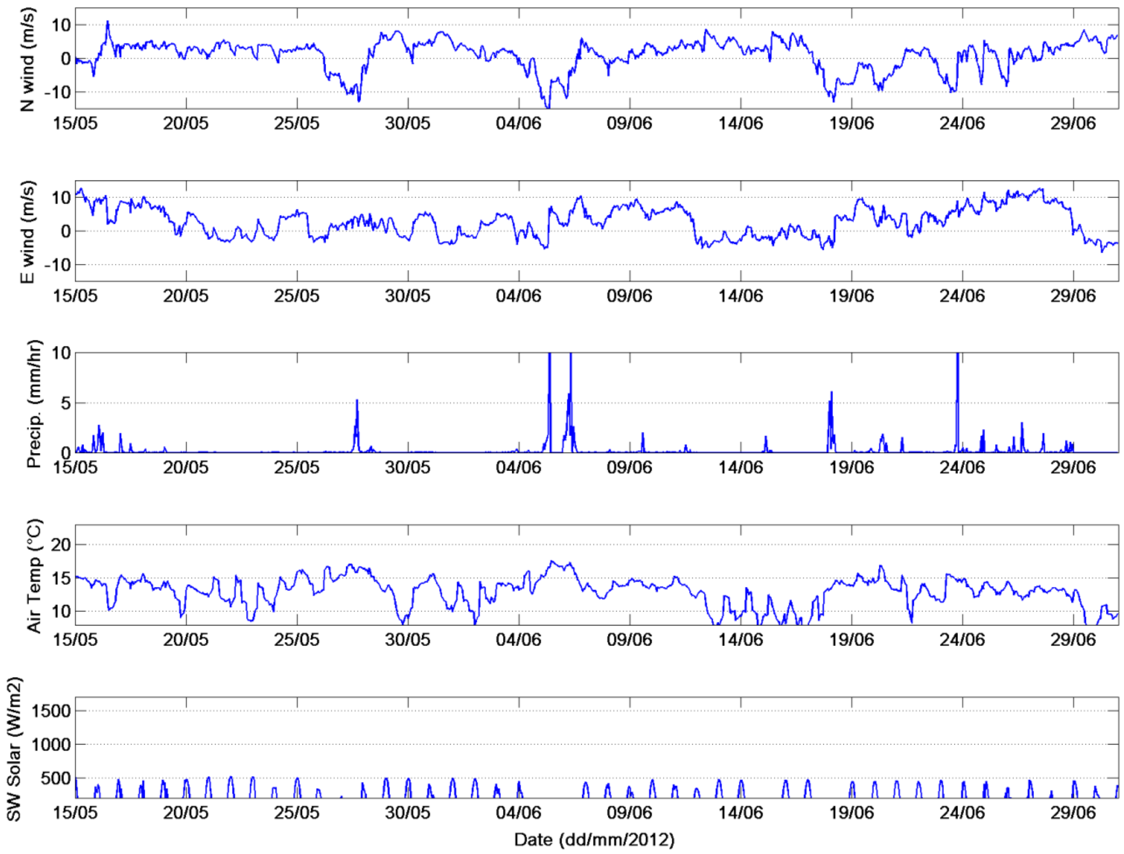

Figure 14. Meteorological data for a 'typical' winter period (Scenario 1).

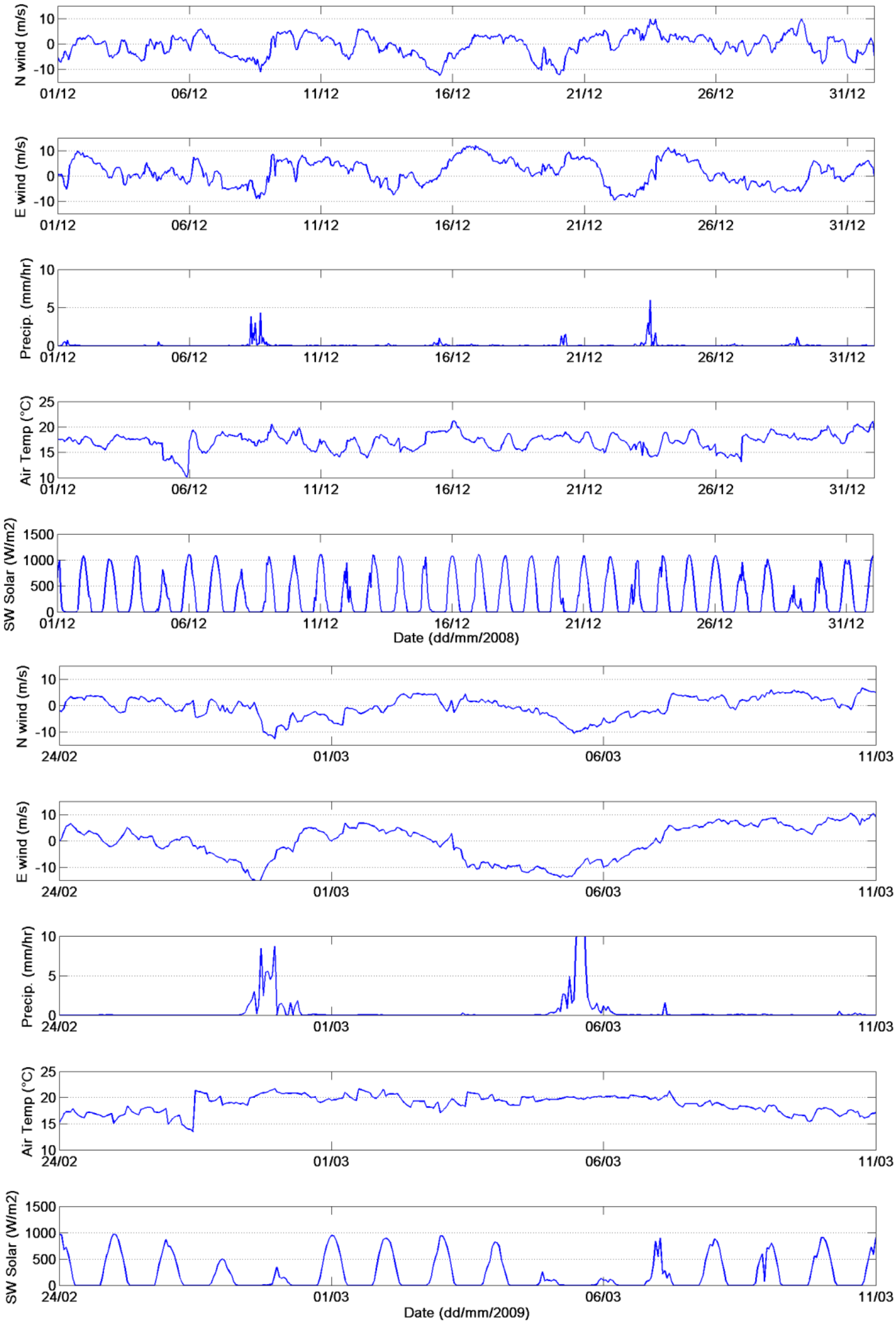

Figure 15. Meteorological data for calm summer (Scenario 2; top) and stormy summer (Scenario 3; bottom) periods.

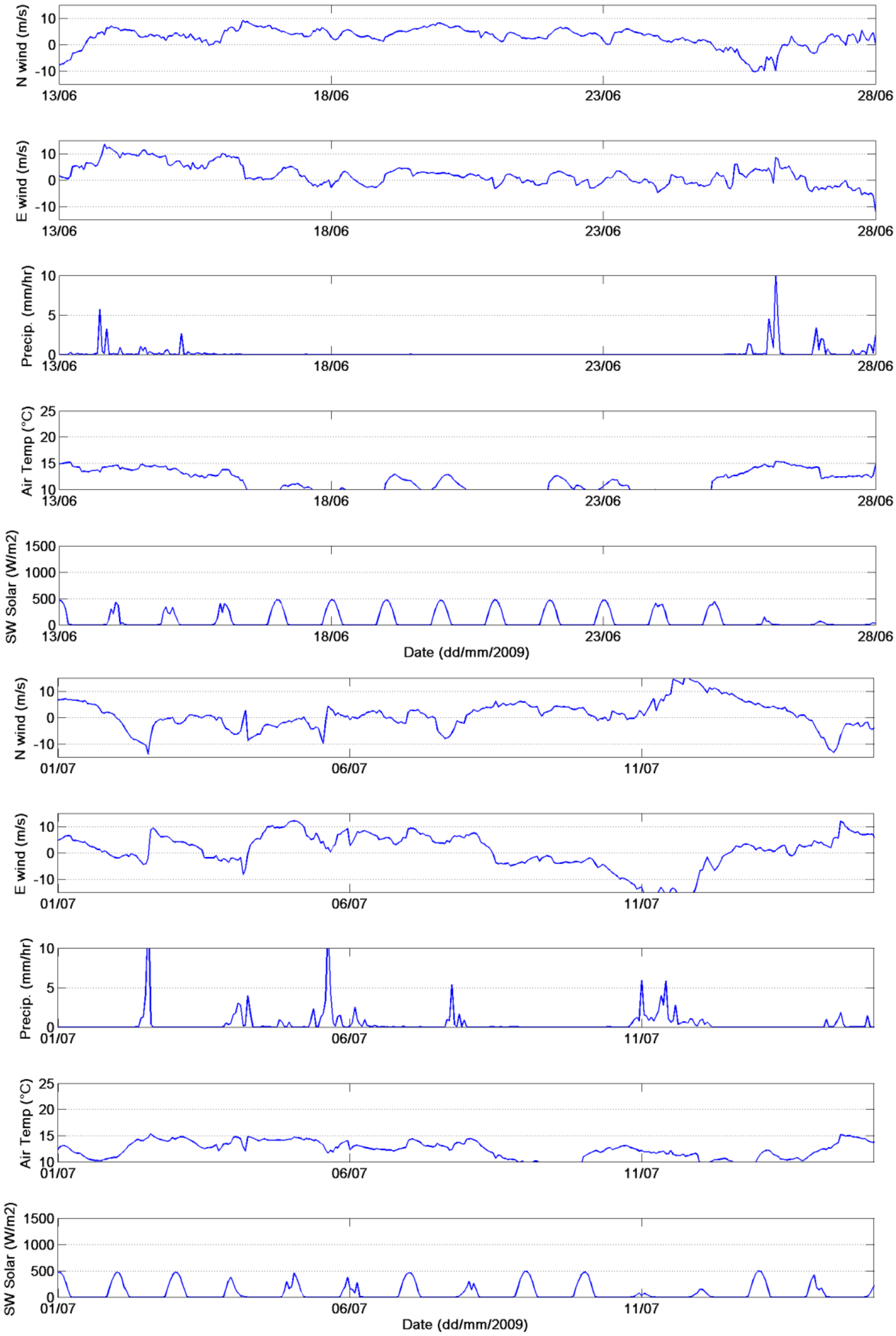

Figure 16. Meteorological data for calm winter (Scenario 4; top) and stormy winter (Scenario 5; bottom) periods.

Appendix 2. Passive tracer release information.

Tracer number	ID	Discharge type	Location	NZTM E (m)	NZTM N (m)	Longitude	Latitude	Release radius
1	CMFZ_1	Fish farm	CMFZ*	1807850	5928220	175.329	-36.769	56.42
2	CMFZ_2	Fish farm	CMFZ*	1808450	5928220	175.336	-36.769	56.42
3	CMFZ_3	Fish farm	CMFZ*	1809050	5928220	175.342	-36.769	56.42
4	CMFZ_4	Fish farm	CMFZ*	1809650	5928220	175.349	-36.769	56.42
5	CMFZ_5	Fish farm	CMFZ*	1810250	5928220	175.356	-36.769	56.42
6	CMFZ_1-1	Fish farm	CMFZ*	1807655	5928545	175.327	-36.766	26.00
6	CMFZ_1-2	Fish farm	CMFZ*	1808045	5928545	175.331	-36.766	26.00
6	CMFZ_1-3	Fish farm	CMFZ*	1808045	5927895	175.331	-36.772	26.00
6	CMFZ_1-4	Fish farm	CMFZ*	1807655	5927895	175.327	-36.772	26.00
6	CMFZ_1-5	Fish farm	CMFZ*	1807850	5928220	175.329	-36.769	26.00
7	Area C_W	Fish farm	Wilson Bay - Area C	1810450	5910700	175.363	-36.926	56.42
8	Area C_E	Fish farm	Wilson Bay - Area C	1811450	5911670	175.374	-36.917	56.42
9	Ocean Boundary	Ocean						Open Boundary
10	Kauaeranga	River	Kauaeranga River					Open Boundary
11	Waihou	River	Waihou River					Open Boundary
12	Piako	River	Piako River					Open Boundary

Appendix 3. Information required to calculate scaling of final concentrations from SELFE.

Tracer number	Location	SELFE conc. rate (trcr/m^3/s)	SELFE volume (m^3)	Feed (kt/yr)	N load rate (tonne TN/yr)	N load rate (g TN/s)	Actual conc. change (g TN/m3/s)	Scale values for output conc. in g TN/m^3
1	CMFZ1	0.01	187642	2.85	159.6	5.06	0.0000270	0.00270
2	CMFZ2	0.01	189146	2.85	159.6	5.06	0.0000268	0.00268
3	CMFZ3	0.01	187056	2.85	159.6	5.06	0.0000271	0.00271
4	CMFZ4	0.01	201741	2.85	159.6	5.06	0.0000251	0.00251
5	CMFZ5	0.01	199416	2.85	159.6	5.06	0.0000254	0.00254
6	CMFZ1-1to5	0.01	209618	3.63	203.28	6.45	0.0000308	0.00308
7	Wilson 1	0.01	180342	2.67	149.52	4.74	0.0000263	0.00263
8	Wilson 2	0.01	195065	2.67	149.52	4.74	0.0000243	0.00243
9	Ocean	0.01	15030.44	NA	3600	NA	0.0075949	0.75949
10	Kauaeranga	0.01	7.23	NA	100	NA	0.4385	43.85
11	Waihou	0.01	67.35	NA	3000	NA	1.4149	141.49
12	Piako	0.01	22.26	NA	2000	NA	2.8490	284.90

Scale = Actual change rate / SELFE change rate

Appendix 4. 'QuickFeed' flux calculator background.

'QuickFeed' is a simple MS Excel™ spreadsheet tool that has been developed to quickly assess an approximate average seabed flux in a region around a finfish farm given some basic information on tidal current flows and water depth. The basic aim of the tool to estimate the total solid flux per annum for a given site.

Estimating benthic flux in QuickFeed

The flux of solids to the benthos (φ) is calculated from the amount of solids (the 'load', S) for a given feed loading and divided by the total areal footprint (Ar_{tot}) likely to be affected, *i.e.*:

$$\Phi = S/ Ar_{tot} \quad (1)$$

The solids load is defined as:

$$S = feed \times \%solids/100 \quad (2)$$

The total areal footprint (Ar_{tot}) is determined from the length of time the solids stay in the water (T_{sink}), a given dispersion rate (k) and the mean, depth-averaged, tidal currents.

Tsink also depends on the depth of the site (d) and the sinking rate (w_{sink}), such that:

$$T_{sink} = w_{sink} \times d \quad (3)$$

The tidal currents are defined based on the mean current for the major axis (U_{major}) and the ratio of the minor to major tidal currents (the 'tidal ellipiticity', e_{tidal}), with the area associated with the tidal ellipse movements of a sinking particle (Ar_{tide}) equal to:

$$Ar_{tide} = U_{major} \times (U_{major} \times e_{tidal}) \times T_{sink} \quad (4)$$

The area associated with dispersion (Ar_{disp}) is calculated as:

$$Ar_{disp} = k \times T_{sink} \quad (5)$$

The total areal footprint is assumed to be approximately equal to the sum of the area of an ellipse prescribed by the mean currents, the dispersion rate and the cage footprint (Ar_{cage})[24]. Consequently the total areal footprint (Ar_{tot}) is estimated to be:

$$Ar_{tot} = Ar_{tide} + Ar_{cage} + Ar_{diff} \quad (6)$$

[24] The cage footprint is the horizontal area directly under the cages as defined by their dimensions.

This information can then be used in equation 1 to calculate an average flux (Φ) over the area, or alternatively be rearranged to calculate flux for a given solids load (S):

$$S = \Phi \times Ar_{tot} \qquad (7)$$

Provided the ratio of solids to feed can be estimated, then an equivalent feed loading can be estimated.

Operation of the QuickFeed Spreadsheet

Input parameters relevant to the calculation of flux need to be provided to the QuickFeed spreadsheet to characterise the species to be farmed and the local environmental conditions (Table 6); input variables are highlighted on spreadsheet in *italics*.

Table 6. Input (*italics*) and calculated parameters for the QuickFeed spreadsheet.

Input parameter	Value
Solids conversion (solids/feed,Butz & Vens-Cappell, 1982)	*26.0%*
Water content of feed	*9%*
Depth (m)	*30*
Sink rate (m/s)	*0.03*
Dispersion rate (m^2/s)	*0.01*
Mean tidal speed (m/s)	*0.15*
Tidal ellipcity (minor to major axis)	*0.1*
Cage length (m)	*38.2*
Cage width (m)	*38.2*
Cage footprint (m^2)	*1,146.08*
Number of pens	*8*
Time in water column (sec)	1,000
Mean tidal excursion (m, major axis)	150

After the parameterisation of the site and environment are completed, the spreadsheet can either be used to estimate the flux from a given level of feed (Table 7) or the feed for a given level of flux (Table 8).

Table 7. Example feed to flux calculation for QuickFeed spreadsheet. Inputs indicated by *italics.*

Parameter	Value
Feed in (tonne/year)	*2,000*
Solid waste (tonne per year)	473.20
Average flux (kg solids/m2/yr)	7.19

Table 8. Example flux to feed calculation for QuickFeed spreadsheet. Inputs indicated by *italics.*

Parameter	Value
Average flux (kg solids/m2/yr)	*6.36*
Solid waste (tonne per year)	52.31
Feed in (tonne/year/cage)	221
Total annual feed all pens (tonne feed/year)	1,769

For the purposes of assessment we recommend the use of the more sophisticated DEPOMOD model for determining flux. QuickFeed has been only been designed to reduce iteration times within DEPOMOD by helping to give some guidance on the initial levels of feed that could be used in initial DEPOMOD runs. In order to do this it should be used in 'Flux to Feed' mode (*i.e.*Table 8) with an appropriate level of flux to be determined based acceptable enrichment stage (ES) scores. Tables relating flux to ES scores presented in Keeley *et al.* (2013) for 'no resuspension' scenarios are able to provide some guidance on the appropriate level of flux to use for a given ES limit.

A thorough comparison of this tool with DEPOMOD has not been undertaken and hence it should not be used as an assessment tool.

Appendix 5. Cumulative effects scenarios for all modelled sources.

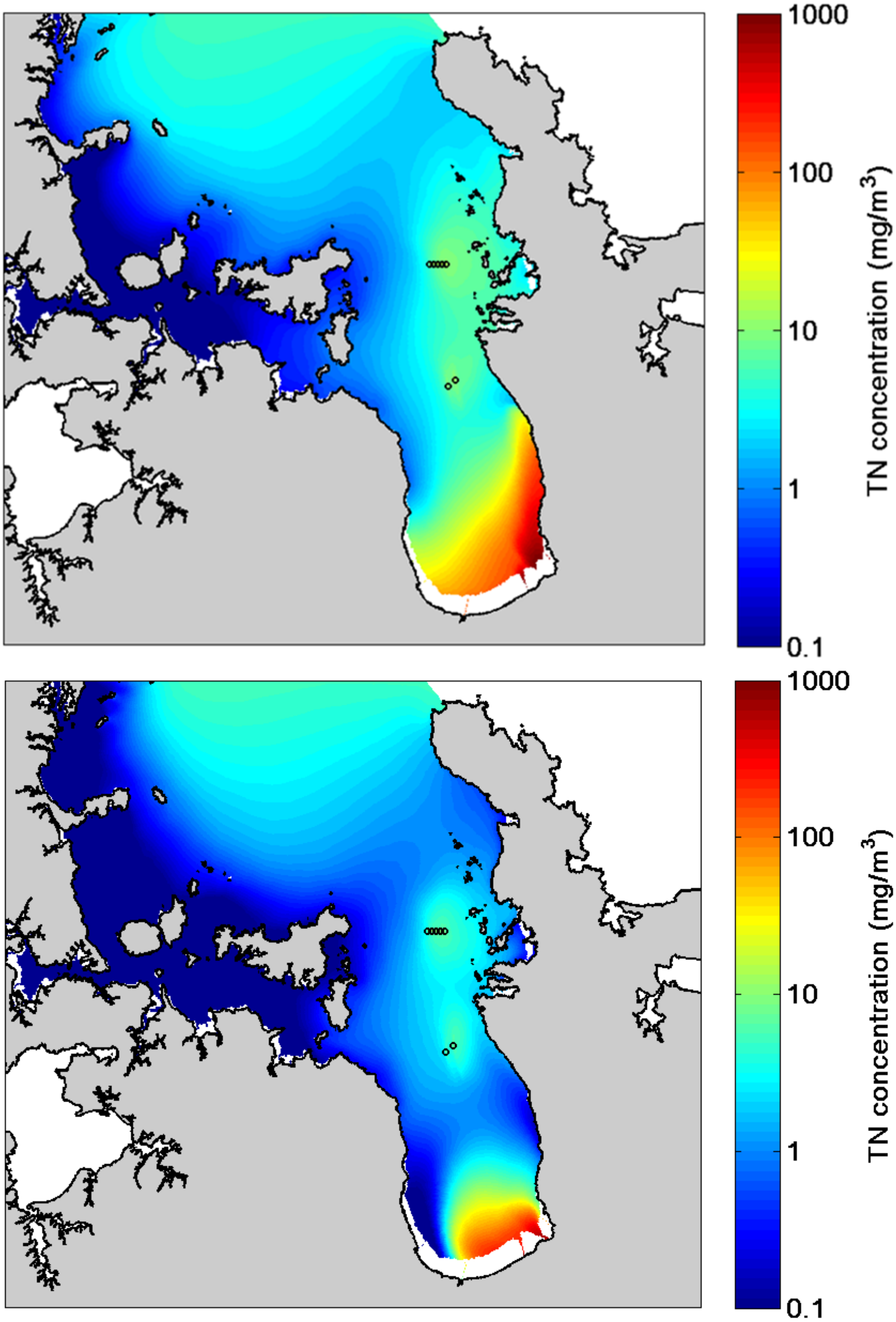

Figure 17. Mean cumulative effects of all riverine, ocean and aquaculture nitrogen inputs after two weeks during periods associated with Scenario 1 (Calibration period 2012) and Scenario 2 (1–16 December 2008). Note that a logarithmic scale has been used to present these results.

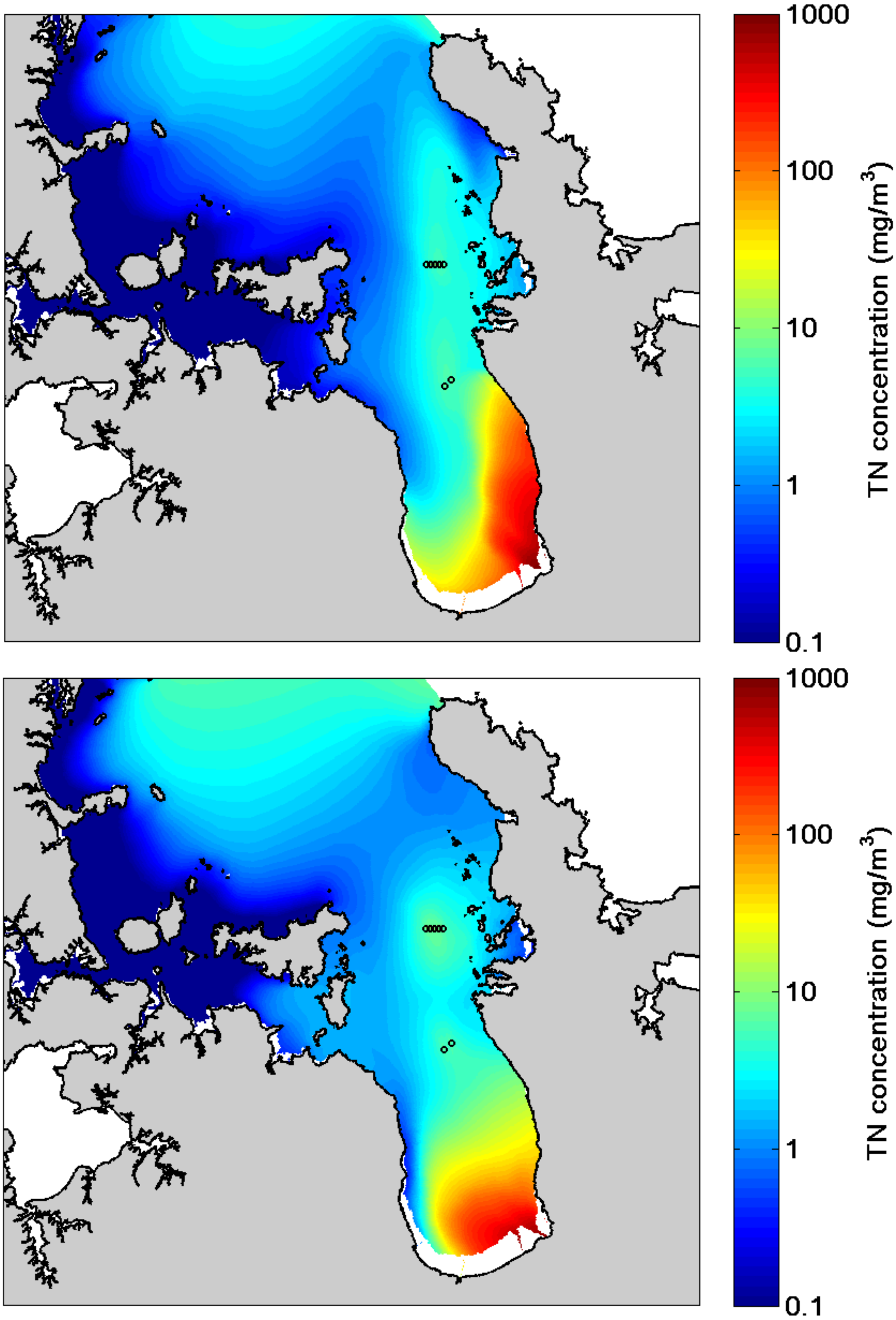

Figure 18. Mean cumulative effects of all riverine, ocean and aquaculture nitrogen inputs after two weeks during periods associated with Scenario 3 (24 February–12 March 2009) and Scenario 4 (13–29 June 2009). Note that a logarithmic scale has been used to present these results.

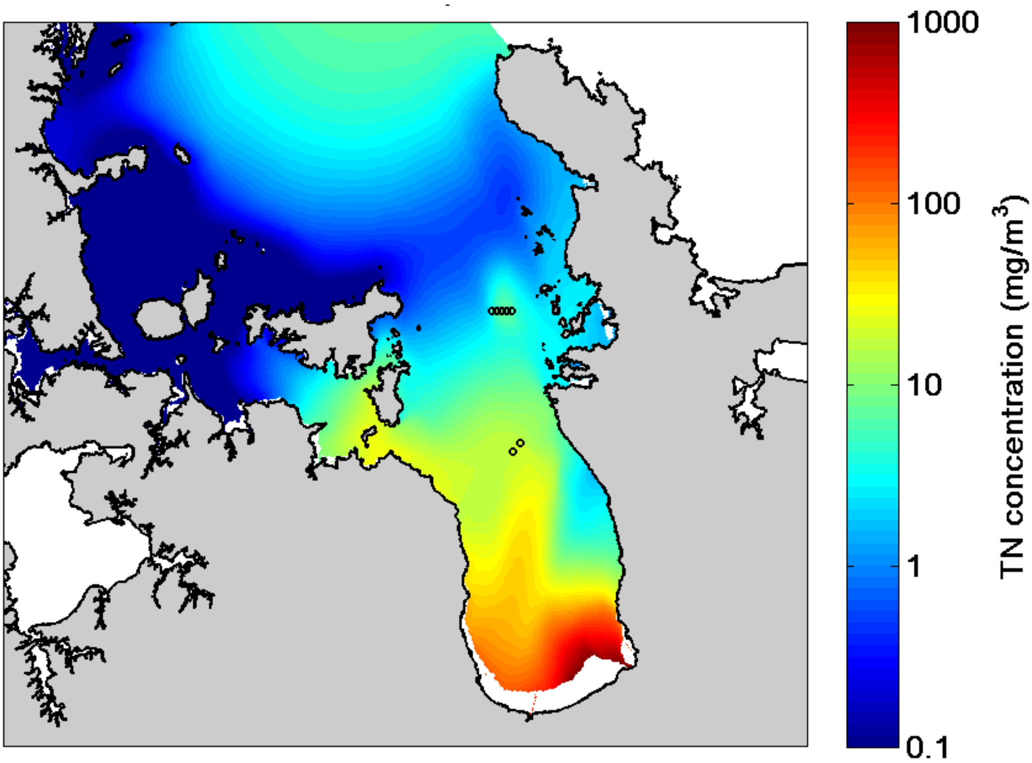

Figure 19. Mean cumulative effects of all riverine, ocean and aquaculture nitrogen inputs after two weeks during periods associated with Scenario 5 (1–17 July 2009). Note that a logarithmic scale has been used to present these results.

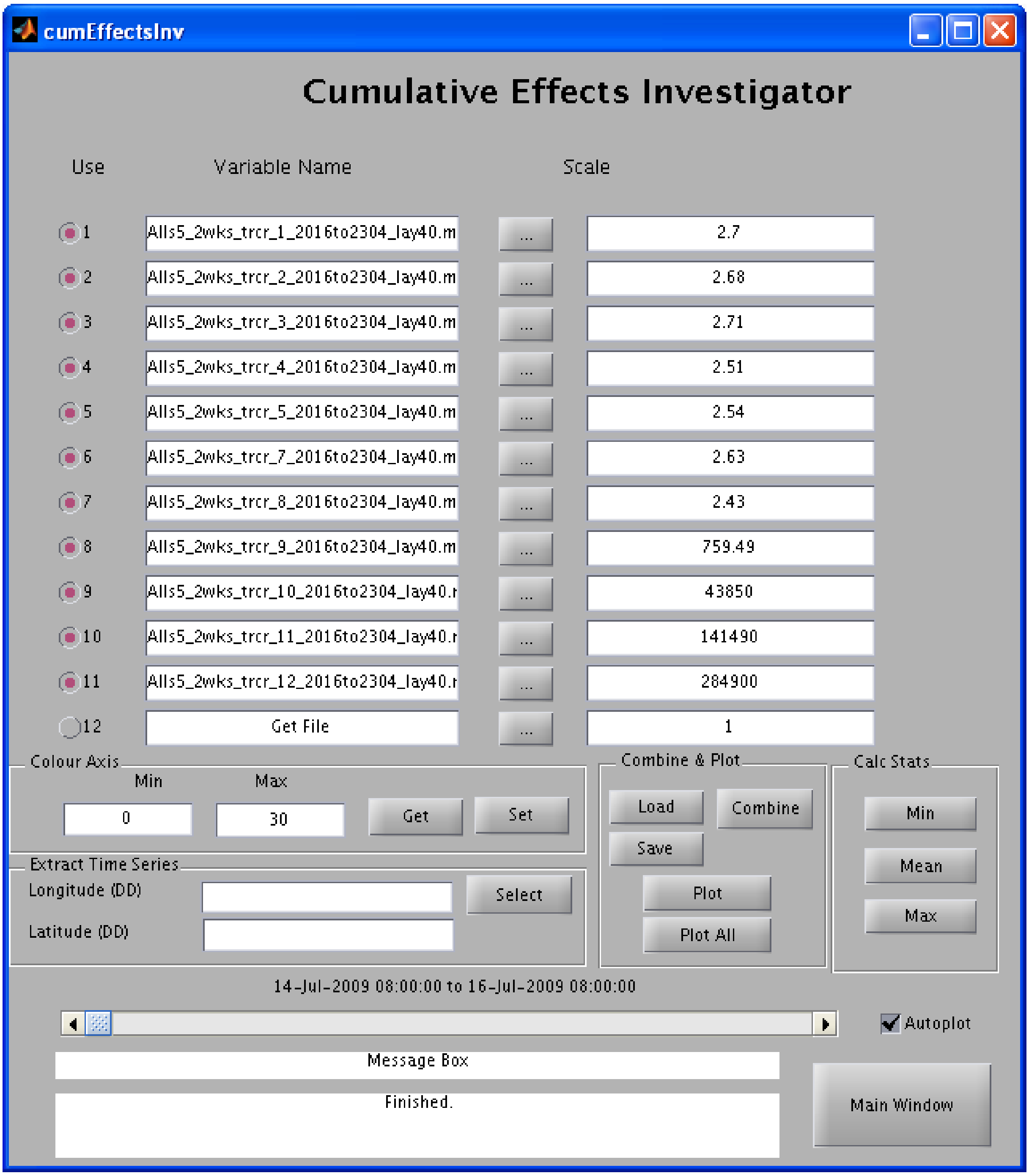

Figure 20. Screenshot of graphical user interface and scaling factors used to generate the cumulative scenario plots.